Environment Agency

Evidence

Uncertainty Assessment of Phosphorus Risk to Surface Waters

Report – SC050035

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Miranda Kavanagh

Director of Evidence

Executive summary

Phosphorus (P) can be a critical driver of freshwater eutrophication but the complexities of terrestrial and aquatic systems make the estimation of P risk to the ecology of surface waters difficult. These complexities include temporal and spatial heterogeneities that make estimates made from observed data uncertain and modelling estimates constrained on such data even more uncertain. A simple relationship between increasing P levels and decreasing ecological status is rare, which has a profound effect on the likely impact of policy decisions made by regulatory authorities. With the implementation of the Water Framework Directive (WFD), an ongoing assessment of the risk to ecology from elevated P levels in lakes and rivers is a priority as is the achievement of proposed water quality standards.

An initial P risk analysis carried out by the Environment Agency (EA) for the first River Basin Characterisation highlighted significant uncertainties in risk estimates made. Since risk characterisation for the WFD is an ongoing process subject to revision, and the monitoring systems required by the WFD are being established, there is an opportunity and a need to review the knowledge and information we have in order to target further research and monitoring.

The specific objectives for this project were to: identify and assess the magnitude of principal uncertainties in the factors that determine the risk from elevated P levels to the ecological status of lakes and rivers; identify areas of academic agreement and disagreement regarding conceptual models diffuse and point source P pressure and relationships with ecological status; provide a consensus regarding an appropriate conceptual model for WFD P risk assessment; evaluate the consequences of uncertainty for UK policies for P management; and to identify phosphorus research priorities based on 1-4 above. To meet these objectives, this study primarily used the process of eliciting expert opinion together with a focussed literature review. The elicitation was carried out using structured one-to-one interviews with experts identified for particular subject areas spanning catchment processes and agronomic practices to the ecological functioning of freshwaters. Experts were asked to provide information in a variety of quantitative/qualitative ways describing their beliefs regarding chosen components of conceptual models. Scale-dependent conceptual models were developed with the experts, which were refined as the project progressed. The information collected during the interview phase was combined with that of the literature review into recommendations for Environment Agency WFD modelling and risk analyses.

For estimation of diffuse P pressure to surface waterbodies, the risks posed by intrinsic catchment characteristics and agronomic practice were explored. The conceptual model components chosen were dominated by physical factors that control hydrological pathways and connectivity of landscape units to surface waters together with livestock management. There was generally a good degree of agreement between experts on which components were included in the model and on the relative importance and uncertainty weightings assigned to each component. In many cases, however, the supporting scientific evidence was identified as being based upon small scale studies such that the changes in relative importance of model components with increasing scale was shown to be not well known, leading to significant uncertainty. P pressure from point sources was considered for wastewater treatment works (WwTW) and septic tank systems. The calculation of point source P fluxes from WwTW is crucial for source apportionment, which serve to isolate the diffuse contribution to catchment P fluxes. The uncertainty associated with point source flux estimates is derived mainly from the data available and there is likely to be a gradual increase in confidence from methods that used generalised, back-calculated per capita export coefficients to those using measured flows and measured concentrations. The latter may now be possible with the current availability of WwTW daily flow data coupled with WIMS P concentration data. Calculation of fluxes in this way would be time-consuming, but would provide robust estimates of point source P fluxes for a number of years.

It is now recognised that septic tanks have the potential to contribute a significant proportion of P to waterbodies in rural areas, although the evidence base to support this is limited, uncertain and sometimes anecdotal. The magnitude of P flux from septic tanks is influenced by tank construction, level of maintenance and location.

With little quantitative evidence of the magnitude of P fluxes (at larger scales) most attempts to quantify them will be highly uncertain and have often been based on published export coefficients or retention coefficients. At all scales, without more quantitative studies that apportion the relative magnitudes of P flux between sources there is likely to be little improvement in our confidence in estimates of P flux from septic tanks. Better quantitative information may allow simplistic rule-based approaches to estimate the likely variability in flux given certain catchment properties. Perhaps one of the most positive directions for future research is the use of tracers, although more progress will need to be made for quantitative source apportionment.

The instream processing of phosphorus is of particular importance for the EA's risk assessments as many of the models currently used to predict diffuse P fluxes make estimates of P delivered to the stream edge and do not explicitly take into account instream processes. Additionally, instream P measurements need to be put into context within their WFD catchments or reaches to assess their representivity. Multiple complex and interacting instream processes act at many timescales from effectively instantaneous chemical reactions to the gradual and stepwise transfer of P and sediment during hydrological events. The significance of these processes, with regard to the risk from P to ecological status, is primarily associated with the net retention, or release, of bioavailable P concentrations during the growing season. The identification of the effect of individual processes is highly uncertain and studies are needed which allow the development of simplistic scale-dependent rules for river reach typologies.

As there are no drinking water standards for P and there are no measures of direct ecological effects in groundwater, surface waterbodies are the ultimate receptors for groundwater P. Principal sources of P in groundwaters (in addition to naturally occurring sources) are primarily from diffuse agricultural sources, point agricultural sources (e.g. leakage from slurry tanks) and quasi-point sources: septic tanks and sewer leakage. Both point and diffuse derived P also enter groundwaters during recharge from rivers. The flux of P to groundwater from terrestrial systems is primarily controlled by the degree of attenuation in soil and drift deposits, further attenuation within the aquifer and the degree of connectivity between the groundwater body and surface water bodies. These controls can be simply described by P pressure estimates and the degree of connectivity between the groundwater and surface water but in reality groundwater contributions can be highly variable often with higher base flow contributions during ecologically sensitive summer low flow periods. The degree of uncertainty associated with estimating groundwater P effects on surface water bodies is determined primarily by the limited observational data and system heterogeneity.

The link between elevated P concentrations and ecological status in rivers and lakes is not simple owing to the many complex, nonlinear and interacting processes that can lead to shifts in ecological status for a given P status. This makes estimates of current and reference ecological status estimates uncertain. The experts consulted generally agreed on the effects that eutrophication had on the abundance and composition of the aquatic plant community and on the factors that control P-ecological indicator relationships. They also agreed that system complexities confounded the identification of simplistic P-ecological indicator relationships and that more holistic approaches should be used rather than one-dimensional dose-response relationships.

Although problematic, the definition of reference and current ecological status is crucial and should be done in a holistic manner using a number of relevant variables combined in such a way to give robust designations. For WFD purposes, where estimates are required nationally, data limitations will often preclude the use of directly observed measures such that some form of modelling will be required. Although uncertain, the modelled estimates do provide indices of pressures on surface waters and their sensitivities that are the first step in focussing measures for environmental protection and highlighting areas for more detailed investigation. Methodologies are required that can combine uncertain indices of relevant factors that can take into account natural variability such that chances of misclassification are minimised. These methodologies are, in some cases, likely to require a higher level of monitoring to support or refute Tier 1 risk estimates and to inform higher-tier modelling approaches aimed at improving confidence in estimates made and providing more specific information for planning and implementation of mitigation programmes.

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1 Introduction

Phosphorus (P) can be a critical driver of freshwater eutrophication but there are many other complexities of terrestrial and aquatic systems that make the estimation of P risk to surface water bodies difficult. These complexities include temporal and spatial heterogeneities which make estimates made from observed data uncertain and any modelling estimates constrained on such data even more uncertain. A simple relationship between increasing P levels and decreasing ecological status is rare, which has a profound effect on the likely impact of policy decisions made by regulatory authorities charged with protecting surface waters. With the implementation of the Water Framework Directive (WFD), an ongoing assessment of the risk to ecology from elevated P levels in lakes and rivers is a priority and achieving proposed water quality standards is potentially the greatest technical and financial challenge for the Environment Agency (Crabtree *et al*., 2007). These type of challenges have led RCEP (1998) to state that:

"No satisfactory way has been devised of measuring risk to the natural environment, even in principle, let alone defining what scale of risk should be regarded as tolerable."

Under the WFD the Environment Agency (EA) has the responsibility to deliver risk assessments[♣] for all surface water bodies in England and Wales. The initial P risk analysis carried out under the first River Basin Characterisation (RBC 1) highlighted significant uncertainties associated with the risk estimates for surface waters, which was mainly attributed to P pressure estimates made. In the case of the RBC1 analysis, uncertainties associated with the link between elevated P concentrations and ecological status was not, however, considered but which are critical when working towards achieving discrete threshold-based standards.

Since risk characterisation for the WFD is an ongoing process subject to revision, and the monitoring systems required by the WFD are to be established, there is an opportunity and a need to review the knowledge and information we have in order to target further research and monitoring. In addition, there is a need to refine risk assessment methodologies to be explicit about the level of confidence water body risk designations and the identification of catchments, and areas within catchments, which are 'hotspots' with significant contribution to risk.

The specific objectives for this project were to:

1. identify and assess the magnitude of principal uncertainties in the factors that determine the risk from elevated P levels to the ecological status of lakes and rivers;
2. identify areas of academic agreement and disagreement regarding conceptual models diffuse and point source P pressure and relationships with ecological status;
3. provide a consensus regarding an appropriate conceptual model for WFD P risk assessment;
4. evaluate the consequences of uncertainty for UK policies for P management;
5. identify phosphorus research priorities based on 1-4 above.

To meet these objectives, this study primarily used the process of eliciting expert opinion together with a focussed literature review and built upon a number previous studies including:

- the RBC1 diffuse P statement for rivers (Environment Agency, 2004);
- the Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies (NUPHAR PROJECT: Carvalho *et al.,* 2005);
- the WFD (RBC2) Risk Assessment Strategy – Policy Report (RBC/S005 – Feb 2006);
- the RBC2 Method Statement - Rivers at risk from diffuse source pressures of phosphorus (Environment Agency, 2008).

The expert elicitation procedure used is described in Section 2. The elicitation was carried out using structured one-to-one interviews with experts identified for particular subject areas (see

[♣] For the purposes of this report risk assessment/analyses is used in a general sense for EA WFD modelling requirements, rather than specifically for those required for River Basin Characterisation.

Figure 1.1). The approach draws from diverse methods such as: NUSAP (Numeral Unit Spread Assessment Pedigree; Funtowicz and Ravetz, 1990), Bayesian Belief Networks (Barton 2006), fuzzy logic and subjective probabilities (Tversky and Kahneman, 1974), which include mathematical, psychological and social science elements. Experts were asked to provide information in a variety of quantitative/qualitative ways describing their beliefs regarding chosen components of conceptual models. Scale-dependent (WFD scales defined below) conceptual models were developed with the experts; these models were central in the stimulation of conversation and as reference for identification of dominant model components. For these components, factors such as scientific understanding, scale-dependent transferability of scientific principles and data quantity and quality were considered to try and identify the origins of the primary uncertainties. For the quantitative part of the study, experts' opinions were combined to gain an overview of areas of agreement and disagreement. Combination of experts' opinions was achieved in a manner that assured no loss of information and areas of disagreement these were treated as valid uncertainty, rather than seeking to achieve consensus (the combination of opinion is discussed in more detail in Section 2). The information collected during the interviews was used to carry out focussed literature reviews in each of the subject areas (see Figure 1) covered by the project which are reported in Sections 3 to 11. Section 11 draws together the ideas elicited for the modelling and risk analysis for the Water Framework Directive subject area and all other subject areas discussed in Sections 3-10

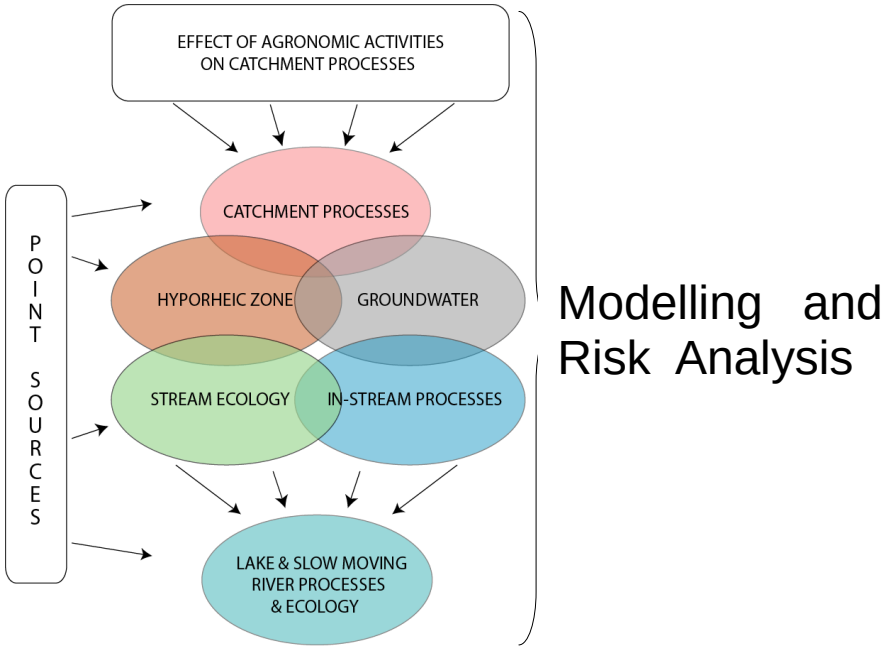


Figure 1.1 – Scope of the project showing individual subject areas which cover primary factors from agronomic perturbations on terrestrial catchment systems to ecological effects in surface water bodies.

2 Expert Elicitation

2.1 Introduction

The stages of the expert elicitation process used for this study are presented schematically in Figure 2.1 illustrating the steps from background work needed for the project to the collation of the elicitation results. This structure was developed using formats and experiences published in previous studies including: Clemen and Reilly, 2001; Cornelissen *et al*., 2003; Daneshkhah, 2004; Funtowicz and Ravetz,1990; Jenkinson, 2005; Meyer and Booker, 1991; Meyer *et al.,* 2002; Phillips, 1999; Stiber *et al*., 2004; von Krauss *et al*., 2004 and Yu and Park, 2000. Much of the literature regarding expert opinion considers the elicitation of prior probability distributions for use in statistical and/or process-based models. For example, an expert may be asked about the probability of a certain event or the likely value of a model parameter (e.g. Hora and Jensen, 2002; Roman, 2008). In this study we seek to determine expert opinion on the likely magnitude of effect of various phenomena on the risk from elevated P levels to the ecology of surface waterbodies. The opinions sought do not relate to true probability distributions, as in the aforementioned studies, but the theoretical basis and structure of these studies are useful in reducing the many associated potential biases (discussed in more detail below) and in the combination of opinion from multiple experts (Stiber *et al*., 2004; Yu and Park, 2000). Moreover, as we do not consider the elicited opinions to be probability distributions, we have chosen to collect information in a fuzzy manner: as fuzzy distributions (see below for a brief explanation). Although the collection of fuzzy distributions was primarily aimed at showing the uncertainties associated with expert opinion, it is also possible that they could be used as fuzzy weightings for different components in revised risk analyses structures where uncertainties are taken into account.

2.2 Selection of experts

A list of experts was generated by the project board from which one expert was chosen from each subject area (see Figure 1.1, Section 1). These ‘core’ experts were consulted as to which other experts would be most suitable for the study: were possible, three experts were chosen for each subject area (see Table 2.1). Although not compulsory, the core experts were asked to bear in mind that, in order for the study to appear unbiased, their peers would not normally come from the same research group/organisation and that they may come from academia, consultancies, government organisations etc. Where possible, all three experts in any given subject area were asked, prior to the interviews, if they were comfortable with their peer group and that they were representative of the subject area. In some cases, experts were made aware that it was necessary to consider the breadth of experts’ experience (in particular those that had knowledge that spanned different subject areas) or to choose experts with experience that covered the specific areas seen as gaps in other experts’ knowledge (e.g. experts that were familiar with upland and/or lowland catchment processes).

2.3 Conceptual model development

A conceptual model was developed for each subject area. Initially this was done with the core expert but its form was not fixed and components of the individual conceptual model could be added or removed at any point, including during the interview phase of the elicitation. The conceptual models were developed to stimulate conversation during the interview and help focus on the dominant factors affecting the risk from elevated P levels in the given subject area; they were not necessarily used as a model structure, particularly as it was not possible to show all causal links and feedback loops. Figure 3.2 shows an example of the catchment processes conceptual model developed.

Table 2.1 – Experts consulted during the elicitation, their subject area and their affiliation

Expert	Affiliation	Subject Area	Ref.
John Quinton	Lancaster University	Agronomic Practice	1ag
Paul Withers	ADAS	Agronomic Practice	2ag
Phil Haygarth	IGER	Agronomic Practice	3ag
Louise Heathwaite	Lancaster University	Catchment Processes	1c
Andrew Sharpley	University of Arkansas, USA	Catchment Processes	2c
Dave Nash	PIRVic, Australia	Catchment Processes	3c
Daren Gooddy	BGS	Groundwater	1g
Anonymous	-	Hyporheic Zone	2hz
Jonathan Smith	EA	Hyporheic Zone/Groundwater	1hz & 2g
Colin Neal	CEH	Instream Processes	1r
Helen Jarvie	CEH	Instream Processes	2r
Andrew Wade	Reading University	Instream Processes	3r
Stephen Maberly	CEH	Lake Ecology	1l
Brian Moss	Liverpool University	Lake Ecology	2l
Lawrence Carvahlo	CEH	Lake Ecology	3l
Keith Beven	Lancaster University	Modelling	1m
Steve Anthony	ADAS	Modelling	2m
Colin Reynolds	Consultant/CEH Fellow	Modelling	3m
Linda May	CEH	Point Sources	1p
Eleanor Jennings	Trinity College Dublin[Ξ]	Point Sources	2p
Linda Pope	EA	Point Sources	3p
John Hilton	formerly CEH	River Ecology	1r
Chris Mainstone	Natural England	River Ecology	2re

[Ξ] Now at: Dundalk Institute of Technology, Ireland.

2.4 Interview development

There were a number of factors to be taken into account when developing the structure for the interview phase of the elicitation. These were primarily aimed at reducing potential biases and assuring that useful data was collected. They included the factors listed below:

- Interviewing suffers from the drawback that experts can be encouraged to 'speculate' and hence 'valid' knowledge is not obtained. It is possible that the interviewees may interpret the questions in different ways; interviews can, however, be better than questionnaires as the interviewer is present to ‘normalise’ responses.

- The more general the questions the more likely that they could be ambiguous. Subsequently care must be taken when considering the scope of the questions as the more specific they are, the more likely consistent answers will be obtained.

- Personal bias of the expert.

- *Substantive expertise*, or capability within an individual’s specialized knowledge domain, has no necessary relationship to *normative expertise* or ability to provide coherent and unbiased probability assessments (Kahneman *et al*.,1982).

- When eliciting distributions (e.g. probability distributions) it is important to include a process of feedback to verify that the fitted, or final distribution, is what the expert thinks it is (Oakley and O'Hagan, 2005).

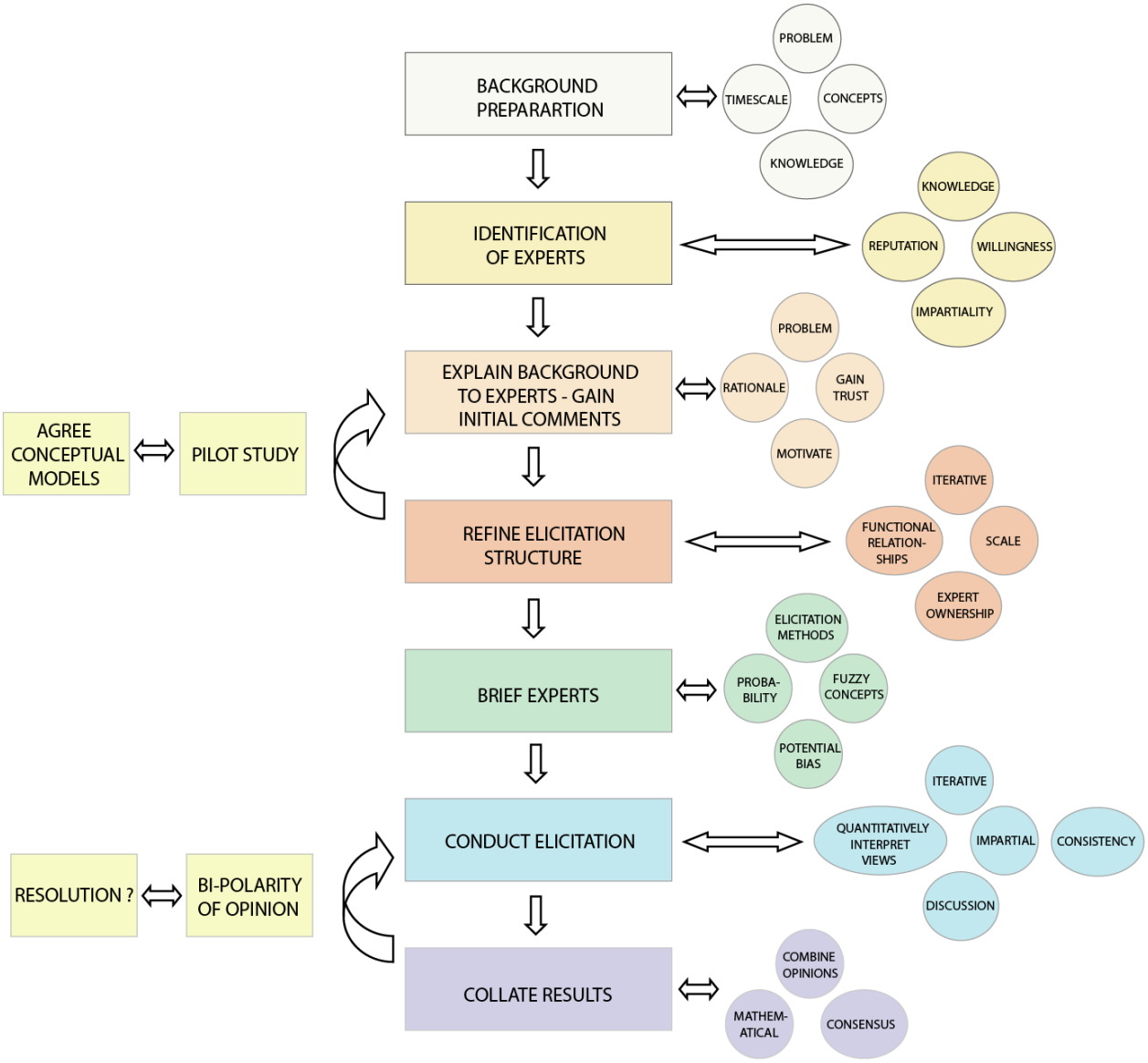


Figure 2.1 – Schematic showing the stages of the elicitation process used for this study

2.4.1 Pilot Study

Pilot studies are particularly useful in determining the number and complexity of questions that could be asked in the time available. They are also useful in identifying where questions need to be made more specific. A pilot elicitation was carried out to identify any problems with the elicitation procedure. There were a number of problems with the initial design that needed to be changed prior to the interviews. The main problems were: each question that the experts were asked needed to be put into context in as much detail as possible otherwise the interviewer would be asked a multitude of questions for every question asked and there were too many questions for the full interview to be completed in the nominal half day. The first problem was remedied by supplying a list of definitions and assumptions (listed in the respective chapter below) for each question, which sometimes included scenarios of available data/information for the risk analyses. The second problem was addressed to some degree by

the remedy to the first problem and also by restricting questioning to Tier 1 analyses and specific P forms.

2.4.2 Interview Structure

In addition to the general considerations above, the interviews were designed under the following constraints:

- Total time given by each expert was approximately 1 day – leaving approximately half a day for the interview phase of the elicitation.
- Quantitative data was to be obtained where possible.
- Qualitative information was required for other project objectives and where quantitative information was deemed unsuitable.

Given these constraints, and the potential problems highlighted above, the interviews were given a formal structure as a formal structured approach to elicitation gives analysts a better idea of conditioning effects (Meyer and Booker, 1991). The structure was, however, flexible enough to give consistency across subject areas but to allow for differences in experts' personalities. The elicitation was designed around the following stages (a detailed structure is shown in Figure 2.1).

I. *Further introduction to the project* (i.e. further to the prior information sent to the experts). Here the specific objectives of the study, with respect to the given subject area, were reiterated. Importantly, the experts were briefed upon the spatial and temporal scale and complexity of the risk assessment(s) concerned (e.g. WFD catchments are generally at a spatial scale of 10-100 km^2). This is of particular importance where experts may have been involved in scientific experimentation at other scales. At this stage experts were asked if the general requirement was clear and if they required any additional explanations.

II. *Assessment of conceptual model components*, with particular focus on missing components. Experts were also given the opportunity to develop their own conceptual models from scratch and to use the conceptual model developed by the core expert as reference only: the conceptual model could be modified at any time during the interview.

III. *Consideration of assumptions and definitions specific to the study*. Assumptions considered included:
   - Spatial and temporal scaling issues;
   - interaction (and overlap) between conceptual models of different subject areas;
   - a definition of the quality and quantity of information/data that may be available for risk analyses;
   - a definition of risk for each subject area (e.g. the risk of elevated P concentrations as a result of increased P fluxes from terrestrial systems (catchment model) or the risk to given ecological indicators from elevated P concentrations (lake and river models).

IV. *Identification of conceptual model components for further consideration*. Components were chosen by the experts in any order. They were asked to consider components that they considered to be important enough to be included in a risk analysis. This initial list was amended iteratively where necessary; experts were informed that the list was not fixed at any point during the interview and could also be changed (along with any associated information collected) post interview phase when interview write-ups were returned for approval.

V. *Informing experts of potential biases in elicitation of opinion*. Of the many potential biases listed above, four, that were thought to be most relevant to the interview, were presented to the experts during the introduction to the elicitation; theory suggests that being aware of these problems may reduce bias:

- *Overconfidence* - Experts tend to be overconfident in their knowledge: this is less of a problem if they are within their own area of expertise (Morgan and Henrion, 1990; Bilal, 2000).

- *Availability* - People consistently overestimate the likelihood of events similar to ones which they have recently experienced (or read about), and underestimate the probabilities of less familiar events (Morgan and Henrion, 1990; Daneshkhah, 2004).

- *Framing* -The phrasing and order of questions. People have been found to overestimate the likelihood of events depending on how the question was framed (McNeil *et al*., 1988; Bilal , 2000; Daneshkhah, 2004). To avoid this bias the inverse question can be asked or asked with regard to positive or negative outcomes.

- *Anchoring and adjustment* - The tendency to anchor assessments on the median, or 'best guess', estimate and stick too close to this value when adjusting to reflect uncertainty: thus systematically underestimating variability or uncertainty (Jenkinson, 2005; Morgan *et al*., 2001; O'Hagan, 1998). This can be overcome by considering the extremes of likely system behaviour.

VI. *Introduction to the concepts associated with the collection of fuzzy distributions*. Fuzzy distributions were elicited so as to retain the uncertainty associated with the experts' opinions in a simplistic manner. The experts were shown a number of example fuzzy distributions and their meaning (linguistically and mathematically) and with respect to the response scales used (see below).

VII. *Collection of quantitative information*. Quantitative information was collected purely as fuzzy distributions. Initial rankings of conceptual model components were retained only as a guide for experts when drawing the distributions. Fuzzy distributions were recorded on answer sheets where response scales were provided for each question.

VIII. *Collection of qualitative information*. Qualitative information was collected as a response to structured questions (see below) and during unstructured conversations. Unstructured conversations occurred during the collection of both structured quantitative and qualitative information and during the review of the information gathered: the interview was audio-recorded. For the Modelling subject area, all information was collected qualitatively.

2.4.3 Question design

Questions were designed to answer the specific objectives defined above. The questions which had quantitative answers were designed to be as generic and relevant as possible across all subject areas. Quantitative questions were used to rank (using the fuzzy distributions as fuzzy weightings) the conceptual model components for importance, overall uncertainty and for the sources of the uncertainty. Qualitative questions were more subject-area-specific and often were aimed at more general effects, which did not lend themselves to quantitative answers: for example the influence of climate/landuse change, scaling issues, system responses.

2.4.4 Quantitative questions

The generic questions were as follows (more specific definitions were available for each question for each subject area):

1. What is the ***importance*** of the chosen model component with respect to its magnitude of effect on the defined risk (all definitions of risk were defined *a priori;* see below).
2. What is the ***overall uncertainty*** associated with predicting the magnitude of the effect of the chosen component (uncertainty from all sources and also related to the data scenario given for each subject area).

3. What is the relative contribution to the uncertainty in 2. above, associated with our current level of ***scientific understanding*** (to clarify scientific understanding it was loosely defined as 'text book' knowledge).
4. What is the relative contribution to the uncertainty in 2. above, associated with the ***practical application*** of our scientific understanding (this question was determined after consultation with experts and relates to the usefulness of our 'text book' understanding is given the spatial and temporal scale of interest and the data availability scenario considered).
5. What is the relative contribution to the uncertainty in 2. above associated with the ***available data*** (given the data availability scenario considered).
6. What is the relative contribution to the uncertainty in 2. above associated with ***spatial heterogeneity*** of the system.
7. What is the relative contribution to the uncertainty in 2. above associated with ***temporal heterogeneity*** of the system.
8. What is the relative contribution to the uncertainty in 2. above from any other source(s) (additional source(s) specified by the expert).

For the quantitative questions laid out in the previous section, an answer sheet was developed (Figure 2.2) which was generic for all subject areas but related to the specific definitions given below. In general terms, all response scales were on a continuous rating scale from low to high but questions 2 and 3 had additional semantic anchors designed to normalise expert response: these semantic scale anchors were modified from the NUSAP approach of Funtowicz and Ravetz (1990). These anchors help reduce problems associated with varying interpretation of the simple semantic anchors such as: high, medium or low. Continuous ratings have three advantages:

1) They avoid discretisation effects in computations.
2) They capture opinion with finer resolution where appropriate.
3) Continuous rating bars are easy to use (Albaum *et al.*, 1981).

2.4.5 Fuzzy Distributions

Fuzzy mathematical techniques are useful for dealing with imprecise data and knowledge where a range of outcomes are possible and hence were chosen for this project. In this case we employed fuzzy distributions in the place of rankings as the use of discrete rankings has the disadvantage that information on the confidence associated with the ranking is lost. Before the fuzzy distributions were elicited the experts were shown the types of distribution that were valid, ranging from very wide flat-topped distributions (e.g. Figure 2.3 a and d) where confidence was low and much narrower triangular distributions where confidence was high (e.g. Figure 2.3 b). Given the high level of uncertainties associated with the relationships considered during this study, experts were advised that triangular distributions and distributions with vertical sides (e.g. Figure 2.3 c) were perhaps too 'precise' and that some degree of flat-top and sloping sides would probably be more suitable. However, experts had the freedom to use any valid distribution as the guidance was aimed at making sure they understood the implications of sharp changes in distributions.

NAME ..... CONCEPTUAL MODEL .....In-stream Processes.....

MODEL COMPONENT .....EXPERT FAMILIARITY RATING (0-10).....

TYPOLOGY .....

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

IMPORTANCE OF COMPONENT

The importance of the chosen component with respect to P loss from a catchment (i.e. the sensitivity of the magnitude of likely P loss to changes in the chosen component)

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

VERY GOOD KNOWLEDGE OF UNDERLYING PROCESSES; ABILITY TO PREDICT MAGNITUDE OF EFFECT ACCURATELY IN SPACE AND TIME.

POOR KNOWLEDGE OF UNDERLYING PROCESSES; INABILITY TO PREDICT MAGNITUDE OF EFFECT ACCURATELY IN SPACE AND TIME.

OVERALL UNCERTAINTY ASSOCIATED WITH COMPONENT

The overall uncertainty (from all sources) associated with our knowledge of the chosen component and its effect upon the magnitude of P loss.

LOW VERY WELL PROVEN GENERAL CONSENSUS | LOW/MEDIUM WELL PROVEN BROADLY ACCEPTED | MEDIUM SOME PROOF SOME ACCEPTENCE | MEDIUM/HIGH LITTLE PROOF EDUCATED GUESS | HIGH NOT PROVEN SPECULATION

SOURCE OF UNCERTAINTY
SCIENTIFIC UNDERSTANDING (1)
Question relates to the source of the overall uncertainty estimated above (relative magnitude).

"TEXT BOOK"

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

SOURCE OF UNCERTAINTY
SCIENTIFIC UNDERSTANDING (2)
Question relates to the source of the overall uncertainty estimated above (relative magnitude).

PRACTICAL APPLICATION

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

SOURCE OF UNCERTAINTY
AVAILABLE DATA
Question relates to the source of the overall uncertainty estimated above (relative magnitude).

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

SOURCE OF UNCERTAINTY
SPATIAL HETEROGENEITY
Question relates to the source of the overall uncertainty estimated above (relative magnitude).

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

SOURCE OF UNCERTAINTY
TEMPORAL HETEROGENEITY
Question relates to the source of the overall uncertainty estimated above (relative magnitude).

LOW | LOW/MEDIUM | MEDIUM | MEDIUM/HIGH | HIGH

SOURCE OF UNCERTAINTY
Question relates to the source of the overall uncertainty estimated above (relative magnitude).
OTHER (SPECIFY BELOW)

.....

.....

Figure 2.2 – Example quantitative answer sheet showing continuous response scales

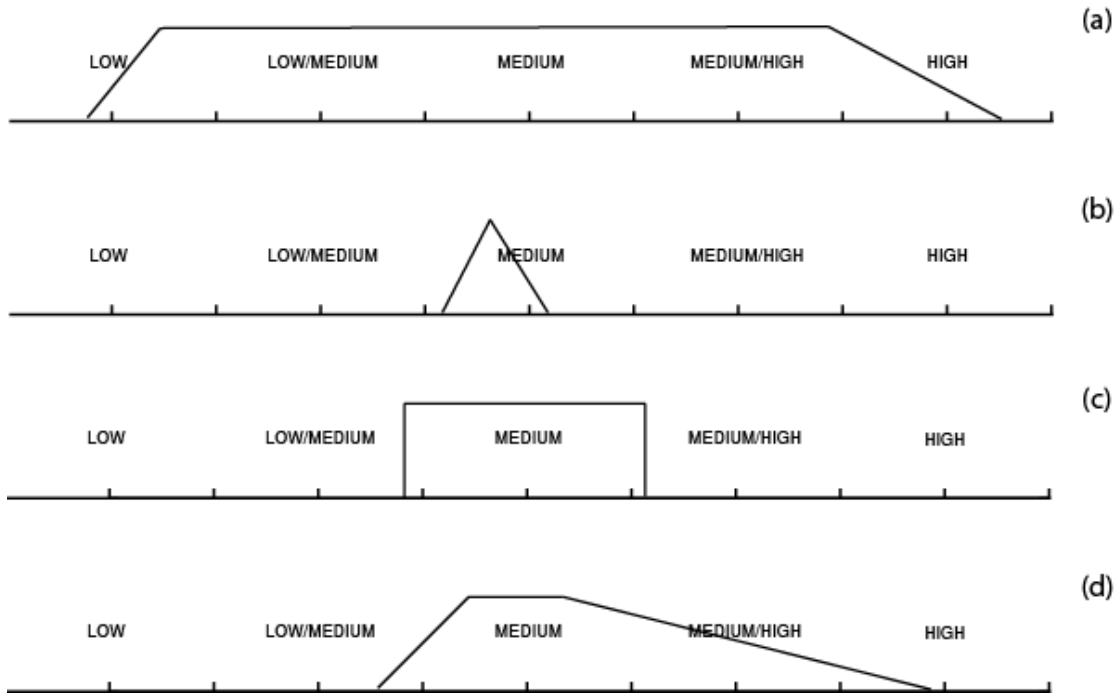


Figure 2.3 – Example fuzzy distributions similar to those used to demonstrate the meaning of different distributions during the elicitation

2.4.6 Qualitative questions

Qualitative questions were to some degree subject area specific, but also had some similarities in that they tried to elicit information in the following areas:

- Consideration of underlying assumptions.
- Discussion of data (quantitative data and qualitative information) availability and resolution for risk analyses.
- Discussion of reference conditions.
- Consideration of the link between ecological status and P status.
- Risk assessment scale.
- Consideration of P form/speciation.
- Consideration of approaches for higher tiered risk assessments.
- Consideration of likely changes by 2015 (e.g. climate or land use change).

There was no fixed structure within this section of the interview and in most cases these areas (and more) were covered whilst discussing points during collection of the quantitative information.

2.4.7 Combination of expert opinion

According to Clemen and Winkler (1999) combination or aggregation approaches may be categorised as mathematical and behavioural approaches. Behavioural approaches attempt to generate agreement among the experts by various interactions. The interaction may be face-to-face or unidentifiable exchange of information. These approaches consider the quality and dependence of the experts' probabilities implicitly.

Winkler *et al.* (1992) presented their reasons as to why assessments of several experts should be combined:

- A combined distribution produces a better appraisal than the individual distribution (in accordance with the psychological perspective that "several heads are better than one head", or with the statistics fact that "a sample mean is better than one observation").
- The combined distribution can be sometimes considered as some sort of consensus.
- It is more reasonable and feasible to use a single distribution as a representative of several distributions for some further analysis.

Other methods for aggregating probabilities from several experts are reported by Lindley *et al.* (1979), West (1988), Genest and Zidek (1986), and Clemen and Winkler (1999).

For this study the quantitative measures of expert opinion were combined by a simple summation of the fuzzy distributions (i.e. a logical OR operation): these distributions are presented in their respective sections. Qualitative information was not explicitly combined but was used to focus the discussion in the sections that report on each subject area. In these cases differences of opinions are valid and a useful way of critically reviewing the best approaches for WFD risk analyses given current knowledge.

2.5 Conclusion

A structured elicitation procedure was developed to elicit information on the appropriate conceptual models and likely uncertainties for the EA's WFD risk analyses. The elicitation structure was based upon previously published work on elicitation including psychological studies concerned with reducing the many associated potential biases.

The elicitation procedure worked well in all cases although some of the experts expressed the concern that quantitatively ranking individual components on their likely effects at the scales of interest was difficult owing to interactions within the system of interest, preventing simplistic cause-effect relationships being established. Hence, estimation of mutually exclusive scale-dependant importance and uncertainty ratings for individual conceptual model components is problematic for such complex, dynamic systems where nonlinearities and feedback effects can have a large affect on system response. Additionally, where experts were questioned about the likely risk to ecology from P, there is the added complication that the link between P and ecological status is not clear (Foy, 2007): i.e. it is somewhat artificial to consider P alone when other factors interact significantly. Thus, the elicitation questioning meant that experts were required to 'think across scales' and provide informed estimates of likely effects for which there is often little or no supporting scientific research at appropriate scales. Therefore the experts needed to account for the limitations of individual studies at smaller scales and extrapolate to the scales required for WFD risk analyses: the fuzzy weightings elicited are hence subjective. These observations highlight the difficult task of undertaking structured scale-dependent risk analyses in systems where experts' conceptual understanding is still not always clear.

Even under such difficult circumstances a number of the experts expressed the opinion that the quantitative phase of the interview was a useful exercise that made them think very carefully about the questions; without a structured elicitation questions may have been answered in a vague manner. The information gathered during the elicitation will help the EA structure and allow for uncertainty in their WFD risk analyses, as well as focussing future research. The way in which this may be best achieved is discussed in the following chapters.

3 Catchment Processes and Agronomic Practices

3.1 Summary

The results of the elicitation from the Catchment Processes and Agronomic Practice areas have been combined to produce a simplified conceptual model (Figure 3.1) of the primary factors determining the magnitude of P pressure at WFD risk analysis scales. There was generally a good degree of agreement between experts on which components were included and on the fuzzy weightings assigned. The chosen components are discussed with reference to the scientific studies to support their inclusion in the model. In most cases, however, the supporting scientific evidence is based upon small scale studies making extrapolation of understanding to WFD catchment scales problematic and a source of significant uncertainty. At these scales, the spatial and temporal heterogeneity of the system makes it difficult to apply our relatively high levels of process-understanding and makes low resolution monitoring data weak for evaluation of estimation methodologies and in resolution of the effect of individual processes. The absence of scale-dependent rules meant that experts were required to 'think across scales' during the elicitation making the results somewhat subjective. Thus, for certain elements of EA WFD risk analyses it may be beneficial to collect data at WFD catchment scales such that scale-dependent rules may be derived to inform future modelling approaches (see Section 11 for a fuller discussion). The experts highlighted a number of areas as research priorities:

- The significance of instream processes in the modification of the terrestrial P signal;
- More focus on P pressure during ecologically sensitive periods;
- Long-term catchment manipulation studies;
- Exploration of tracer techniques for source apportionment;
- More investigation of upland systems;
- More work on how predicted climate change patterns (e.g. rainfall patterns) affect P fluxes.

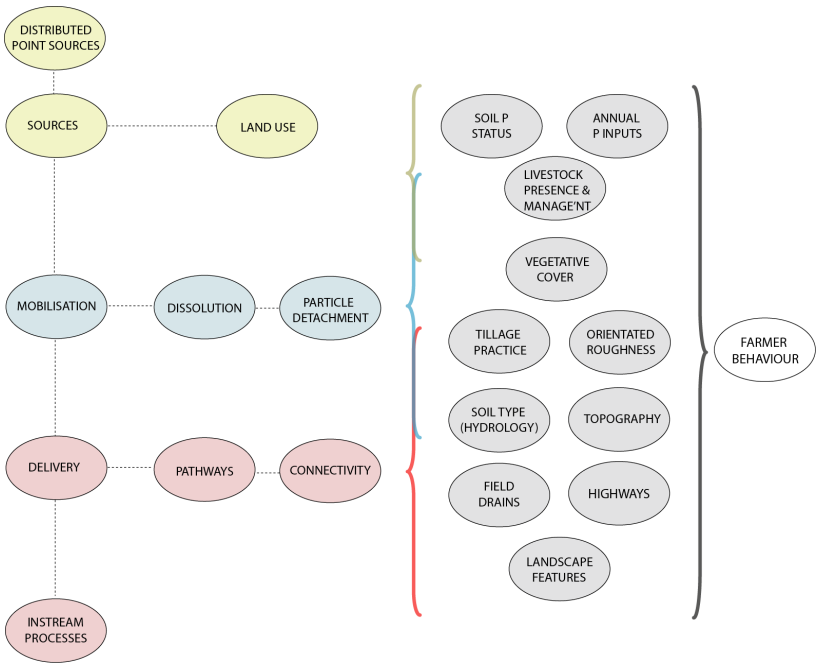


Figure 3.1 - Simplified Catchment Processes and Agronomic Practices conceptual model

3.2 Introduction

This chapter considers how intrinsic catchment characteristics and agronomic practices are thought to combine to determine the magnitude of P pressure for a given catchment. Separation of the individual effects of catchment characteristics and catchment management is difficult but was investigated, to some degree, by the consideration of the differences between two extreme catchment typologies (surface water dominated and groundwater dominated: defined below).

Although P fluxes are known to vary considerably in space and time, experts were asked to consider a pseudo 'annual' average P flux (i.e. a moving window in time of approximately 3-5 years) at WFD catchment scales (approx. 10-100 km^2); higher resolution behaviour is however discussed where relevant. Processes determining the magnitude of *diffuse* P fluxes were the main focus, although there was also consideration of semi-diffuse sources such as farmyards. Hardstandings and septic tanks are also considered (defined here as *distributed point sources*).

Conceptual models describing the factors that contribute to diffuse P fluxes from catchments are often operationally divided into *source*, *mobilisation* and *delivery*[Π] (Viney *et al*., 2000) and include those described by: Haygarth *et al*., 2005a; Haygarth and Jarvis, 1999; Heathwaite *et al*., 2003; Lemunyon and Gilbert, 1993. These divisions are a convenient way of grouping processes that determine P behaviour and were the framework for the development of the initial *catchment processes* conceptual model (Figure 3.2). Source, mobilisation and delivery are defined as follows:

> *Source* - The raw inputs of phosphorus to the system and system P 'stores' (see discussion below);
>
> *Mobilisation* - The start of P transfer, the process by which phosphorus molecules begin movement from the soil (Haygarth *et al*., 2005a): this can be via solubilisation and/or detachment (detachment of particles and associated P);
>
> *Delivery* - The linkage between the point (in time and space) of mobilisation and transport to a given waterbody (see also the discussion in Beven *et al*., 2005).

These definitions are, of course, oversimplifications as complex and interacting processes control the quantity of P mobilised and transported. Once mobilised, P is not necessarily transported to a water body in a single 'event' (e.g. particulate associated P lost via the surface runoff pathway may be deposited and resuspended numerous times before it is finally delivered) and there are fluxes between different P species during transport (Heathwaite, 1993).

Although sources of P include a natural background component and inputs from atmospheric deposition, the following discussion focuses on direct 'current' inputs from agriculture and the legacy of past agricultural activity (as quantified by soil P status). These two P 'stores' are conceptually distinct as some P applied or deposited on agricultural land does not become part of the soil store before it is lost to surface waters in an incidental manner (see definition of incidental below). The conceptual model components that were chosen to represent these two sources of P are described as *'current' P inputs and long-term P inputs* and are discussed further below. For a more detailed discussion of P sources see Haygarth and Jarvis (1999).

Many of the publications that describe the processes of P mobilisation and transport have, necessarily, focused on small scale (e.g. lysimeter and plot) experiments where it is more likely that the effect of individual processes can be resolved. In this study, experts were asked to consider which processes were likely to be most important (or dominant) in driving P loss at WFD catchment scales (approx. 10-100 km^2). Although it is not always made explicit in published scientific literature, there appears to be consensus amongst some authors that the principal factors controlling P loss can be scale-dependent (Barlow *et al*., 2005; Brazier *et al*.,

---

[Π] Sometimes used interchangeably with transport in published scientific literature.

2005; Gburek, 2005; Gburek and Sharpley, 1998; Harris and Heathwaite, 2005; Haygarth *et al*., 2005b; Krueger *et al*., 2007; Slaymaker, 2006; Wood *et al*., 2005). Studies which seek to determine scale-dependent rules or investigate the mechanisms that control the magnitude of P flux change with scale, (e.g. by nested studies: Kronvang *et al*., 2007) are relatively few and are considered in Section 11. Consequently, in the discussion below, there is an implicit reliance that experts were able to 'think across scales' to account for the limitations of scientific studies when expressing their opinions (i.e. their experience allowed them to make scale-dependent judgements that may not yet be fully supported by scientific evidence). This, of course, cannot guarantee an accurate representation, but can be taken as a first approximation in the absence of adequate scale dependent evidence.

In this section the components chosen by the experts for inclusion in a simplified conceptual model of diffuse P pressure at WFD scales are presented, together with rankings for their relative importance and associated uncertainty. The components are discussed with respect to the underpinning scientific theories and data availability to support their inclusion. Additionally, where relevant, there is a brief discussion of the consistency between the scale of scientific process understanding and the scale of WFD risk analyses (this is discussed further in Section 11).

3.3 Assumptions, definitions and Caveats

Unless specifically stated, experts' opinions relate to TP flux at the catchment outlet (rather than an 'edge-of-stream' flux): in-stream processes are considered as an aggregated model component. Solely, TP (rather than other P fractions, such as SRP) was considered owing to time constraints during the interview phase and the complexities associated with P fractionation (see the discussion on P fractionation in the next section).

For simplicity, sources are separated into two distinct 'stores':

- a soil P store where P has become incorporated into the soil;
- an *incidental* store from which P can be mobilised without ever becoming part of the soil.

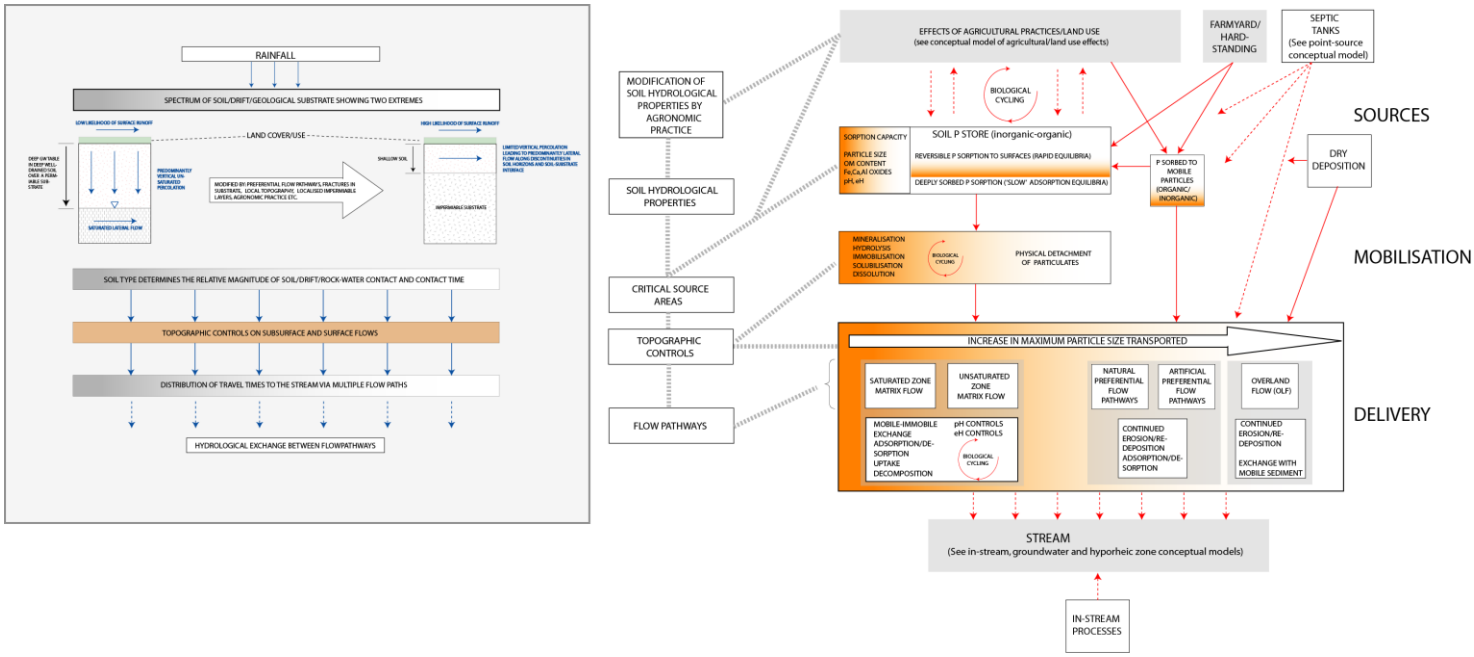


Figure 3.2: Interview phase conceptual model for catchment

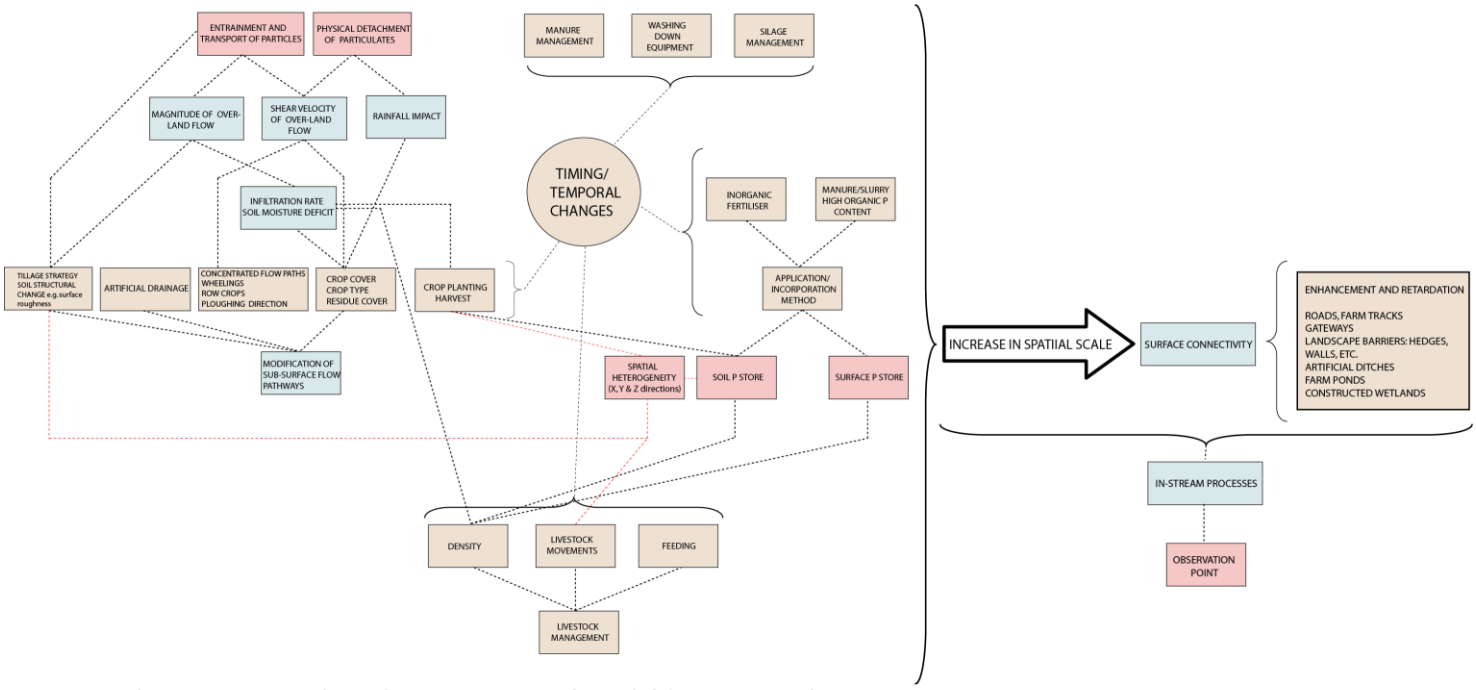


Figure 3.3: Interview phase conceptual model for agronomic.

The two generalised catchment typologies considered were ‘surface water dominated’ and ‘groundwater dominated’ and can be thought of conceptually as described by extremes of Base Flow Index (BFI):

- Surface water dominated catchments (SWDC) - flashy catchments where there is a predominance of lateral subsurface flow and surface flow.

- Groundwater dominated catchments (GWDC) - predominantly matrix flow (little fracture flow), an absence of Impermeable drift and a significant vertical distance from the surface to the water table.

Climate can have a significant effect upon the magnitude of P loss over many timescales from individual storm events to inter-annual variability (e.g. see Jarvie *et al*., 2003) and determining some spatial patterns for different areas (e.g. regional variations in hydrologically effective rainfall, HER[11]). For the purposes of this study, climate was assumed to be approximately constant and regional variations known. Thus, the experts accepted that one element of the spatial component of P loss can be approximated by HER.

3.4 Phosphorus Fractionation

The high level of complexity and uncertainty associated with the fractionation of P in the landscape makes it particularly difficult to discuss in a general sense. It is not possible to predict accurately how P fractionation changes along pathways (although it may be estimated from first principles) but it is difficult to predict quantitatively in space and time. Thus, there is almost no information on how much of each P species is contributed from different land use-catchment type combinations. Consequently, during the interview phase of the elicitation (described in Section 2) only TP was considered explicitly and quantitatively. Experts did, however, comment qualitatively on other fractions of P.

It is however clear that the simplistic operational definitions of TP and SRP can be misleading and may fail to identify some of the more important P fractions for certain landscapes. For example, Jarvie *et al*. (2003) showed that, for diffuse dominated catchments, organic and inorganic polymeric forms of P can be important. Analytical techniques also often omit the

[11] Total rainfall minus evapotranspiration losses for a given period.

detection of fine colloidal material and the organic fraction of sediments, which are believed to be important and which can contribute via surface and subsurface pathways (Quinton, 2004; Quinton et al., 2001; Stone and Walling, 1997). Similarly, the association between soluble P losses from grassland and particulate P losses from arable land is also now seen as too simplistic (e.g. the significant potential for PP transfer to occur from grassland: Hodgkinson and Withers, 2007).

3.5 Expert weightings for chosen model components

In Section 2 above the generation of the individual and composite fuzzy distributions are explained. In this section the significance of the distributions for chosen components is discussed.

3.5.1 Catchment Processes

Presented in Figure 3.4 are the Composite (a combination of the 2 or 3 fuzzy distributions elicited from the experts) fuzzy expert weightings for ten of the most important model components chosen for comment by the experts for SWDC. Figure 3.5 shows expert weightings for GWDC. A comparison between the two figures shows that many distributions have similar shapes and ranges but that there are some significant differences: annual P inputs, mobilisation and connectivity[¥] are thought to be less important for GWDC, although there was a lack of consensus regarding annual P inputs. For SWDC, the highest levels of consensus (and narrowest ranges overall) are apparent for the components connectivity, topography, soil type, annual P inputs and mobilisation. The number of components which show similar levels of consensus for GWDC are fewer: soil type, topography and connectivity. Of these components, and for both GWDC and SWDC, there was consensus that there were high levels of uncertainty associated with estimating connectivity: hence it is the component that has been implicitly identified as most critical, not only in terms of its importance for controlling P loss per se, but in the uncertainty associated with its estimation (this then implicitly highlights connectivity as a research priority).

3.5.2 Agronomic Practice

Presented in Figure 3.6 are the fuzzy expert weightings for nine of the most important model components chosen for comment by the experts for SWDC. Figure 3.7 shows expert weightings for GWDC. In general both figures show broad ranges for both importance and uncertainty of components much of which is a result of broad distributions elicited from individual experts rather than a lack of consensus; indeed this shows agreement that there is no clear evidence of the relative importance of these components at the scales of interest here. Exceptions to this pattern, where there is a clear lack of consensus, relate to the role of vegetative cover, tillage practice and buffer features for GWDC; judging from the qualitative information elicited, this lack of consensus results from the fact that these are factors which primarily control surface runoff effects (and likely connectivity), which may hence be less significant for GWDC: this is supported by the fact that the overall importance for these components has been ranked lower for GWDC compared to SWDC.

Instream processing of P was given a relatively high ranking by all experts from both catchment processes and agronomic practice subject areas, although some experts expressed the opinion that, depending upon factors such as bed sediment type and morphology, flow regime and plant community abundance, its importance would vary (e.g. the long tail of the fuzzy distribution

[¥] Here, connectivity refers to the degree hydrological connectivity from a given terrestrial location to a waterbody (can be general or via a specific flow pathway) which is dynamic in space and time, but may also be expressed by some form of connectivity index such as drainage density. This phenomenon is disticly different to that used frequently by ecologists to mean continuity of streams and rivers to their ultimate outlet (as used in Sections 10 and 11).

towards low importance is associated to some degree with its perceived lower importance for upland streams).

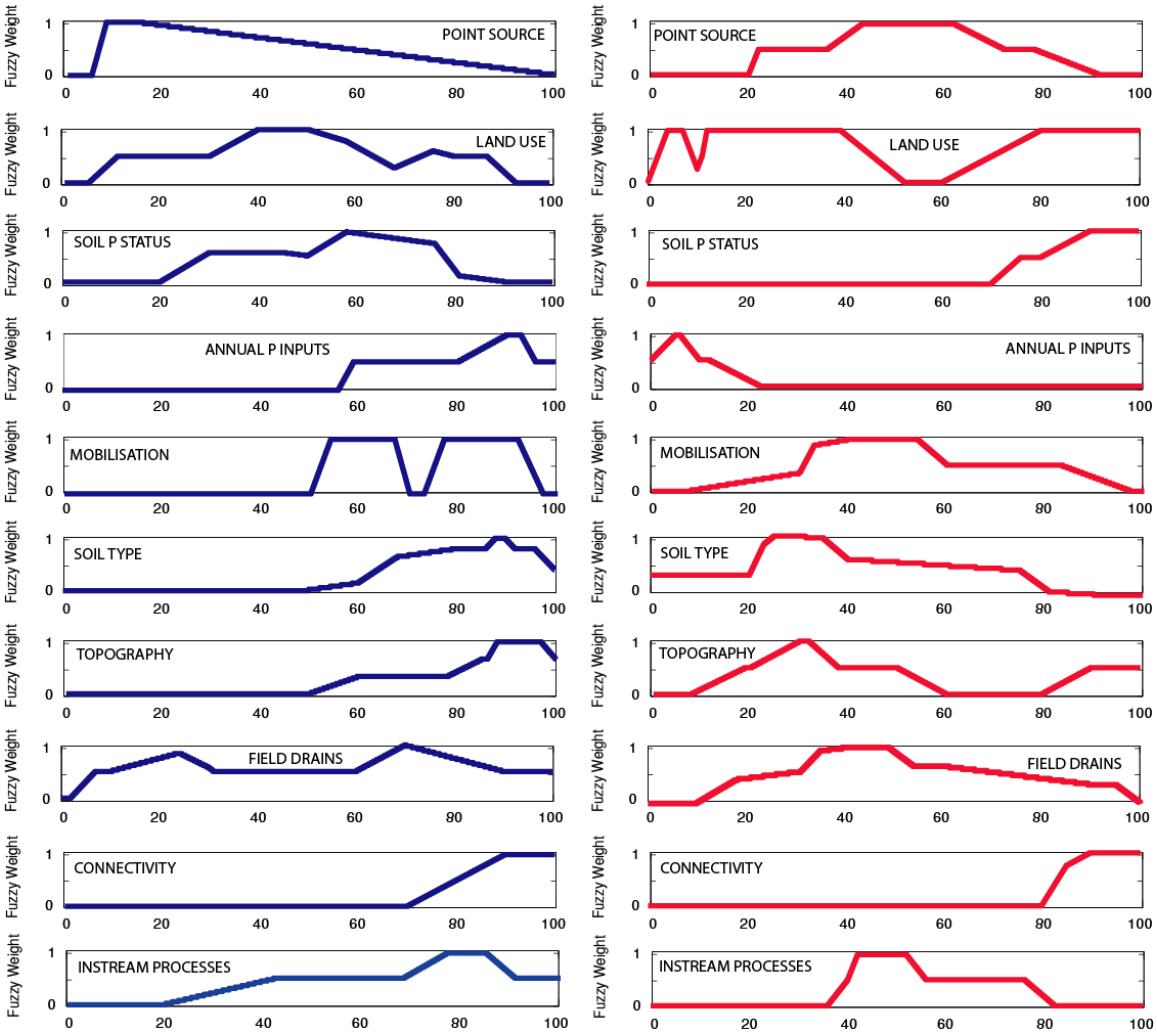


Figure 3.4. Catchment Processes – SWDC Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

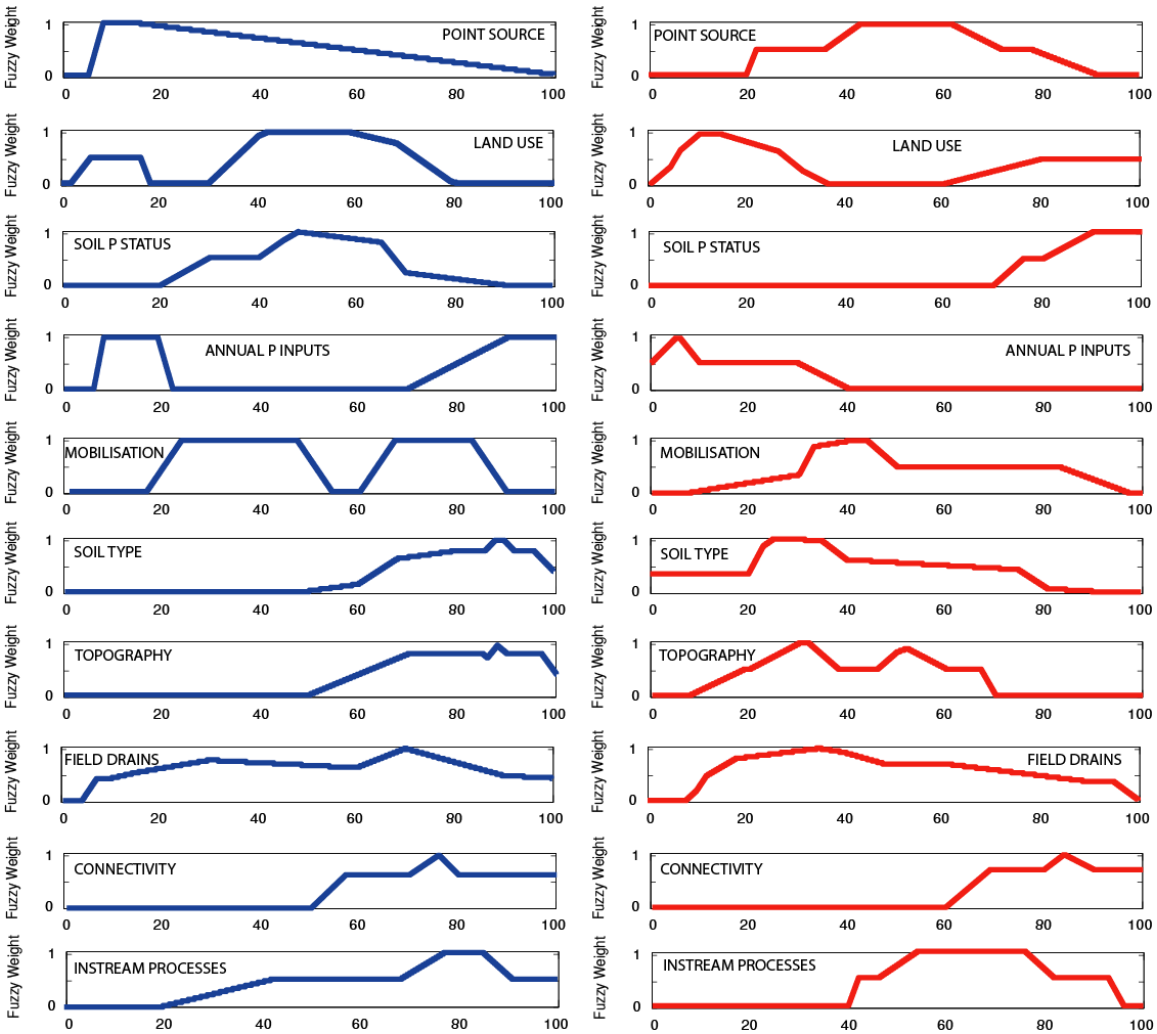


Figure 3.5. Catchment Processes – GWDC Importance and Uncertainty Ratings - In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

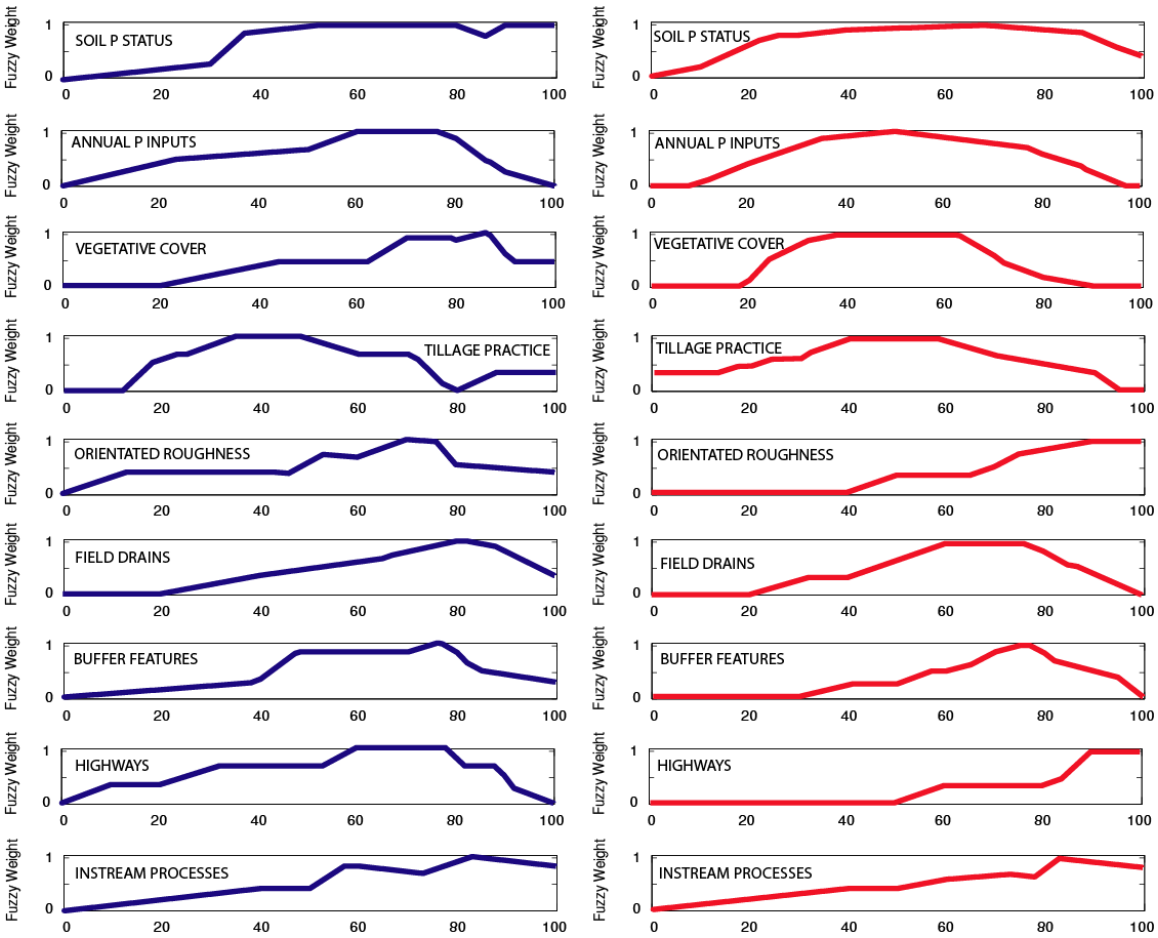


Figure 3.6. Agronomic Practice - SWDC: Importance and Uncertainty Ratings - In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

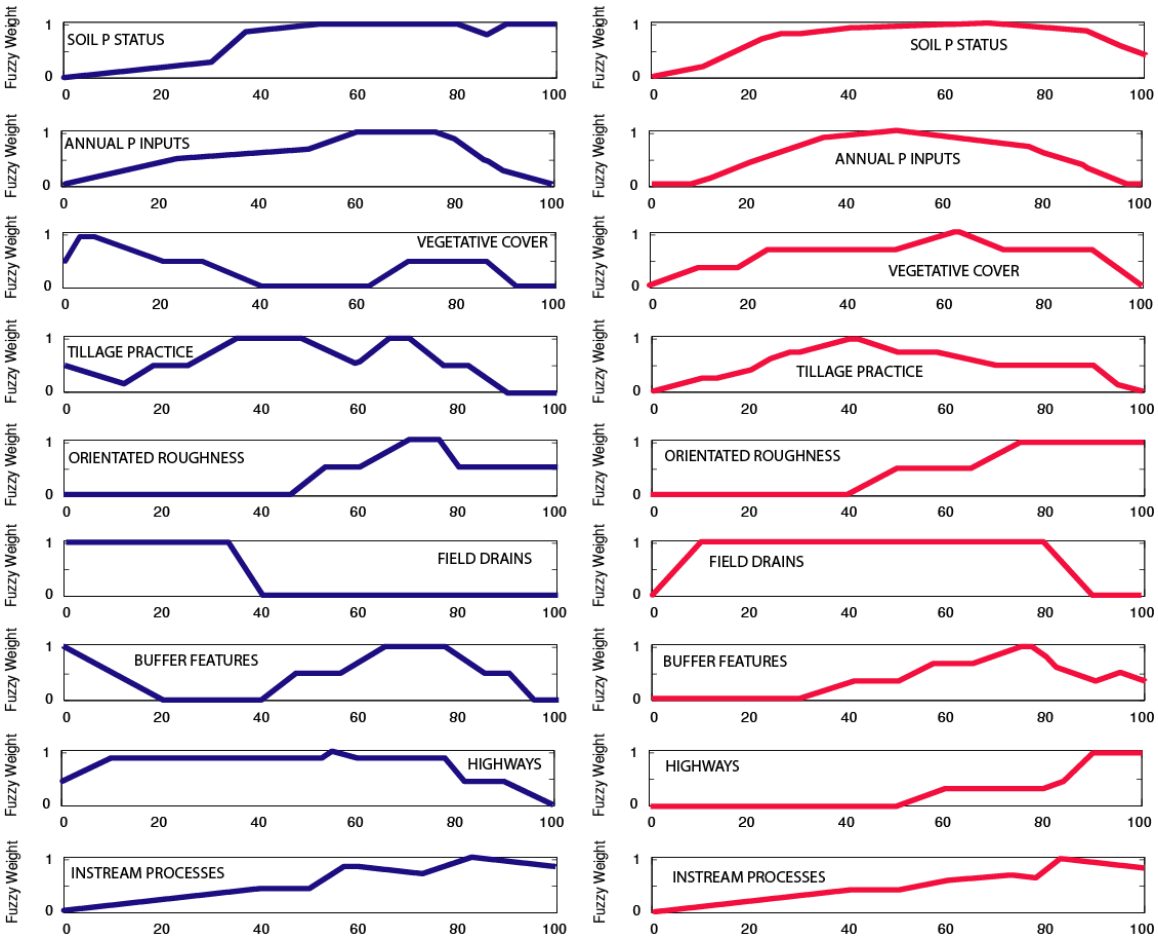


Figure 3.7. Agronomic Practice - GWDC: Importance and Uncertainty Ratings - In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts. Note that only one expert chose to comment on field drains in GWDC; the other experts thought that they would not generally be present.

3.6 Discussion of the chosen conceptual model components

3.6.1 Land Use

Although land use is a general term that covers many of the components discussed below, it was highlighted by the experts for inclusion into a simplified/focussed conceptual model. Land use has played a central role in simplistic models such as those utilising export coefficients (e.g. Johnes et al., 1996) and may be in some cases an adequate division for a risk analysis or adequate to derive estimates of other model components where detailed data is unavailable: e.g. soil P status and/or 'current' P inputs (see the discussion below). The 'long-term' land use and associated P inputs are the primary control of soil P status and the current land use determines the 'current' P inputs which control the status of more short-term P stores such as those which may contribute to incidental P losses, which are thought to be quantitatively important in some areas (Defra, 2002; Haygarth and Jarvis, 1999; Heathwaite *et al*., 2006; Hooda *et al*., 1997).

In the UK, there is a reasonably good understanding of the likely land use for a given location, or types of land use within a given area, although there will be uncertainties associated with change over time. For example the CEH land cover map dates from 2000 (an updated version will be published in 2009). It is worth noting that land use describes a general risk which implicitly includes the risks from land management practices which can contribute significantly to variability in P loss risk from land use per se (Hodgkinson and Withers, 2007).

3.6.2 'Current' P inputs

Current P inputs are defined here as those which are likely to be lost from a conceptual incidental store (as discussed above P that is lost before becoming part of the soil). They comprise inputs of farmyard manure, fertiliser etc. in a 'present day' 3-5 year 'moving window'. This implicitly includes the timing of inputs and does not include inputs from distributed point sources (see below).

All six experts from both the catchment processes and agronomic practice subject areas identified annual P inputs as important. Some of the experts chose to disaggregate this component further to include the management of P inputs and livestock. As there was not consensus that it was necessary to include this much detail, a qualitative, rather than quantitative description follows for these factors.

3.6.3 Management of P inputs

The management of P inputs to agricultural land (not including grazing inputs which is covered in the next section) is defined here as the type, timing and method of P application as fertilisers and manures, rather than the magnitude of the inputs per se. These inputs are thought to be primarily associated with the risk of incidental P losses. Incidental losses have been shown to contribute to high P concentrations in surface runoff and drain flow and to be a significant proportion of the P fluxes for individual rainfall events (Heathwaite *et al*., 2006; Preedy *et al*., 2001; Withers *et al*., 2003; Catt *et al*., 1998). Preedy *et al*., 2001, however, warned of the difficulties of upscaling the results from small scale experiments of incidental loss.

Representation of 'current' P inputs in space and time for risk analyses is challenging owing to a lack of information at smaller scales (information collected under the Agricultural Census is aggregated to the Defra grid cell scale). There will be a huge difference in the uncertainties associated with the P content and bioavailability of bag fertiliser compared to animal manures, which are more variable.

3.6.4 Livestock management

Livestock management, as defined here, includes the timing of the grazing season, and how this interacts with soil conditions that can aggravate the effects of trampling by grazing animals (poaching and pugging), direct deposition of manure and livestock access to streams.

Poaching and pugging refer to the compaction and breakup of soil due to trampling that can lead to a reduction soil infiltration rate (Heathwaite *et al*., 1990) and can enhance the initiation of overland flow (McDowell *et al*., 2007; McDowell and Sharpley, 2003). Poaching and pugging are of critical importance in localized areas and hence may be more important at the scales of interest here for SWDC compared to GWDC, where their effects are likely to be more hydrologically isolated. Expert 3c expressed the opinion that there is no question that poaching and pugging have an effect but trying to quantify their effect at catchment scales is difficult: significant uncertainties result from poor knowledge of when and where they occur. However, (Tunney *et al*., 2007) found, using plot scale studies, that although grazing animals can have a significant influence on TP concentrations in overland flow (not on soluble P concentrations) the impact is minor compared to other factors that determine P loss from grassland (such as the effect of soil P status).

Where livestock have access to streams direct deposition of manure can take place. There are few quantitative studies on the likely P flux associated with direct defecation to waters but notably Gburek (2005) showed that lactating dairy cows can defecate 0.34 kg P ha^{-1} yr^{-1} directly into a stream, and as much as 1.25 kg P ha^{-1} yr^{-1} on near-stream areas (within 40 m of stream) prone to surface runoff, while grazing. In situations such as those described by Gburek (2005), exclusion of cows from streams and riparian areas can significantly reduce P loss from the catchments.

For EA risk analyses purposes the way in which livestock numbers are represented will depend upon data availability. At the national scale, it is possible to obtain an approximate number of livestock for a given area and to determine approximately when they will be grazing and when they will be housed, but it is not possible to identify livestock numbers per field (Expert 1c; Expert 2m): Expert 2m thought that we could be in error by as much as 30 to 40% using Agricultural Census data.

3.6.5 Long-term P inputs (soil P status)

There is a well understood relationship between dissolved P fluxes and soil P status particularly at the smaller scale (Heckrath *et al*., 1995). There is however significant variability in the predictive capability of different soil P tests (Nash *et al*., 2007) and their predictive capability may decrease with increasing scale (Quinton *et al*., 2003). There is less of a direct link between soil P status and particulate P losses as there are so many other factors involved, such as soil erosivity and spatial connectivity; thus there is even more uncertainty in this relationship with increasing scale (Harmel *et al*., 2006; Quinton *et al*., 2003). It is not possible to make a good prediction of how fields with different soil P status interact at the catchment scale: larger scale empirical rules are required to make these types of estimate (van der Perk *et al*., 2007). These fundamental scaling issues are compounded by significant spatial heterogeneity in soil P status at sub-field scales (Needelman *et al*., 2001; Page *et al*., 2005). Data on soil P status is sparse; the UK National Soil Inventory gives total and Olsen P values at 5 km nodes across the UK. At each node, 25 cores (15 cm deep) were taken using a 4m grid within a 20 x 20m square. These data include the soil type and land use at the node but in many cases may not be a reasonable measure of the soil P status of a given catchment (Page *et al*., 2005). With the introduction of entry level and higher level stewardship schemes soil analysis is becoming a much more integral part of the system and this information may be useful; it is possible however that this data may not become available for risk analysis purposes owing to issues related to anonymity.

3.6.6 Phosphorus Mobilisation

In this study experts chose to consider TP only, for the reasons discussed above. However, mobilisation describes the initial detachment of the P sources discussed above and principally (although simplistically) includes two different processes, particulate P detachment and P dissolution, which would be seen by some to require different modelling approaches. These distinct processes also tend to mobilise different forms of P: dissolution mobilising SRP and particle detachment mobilising PP. Dissolution primarily reflects the 'long-term' P status of the soil (as described above with regard to the effect of soil P status) and has been shown to increase more rapidly above certain soil P thresholds in some soils (Heckrath *et al*., 1995) where soils are more 'saturated': the potential for P losses from saturated soils (in the present and the future) is thought to be of primary concern and a high risk: Expert 2m). Particulate P mobilisation is primarily linked to soil erosion (Catt *et al*., 1998) and hence is partly controlled by the intrinsic erosion risk of a given soil (Evans, 1990) and the agronomic practices that may alter this risk. Much of the work on P mobilisation has been carried out on controlled experimental plots with little examination of the effects of landscape position (Haygarth *et al*., 2005a). At catchment scales, the amount of P mobilised is complex, is not easily predicted and interacts with delivery in space and time leading to significant challenges in its use as a model component (e.g. see the discussion of Beven *et al*., 2005). For different catchments, there is a changing balance between the relative proportions of mobilisation by dissolution and mobilisation by detachment (depending upon catchment characteristics) which may need to be captured but is often lost by the focus on particulate P mobilisation (detachment) driven by a measure of land surface slope in many models (Expert 3c).

3.6.7 Vegetative cover

The importance of vegetative cover (areal vegetative cover), as considered here, is related to its role in reduction of surface runoff velocity and hence the potential for entrainment of sediment and associated P. Thus vegetative cover is likely to be more important for SWDC and may be significantly less important in GWDC. The experts thought that there was however little quantitative evidence for its effect in GWDC where additional uncertainties arise owing to localised surface runoff and connectivity via preferential flow pathways. Where surface runoff is important, there is a significant body of evidence to show the positive effect of retaining vegetative cover on soil erosion and particulate P loss. For example, Quinton and Catt (2004) showed that erosion (on sandy soils) can be minimized at source by early planting of autumn sown crops and not planting poorly covering crops such as sugar beet on steep slopes. Laubel (2004) showed that, over a 6-year study period, soil erosion was highest on fields with winter cereals and lowest on fields with grass.

Information on vegetative cover is poor at the national scale as it would mainly be based upon land use (see discussion above). Land cover derived from the CEH maps is not particularly useful in some locations owing to seasonal differences in cover. The NDVI (Normalised Difference Vegetation Index) remote sensing database may provide more useful data for smaller geographic areas (not at national scales) but it would be time-consuming to interpret the database information.

3.6.8 Tillage Practice

There was consensus amongst the experts that the extreme practices of *till*[γ] or *no till*[ℜ] were considered to be towards the worst and best case scenarios respectively for TP loss, and with ploughing downslope being worse still (Expert 1; Catt *et al*., 1998). Withers *et al*. (2007) showed that concentrations of sediment and P in runoff from Greensand and Chalk soils were consistently lower when the soil was minimally tilled compared to ploughed; the benefits of reduced cultivation were attributed to better cover and a firmer surface for tractor wheelings. Early drilling, direct drilling (drilling seed directly into the soil without cultivation), timeliness of cultivation to avoid soil compaction, better tramline management and reduced cultivation techniques were suggested as measures to reduce losses (Withers *et al*., 2007). Quinton *et al*., (2001) showed, using experimental plots, that small erosion events could be controlled most effectively by means of minimal tillage techniques and across-slope cultivations; across-slope crop cultivation forms a second line of defence (to vegetative cover) but there is a risk that accumulated water can ‘break through’ and may cause large rills or gullies (Quinton and Catt, 2004).

Information on tillage practice is poor as the data collected by Defra does not include detailed tillage information or planting dates. Individual farmer practices, could however be identified for higher tiered approaches.

3.6.9 Orientated roughness

Orientated roughness, as defined here, encompasses tramlines, plough direction and row crop direction and is important for focussing or preventing downslope overland flow, depending upon orientation. Where continuously orientated downslope, it is associated with facilitating increased runoff and erosion (Martin, 1999; Stein *et al*., 1986). There are a number of studies from the sediment loss literature that show these effects (e.g. Quinton and Catt, 2004) but few from a P loss perspective. The overall effect of orientated roughness at catchment scales will depend upon connectivity with surface waters. Expert 3ag expressed the opinion that “we believe we know more about the effect of orientated roughness (…at larger scales…) than we really do” and that it is a good example of how “Anecdote can prevail over evidence”.

[γ] i.e. conventional tillage practice

[ℜ] In the UK, no-till does not mean that the soil is never inverted: it is recommended that inversion occurs approximately every 3 years.

Information for estimating orientated roughness is poor at the national scale. There are data identifying which crops are being grown (i.e. the annual IACS data: Integrated Administration and Control System) but it would not indicate the orientation. Perhaps the best estimate of orientation would be to assume that farmers generally till along the longest dimension of the field to minimise tractor turns (Expert ag1). Estimates of orientation may be practical for higher tiered approaches where GIS map data or aerial photos may be used.

3.6.10 Soil Type

Soil type is used here with regard to its control on soil hydrology: i.e. partitioning of flow between different pathways. The chemical properties of soil are implicit within the mobilisation component discussed above. Discussion on the details and dynamics of flow partitioning is beyond the scope of this report and is covered in more detail by (Gburek, 2005; Gburek and Sharpley, 1998; Haygarth *et al*., 2000; Haygarth and Jarvis, 1999). It is worth noting here that soil type can interact significantly with the underlying geology and drift deposits.

There was consensus amongst the experts that in general we can only approximate flow partitioning (i.e. in a relative way such that we know that certain soils are more prone to overland flow) but that uncertainties arise owing to control by hydrological connectivity. We have very good scientific understanding of soil hydrology and how it is altered by interactions with natural and anthropogenic factors, but it is difficult to apply this knowledge owing to spatial and temporal dynamics and heterogeneity. This can lead to large uncertainties in predictions made for individual locations and for given points in time: it was thought that perhaps more robust generalised predictions may be possible at larger scales where simplistic measures such as Base Flow Index (BFI) may be used to approximate flow partitioning.

3.6.11 Topography

Topography is used here in a broad sense to represent the slope, and arrangement of slopes within a catchment that ultimately have an effect on surface and near-surface flows, connectivity of flow pathways and soil erosion potential. Although slope (gradient and slope length in some cases) is frequently reported to be a driver of soil erosion and P loss (e.g. the Universal Soil Loss Equation: http://www.omafra.gov.on.ca/english/engineer/facts/00-001.htm and P index approaches: Lemunyon and Gilbert, 1993; Sharpley *et al*., 2003), low slope environments are also important for increased dissolution of P (Expert 3c: see also the discussion of mobilisation).

Our data for determining topography accurately is relatively good, including regional and national scales where high resolution digital elevation models are available. Topography has been used simplistically and successfully in hydrological models for spatial estimation of surface and near-surface flows in shallow soils (Beven and Kirkby, 1979) and for estimating the relative connectivity of surface flows over large scales (Lane *et al*., 2006). This type of catchment scale connectivity analysis may be advantageous as there are significant scaling issues in extrapolating from small scale observations of overland flow and local connectivity.

3.6.12 Artificial drainage

There is some conflicting evidence on the role of drains in determining the magnitude of P loss although there is more evidence for their role in increasing P loss, although this has not been ascertained at a national scale (Chapman *et al*., 2001). The conflict lies in the interaction between drains as a mechanism for reducing overland flow (and hence P lost in overland flow) and the increased connectivity between drained fields and surface waters (e.g. compare the findings of Dils and Heathwaite, 1999 and Heathwaite *et al*., 2006 with Haygarth *et al*., 1998). The conflict here is, however, more to do with the scale of observation and the role of connectivity as scale increases. Thus, it is generally considered to be the case that artificial drainage tends to increase P loss by increasing connectivity.

There have been a number of studies where a significant proportion of the annual P flux has been delivered via field drains (e.g. Catt *et al*., 1998; Gardner *et al*., 2002; Gelbrecht *et al*., 2005; Uusitalo *et al*., 2001); however, these studies have not quantified the overall effect of the presence or absence of drains on P flux. An additional complication lies in the role of in-drain sediments, which have been shown to have the potential to alter P concentrations and sediment fluxes in much the same way as thought to occur instream (Barlow *et al*., 2003).

In the UK there are few accurate data on the location of field drains (although there are some parish records) and there are ways of inferring the presence or absence of drains using soil type. There is additional uncertainty stemming from the unknown condition of drains and whether or not their function is impaired.

3.6.13 Agricultural landscape structural features

For the purposes of this study agricultural landscape structural features are defined as constructed features that have an effect (positive or negative) on connectivity within catchments. The features considered include: gateways, tracks, hedges, walls, constructed buffer zones (Kronvang *et al*., 2005; Uusi-Kamppa *et al*., 2000), ponds, wetlands (Uusi-Kamppa *et al*., 2000). Public highways and artificial drains are considered separately. Many of these phenomena are covered in a more detail by Stevens and Quinton (*in press)* and Stevens and Quinton (*accepted*).

The experts were all of the opinion that these features are widely accepted to be important, but that they not well understood, particularly in a quantitative way. Much of our knowledge is anecdotal and has arisen from qualitative field observations or small scale studies (Expert 2m). Most of the features considered are likely to have more of an effect for SWDC owing to the greater likelihood of surface connectivity. There have been some larger scale studies showing the cumulative effect of riparian buffer zones at field scales. For example, the paired catchment studies reported by Clausen *et al*. (2000) where concentrations of total P and total suspended solids in surface runoff were reduced by 73 percent and 92 percent respectively, and that of Jokela *et al*. (1999) where significant reductions in P were also reported). See also the review by Sharpley *et al*. (2006). However a catchment scale study reported by Hodgkinson and Withers (2007) showed that the establishment of a 6m riparian grass buffer strip at the Cliftonthorpe catchment (an approx. 1 km^2 catchment in Leicestershire) had no beneficial effect on TP loss over a 3-year period and tended to increase the proportion of P transferred in soluble form.

As these features are most important in the modification of surface connectivity, there are significant uncertainties associated with predicting their interaction with spatially heterogeneous surface and near-surface flow. These uncertainties are in addition to those associated with identification the features per se at larger scales.

3.6.14 Public highways

Public highways were identified as potentially important for some catchments with respect to surface flow connectivity to watercourses. There was however a large spread of opinion between experts and in the importance fuzzy distributions elicited (Figures 3.6 and 3.7): highways were ranked as the most uncertain component in the agronomic practices area. The experts explained that the breadth of the distributions reflected the fact that there are no known experiments that show the magnitude of their effect, which may only be important (with respect to increasing P concentrations) locally and periodically, rather than for long-term P flux. The effect of public highways needs to be quantified, but it is perhaps not a priority (Expert 3a).

Hydrological Connectivity and Critical Source Areas
The hydrological connectivity of given areas (i.e. some sub-division of a catchment) for diffuse pollutants has been a priority for over 20 years. For example, the discussion of Critical Source

Area (CSA[ξ]) identification, including the influence of hydrological connectivity, by Maas *et al*. (1985)) shows that we are still struggling with the same questions today as we were more than 20 years ago: the identification of CSAs is still a research priority (Withers and Lord, 2002).

The a priori determination of CSAs inherently requires an estimate of hydrological connectivity, including macropores and artificial drainage (Stamm *et al*., 1998 and McGechan, 2002 respectively), which is a challenging problem even within research catchments. Alternatives in tackling this problem include: CSA identification using GIS overlays of soil, slope, land use, and watercourses, etc. (e.g., Sivertun *et al*., 1988); the probabilistic design curve method of Gburek *et al*. (2002), which identifies a *contributing distance* for riparian areas; a topographic index–weighted method of (Endreny and Wood, 2003); and high resolution topographic information and digital terrain analysis to identify areas most likely to be connected to a stream (Lane *et al*., 2006).

A recent review of methods used within Europe for localisation of CSAs was undertaken during COST action 869 (Mitigation options for nutrient reductions in surface and groundwaters) can be found at http://www.cost869.alterra.nl/hamar/ Report_WG1_Hamar.pdf.

3.6.15 Distributed point sources

Distributed point sources within catchments include septic tanks, farmyards and hardstandings. All six of the experts in the catchment processes and agronomic practices area thought that distributed point sources had the potential to contribute significantly to catchment P losses but that quantitative evidence was lacking. There was a feeling that a few critically located point sources could be crucial, locally, in increasing P concentrations (particularly during summer low flow conditions) but there is very little information on the overall magnitude of their effect at larger scales. Thus P losses from farmyards and hardstandings are akin to incidental losses and losses from septic tanks may contribute to both incidental and chronic losses (see Section 4). Overriding control of the likelihood of these to significantly contribute to P loss is their position in the landscape (in relation to other catchment characteristics) and the associated connectivity to local water courses. Accurate spatial identification of distributed point sources is difficult at large scales but may be possible for more detailed studies where Lidar data or a catchment visual assessment may help.

3.6.16 Farmer behaviour

Only one expert explicitly chose farmer behaviour as a component that should be included in a conceptual model of agronomic practice, but it is worth further discussion as qualitative statements from other experts indicated strong opinions on uncertainties derived from compliance, or lack of compliance, with codes of good agricultural practice. There was consensus that we cannot assume farmers follow good agricultural practice and that there will be extreme behaviour in some cases. In particular, farmer behaviour is key in controlling incidental losses associated with the timing of P applications, livestock management and other events such as washing of farm vehicles. The effect of behaviour is largely unpredictable and hence contributes to uncertainty: perhaps the best information available on farmer behaviour is uptake of stewardship schemes; for example see the work by Chadwick *et al*. (2008) investigating some of the 'social' drivers of farmer behaviour.

3.6.17 Instream processing

The instream processing of P (i.e. the ability of instream processes to act as a net source or sink for P over various timescales) is covered in more detail in Sections 6 and 7. In this section the importance of instream processing relates to its role as a modifier of P concentrations/fluxes

[ξ] Specific areas most vulnerable to P loss owing to interactions between sources and transport mechanisms (Gburek and Sharpley,1998).

observed at catchment outlets. This is particularly important as: analysis of catchment outlet signals is one way that we investigate/hypothesise about the functioning of terrestrial systems and many models (which predict P loss at the edge of field or delivered to the stream) are evaluated using catchment outlet data. Thus the net effect of instream modification of P fluxes may need to be taken into account. Instream processing was seen to be relatively important by all experts within both the catchment processes and agronomic practice areas (see Figures. 3.4-3.7). However, as there are a multitude of complex processes acting at different timescales, there was no consensus on what the net effect of all in-stream processes are likely to be (e.g. a net increase or decrease in P flux over a given period). This is discussed in more detail in Sections 6 and 7.

3.7 Simplified conceptual model of P flux risk

The conceptual model components described above are, with some exceptions, relatively well supported by experimental evidence with regard to their importance in controlling catchment P fluxes. There is, however, far less evidence to show their relative importance for given catchment types. The principal area where scientific evidence appears to be lacking is in the determination of the magnitude of effect of these phenomena (and the role of interactions between them) at larger spatial scales. Other areas where evidence is lacking relate to more qualitative observations, which provide anecdotal evidence, yet are difficult to resolve experimentally given the level of resources normally available (e.g. the magnitude of effect of distributed point sources).

The lack of knowledge at appropriate scales is particularly challenging to overcome by planned experimental observation. An experiment with which to resolve the effect of multiple complex, nonlinear and interacting processes would need to include many observations at multiple spatial scales (e.g. nested observations) for catchments with differing characteristics, would need to operate over a long time periods to reduce the effects of climatic variability and may need to include large scale agronomic manipulations. These findings are not new, and have been highlighted by a number of publications (Beven *et al*., 2005; Dougherty *et al*., 2006; Haygarth *et al*., 2005a; Kronvang *et al*., 2007), and lead to our reliance on model predictions for evidence in some situations. At the scales of interest for national scale WFD risk analyses, some form of modelling is required to interpolate between and extrapolate from GQA observations, which are the main source of evaluation data. Details on experts' opinions regarding suitable modelling approaches and methodologies for incorporating uncertainty estimates are considered in detail in Section 11.

The simplified conceptual model in Figure 3.8 presents the components chosen by the experts in schematic form linked to a source, mobilisation and delivery framework, where there is a disaggregation of general concepts to more specific phenomena from left to right. The schematic representation in Figure 3.8 does not identify causal links or effects of interaction which may be required for predictive modelling. The degree to which any of these components should be included within a P pressure risk analysis (and the 'weighting' they are given within the risk analysis structure) depends partly upon whether or not it is required to attribute risk to different factors and partly on the functional significance of these characteristics for a given catchment or catchment type. Given the brief discussion above on the uncertainties associated with the magnitude of effect of components, in isolation and combination, at large spatial scales (and the temporal scale of the risk analysis) and of the available data, the choice of which components to include and how to combine them is not clear and needs to be assessed quantitatively (this is discussed further in Section 11).

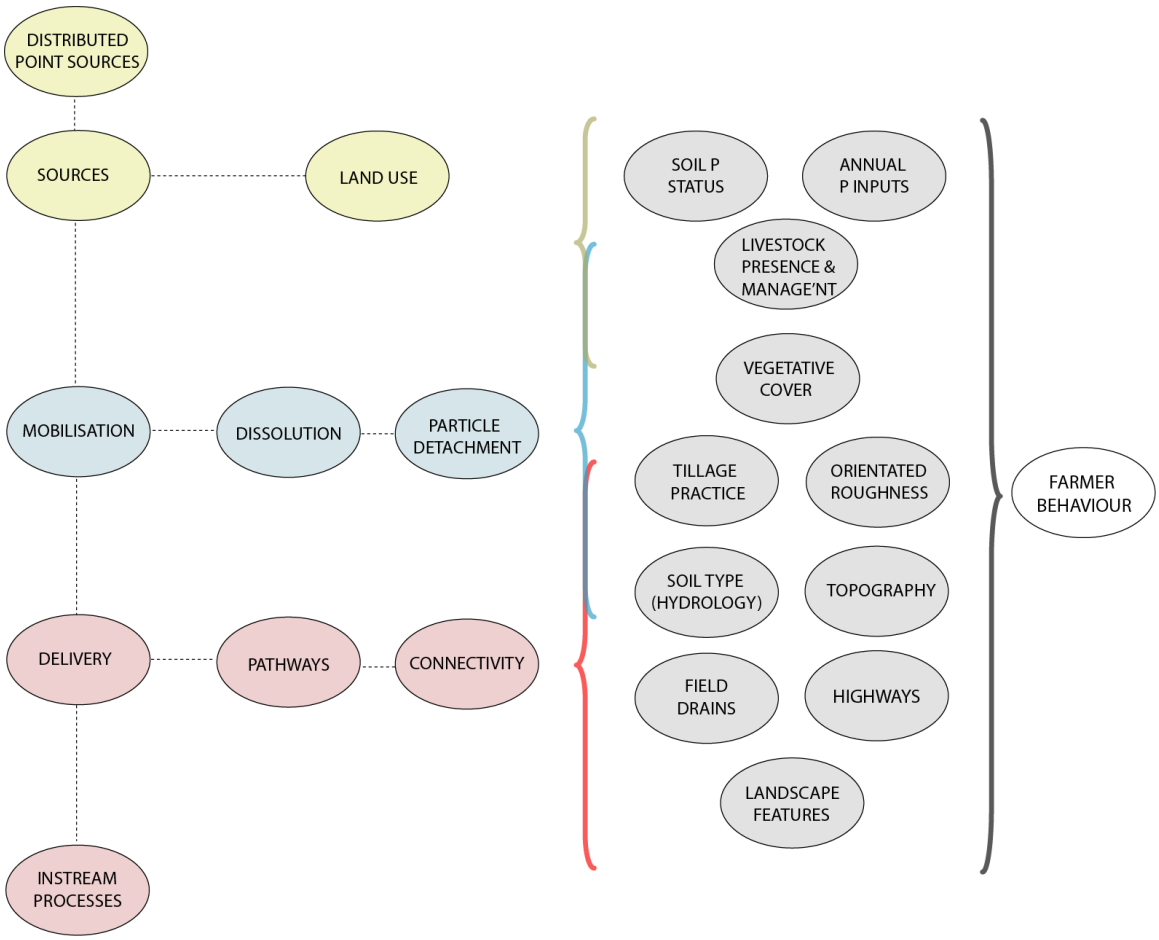


Figure 3.8 – Simplified conceptual model including all components chosen by the experts within the catchment processes and agronomic practices sub areas.

3.8 Additional Considerations

Likely effectiveness of environmental stewardship schemes by 2015
The experts believed that there may be short term gains by 2015 by measures that reduce P loss by management of hardstandings, reducing livestock access to streams and the reduction of inorganic fertilisers as manure but that there will be a lag in effect from some measures owing to the legacy of long-term land management (e.g. owing to soil P hysteresis effects and instream processing). In general improvements from ECSFDI and low level stewardship schemes may be small and there are potentially larger gains to be made with higher level schemes, but these need to be put into context with ecological sensitivity, not just a reduction of annual P loss. The measure being carried out within ECSFDI are a step in the right direction (although there may be some 'pollution swapping[ℵ]' issues) and farmers are becoming more environmentally aware. Thus, even if we cannot measure any effect from the ECSFDI monitoring (owing to difficulties resolving changes due to natural variability, using relatively low resolution monitoring), there will be a more 'social-type' effect in as much as: if you engage with farmers the benefits will eventually come.

However, some of the benefits of better agricultural practice may be negated by economical drivers such as those driving increases in maize production and climate change effects (such as increased summer storms) may change the annual distribution of P loss to coincide more with ecologically sensitive periods. Additionally, although not supported by much quantitative evidence, there was a feeling that distributed point sources may have a significant impact at a

[ℵ] Pollution swapping refers to the negative effect of a given mitigation measure upon another pollutant or another form of the pollutant in question.

local level which may obscure reductions to the diffuse signal. There is a significant issue also with 'true' point sources, as trying to resolve reductions in diffuse P loss from data which cannot separate the diffuse from the point signal (where uncertainties are significant for both) is difficult.

Scope for reduction of risk analysis uncertainties using additional information
The experts generally agreed that, from a practical point of view, information available with national coverage would not, with a few exceptions, be likely to improve significantly in the near future. Thus the data identified as useful in reducing risk analysis uncertainties is aimed at a higher tier approach (i.e. catchment, sub-catchment or farm specific) where the Tier 1 risk designation is to be investigated and/or refined. Useful information highlighted by the experts includes:

- Site-specific soil P status data.
- Data on vegetation cover (in relation to reduced velocity of surface flows), orientated roughness and landscape structural features and how they intersect likely flow paths. These may be estimated by visual assessment basis and/or aerial photography.
- Site-specific visual assessments (by trained personnel) may provide useful information on both catchment characteristics and their interaction with farming practices and infrastructure. Visual assessments may be useful in identifying high risk phenomena, but have the associated problem that they are only one point in time. Expert 2a expressed the opinion that a visual inspection should also include interaction with the farmers (and/or other stakeholders) as building a rapport with the farmer is important.
- Farmer questionnaires may also be valuable but a good framework is needed to accept the mix of quantitative and qualitative information collected.
    - Farmers already provide Defra with much information and it is worth exploring how this may be used (although there may be problems accessing data due to issues regarding anonymity).
    - As part of the ECSFDI programme, personnel have been communicating with farmers; this information should be explored to assess its value.
- Better data regarding slurry applications (may come from farmer questionnaires or existing Defra information as identified above).
- Data from ecologically sensitive periods (i.e. seasonality important).
- For modelling applications spatial land-based data is required as catchment outlet data does not help identify P sources explicitly.
- Higher resolution catchment outlet monitoring data (e.g. see Johnes, 2007).

3.9 Research priorities

Science priorities which are implicit from the elicited fuzzy distributions are those components which were rated as highly important and highly uncertain. This is expressed simplistically in Figures 3.9 and 3.10 by the product of the ratings (the centroid of each fuzzy distribution) for each chosen component. These figures show that the most significant priority from the catchment processes subject area relates to determination of hydrological connectivity (Figure 3.9). Within the Agronomic Practice subject area there are no components which have very high ratings (this is partly an effect of the generally broader fuzzy distributions: see Figures 3.6 and 3.7 above) and many which have similar ratings (artificial drains; orientated roughness, instream processes, highways and buffer features). Where the components identified for further research were associated with high levels of uncertainty, the sources of the uncertainty are important in the determination of how likely we are to be able to reduce it by further research, e.g. by increasing our scientific understanding or by developing scale-dependent rules, or more data collection, which may help us apply our current knowledge more effectively

or help in the determination of scale-dependent rules. For the components chosen, in general, the experts rated a lack of data and spatial heterogeneity of catchment processes as the primary sources of uncertainty closely followed by temporal heterogeneity. In addition, and as a result of the aforementioned sources of uncertainty, our ability to apply our scientific understanding was also ranked highly for many components. Thus, in most cases, it is not a lack of scientific understanding per se that is the limiting factor for estimation of P pressure; it is our ability to determine the magnitude of effect of these components in space and time given limited observations. These results have implications for the way we estimate P pressure, which are discussed in Section 11.

Exceptions to these generalisations, where experts ranked our scientific understanding as limited include: the modifying effects of instream processes, buffer features, orientated roughness and public highways.

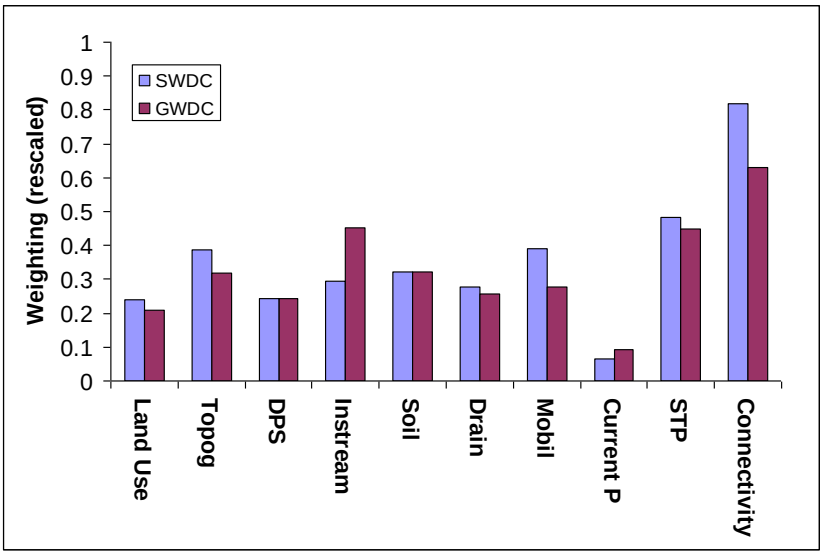


Figure 3.9 – Ratings for Catchment Processes model components with regard to research priority estimated using the product of the centriod of the importance and overall uncertainty fuzzy distributions.

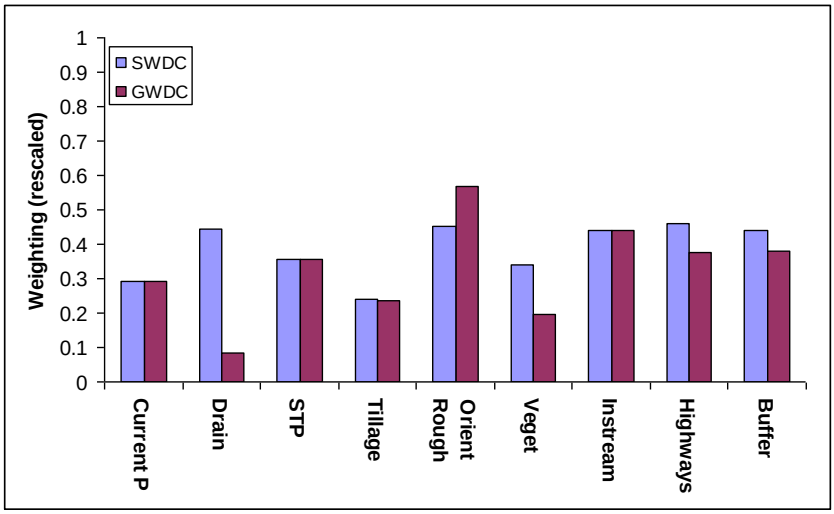


Figure 3.10 – Ratings for Agronomic Practices model components with regard to research priority estimated using the product of the centriod of the importance and overall uncertainty fuzzy distributions.

Research priorities that were explicitly elicited from the experts include:

- The overall effect of In-stream processes: including an extension of current work to small streams.
- Further investigation of the risk to the uplands.

- A focus on P loss during ecologically sensitive periods.
- More focus on defining the sensitivity of the receiving waterbody.
    - Putting land-based emissions into context with riverine load: i.e. a more holistic approach that determines the magnitude of P loss that can be sustained before there are adverse effects.
- Long-term catchment manipulation studies.
- Testing the effectiveness of mitigation measures such as buffer strips at larger scales.
- P tracing techniques need to be explored and improved.

3.10 Conclusions

The results of the elicitation from the Catchment Processes and Agronomic Practice areas have been combined to produce a simplified conceptual model of the primary factors determining the magnitude of P pressure at WFD scales. The chosen model components were dominated by physical factors that control hydrological pathways and connectivity of the landscape to surface waters and by the management of livestock. There was generally a good degree of agreement between experts on which components were included and on the fuzzy weightings assigned. The chosen components are discussed with reference to the scientific studies to support their inclusion in the model. In most cases, however, the supporting scientific evidence is based upon small scale studies making extrapolation of understanding to WFD catchment scales problematic and a source of significant uncertainty. At these scales, the spatial and temporal heterogeneity of the system makes it difficult to apply our relatively high levels of process-understanding and makes low resolution monitoring data weak for evaluation of estimation methodologies and in resolution of the effect of individual processes. The absence of scale-dependent rules meant that experts were required to 'think across scales' during the elicitation making the results somewhat subjective. Thus, for certain elements of EA WFD risk analyses it may be beneficial to collect data at WFD catchment scales such that scale-dependent rules may be derived to inform future modelling approaches (see Section 11 for a fuller discussion). The experts highlighted a number of areas as research priorities: instream processes; investigations of upland systems; more of a focus on ecologically sensitive periods; long-term catchment manipulation studies; exploration of tracer techniques and the likely effects of climate change.

4 Phosphorus Fluxes from Septic Tanks

4.1 Summary

Septic tanks rely on the natural propensity of soils to retain P from tank effluent. In the past this has been shown to be the case by the scientific community. However, recent studies have shown that septic tanks have the potential to contribute a significant proportion of the P to which waterbodies are exposed in some cases (Arnscheidt et al., 2007), primarily for rural areas. However, the evidence base to support this remains limited, uncertain and sometimes anecdotal from localised observations (Dudley and May, 2007). A particular weakness of the evidence base is quantitative source apportionment studies.

Estimation of the P flux from septic tanks can be operationally divided into three principal areas:

- The number of people connected to a tank, or to tanks distributed in any given area.
- The functioning and maintenance of the tank(s).
- The degree of P attenuation/retention afforded by the soil, drift and underlying geology.

With little quantitative evidence of the magnitude of P fluxes (particularly at larger scales) most attempts to quantify these factors will be highly uncertain and this has generally resulted in the use of crude population estimates and generalised per capita export coefficients from the literature.

At all scales, without more quantitative studies that accurately apportion the relative magnitudes of P flux between sources, there is likely to be little improvement in our confidence in estimates of P fluxes from septic tanks. Better quantitative information may allow simplistic rule-based approaches to estimate the likely variability in flux given certain catchment properties (such as an index-based system called for by Gold and Sims, 2000). Perhaps one of the most positive directions for future research is in the use of tracers, although more progress will need to be made for quantitative source apportionment (Dudley and May, 2007).

In the sections below, methodologies for estimation of P fluxes from septic tanks and the mechanisms which determine the likely magnitude subsurface attenuation are discussed. These attenuation mechanisms also apply to other P sources and in particular those similar in nature to septic tank effluent plumes: such as leakage from unlined farm waste stores (Gooddy, 2002; Armstrong *et al*., 2004; see also Section 8) and leakage from sewers (Holman *et al*., 2008).

4.2 Introduction

The likely magnitude of P flux from septic tanks has been highlighted as an issue in recent years (e.g. see Dudley and May, 2007). Previously, however, non-WwTW P fluxes were assumed to be primarily from agricultural sources. This was partly because of a large body of published evidence (principally from the USA) indicating that most of the phosphorus emitted from septic tanks is retained in soils close to the septic tank percolation area, resulting in a low proportion of phosphorus reaching surface waters (e.g. see the review of Gold, 2006 and Gill *et al*., 2007 where septic tanks were shown to provide a similar level of treatment to packaged secondary treatment systems). It is possible that many of these publications studied well-maintained and well-located systems in more sparsely populated rural areas than those in England and Wales. Problems arising from septic tanks are likely to be situation specific and, even if only a small fraction small fraction of inputs reach a water body, may be significant

cumulatively in some areas, although quantitative evidence to support these claims is lacking. Fluxes are dependent on septic tank function and maintenance, soil/geological characteristics and proximity to watercourse (Ptacek, 1998; Robertson *et al*., 1998). In the UK and the Republic of Ireland there are examples of significant P fluxes from septic tanks; e.g. see Carvalho *et al.* (2005) and Arnscheidt *et al*. (2007). Dudley and May (2007) present results from reviews of multiple catchment studies where estimated septic tank fluxes generally ranged between 1 and 22% of the total annual flux and where one study estimated a contribution of 40-76%. Hence, it is now thought that septic tanks have the potential to contribute, in particular for rural catchments, a significant proportion of P to waterbodies, although the evidence base for this is limited, uncertain and sometimes anecdotal (Dudley and May, 2007). It is essential to note that the importance of septic tanks will vary hugely between sparsely populated rural catchments where streams may be more sensitive to septic tank P inputs compared to larger and/or more populated areas where most dwellings are connected to the sewerage system and septic tank inputs may be insignificant.

The controls on the magnitude of the contribution can be operationally divided into three principal areas which will be expanded upon below:

- The number of people connected to a tank, or to tanks distributed in any given area.
- The functioning of the tank(s).
- The degree of P attenuation/retention afforded by the soil, drift and underlying geology.

A conceptual model of the main components to be taken into account when estimating P fluxes from septic tanks is presented in Figure 4.1. This conceptual model was the focal point for the interview phase of the elicitation.

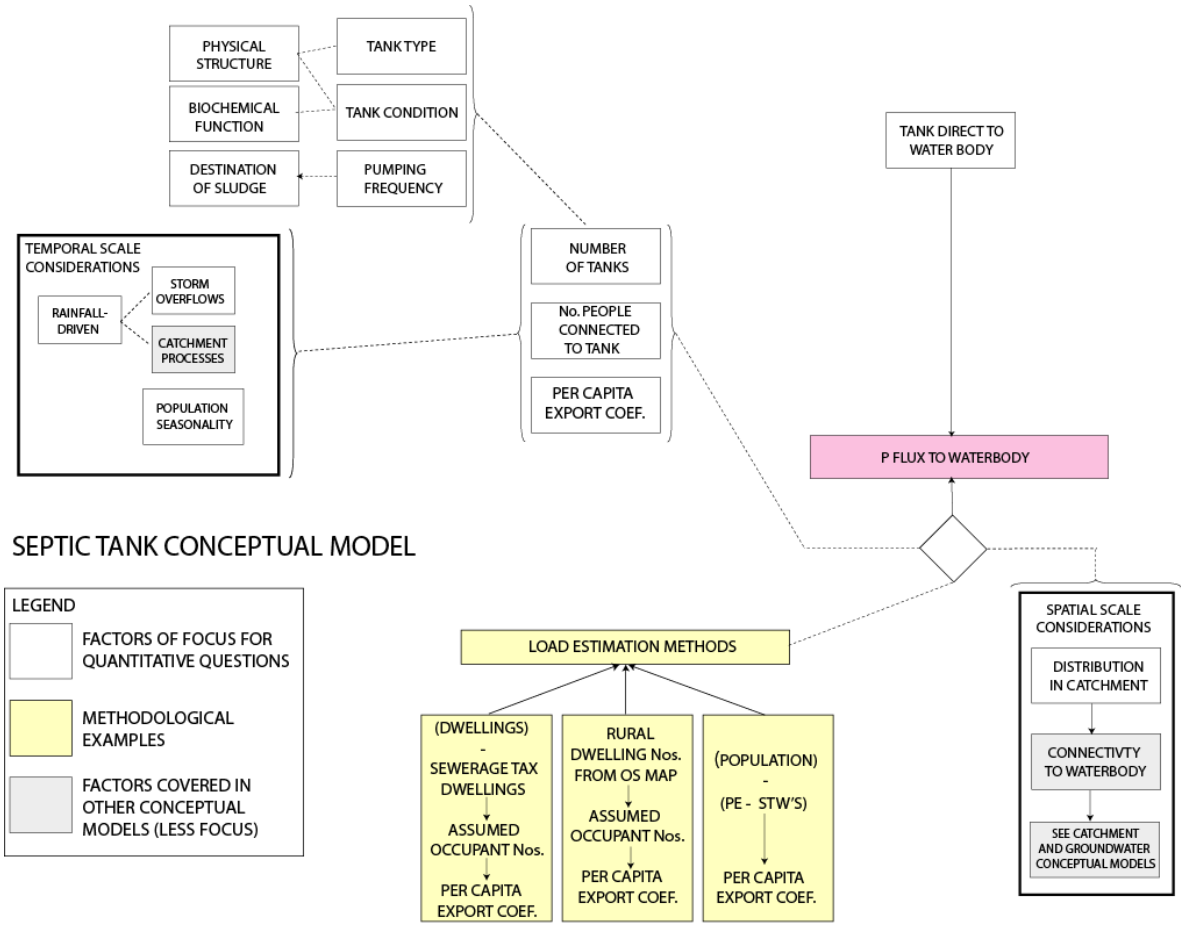


Figure 4.1 – Interview phase conceptual model for estimation of P fluxes from septic tanks.

4.3 Estimating the number of people served by septic tanks

At the scales of interest for the Environment Agency's Tier 1 risk assessments, the number of people connected to septic tanks will probably need to be estimated using the difference between the total number of people in a catchment (e.g. from census data) and the number of people connected to a sewer (e.g. from water company data, if available). The main uncertainties in these estimates relate to transient populations such as tourists and students. Tourists may be a particular problem as they increase the potential for phosphorus (P) loss during the summer when water body ecology may be more sensitive to increase P level. Tourist numbers are difficult to estimate as only an approximate number of beds can be obtained from tourist boards; there is no estimate of bed occupancy or division between sewered and non-sewered accommodation.

For higher tiered approaches, more detailed GIS- or survey-based approaches may be practical in determination of population served (e.g. see Carvalho *et al*. 2005) and may also provide information of tank type and levels of maintenance (e.g. see LLCMMS, 2000).

4.4 Septic Tank Function

Septic tank function is primarily controlled by the following:

- Tank Type
- Tank Condition
- Tank Maintenance (e.g. pumping frequency)
- Storm Overflows (linked to tank type)

Tank type, condition and maintenance are known to be important (Dawes and Goonetilleke, 2001) but at larger scales, where a full survey in impractical, there is large uncertainty associated in our knowledge of these factors. Where surveys have been carried out, they have shown that there can be a high proportion of residences that are unaware that pumping of sludge is required (Expert 1p; LLCMMS, 2000) and that a significant number of systems do not function properly owing to poor construction, installation and maintenance or because they are located in areas with unsuitable subsoils (EPA, 2000). Storm overflows can also cause problems where rainwater from roofs and paved areas discharges to a tank, a practice which contravenes guidelines (Environment Agency, 2006; EPA, 2000).

4.5 Connectivity to water body and subsurface attenuation

The location of a tank with respect to a water body (both surface and groundwater) and the localised physico-biogeochemical characteristics are critical in determining what proportion of P in tank effluent reaches the water body and what is attenuated (Dawes and Goonetilleke, 2001; Gold and Sims, 2000). This is controlled primarily by soil type and drift/geological characteristics, but also on the contaminant plume itself (as sorption of P to the soil is reduced under anaerobic conditions: Ptacek, 1998; see also Section 8). In general we will have more confidence in the functioning of newer installations as control of septic tank location is now by consent where the suitability of a site will have been assessed (Environment Agency, 2006). However, with older tanks their location may not be ideal and some are known to discharge directly to surface waterbodies (Patrick, 1988). Published retention coefficients for septic tanks are few and are likely (necessarily) to be somewhat arbitrary (e.g. Endreny and Wood, 2003 used a coefficient of 0.5 owing poor soil drainage and the high stream density; the International Centre for Island Technology, 2004 used a factor of 0.25 for relatively flat land where houses were generally in close proximity to watercourses). Where estimation of likely retention is

required, catchment characteristics can be approximated using soil and geological maps and it may be possible to derive some simplistic rules (Ebers and Bischofsberger, 1987; Gold and Sims 2000). For example, Ptacek (1998) and Robertson *et al*. (1998) concluded that septic tanks situated within one hundred metres of surface waters or in areas with calcareous sandy soil are likely to be problematic; Reneau *et al.* (1989) similarly identified coarse textured soils with low P sorption capacity and soils with poor effluent distribution to have the highest likelihood for P loss. Simplistic rules will be complicated by localised heterogeneity that can lead to significant uncertainty. Additionally, studies have shown that the effectiveness of soils to retain P declines with time as sorption sites become saturated and hydrologic flowpaths become altered due to clogging, both of which reduce P retention. (Gold and Sims, 2000).

4.6 The Export coefficient approach

The difficulties in estimating *per capita fluxes to septic tanks* and subsequent estimates of *P retention* has led to the use of more simplistic *direct* (to waterbodies) *per capita export coefficient* estimates (i.e. direct from people served to flux to water body). Methods to estimate export coefficients vary and many literature values can be traced back to a few early estimates. Export coefficients are likely to be very uncertain as they are often calculated on the basis of poor estimates of population served and by poor estimates of P flux that can be attributed to septic tanks (i.e. there are no well-founded, large scale source apportionment studies based upon accurate data). Export coefficients published in the literature show significant variation (Table 4.1) but it is not apparent if this results from variation in septic tank function and soil/geological properties or methodological uncertainties. Without ways of predicting tank function and likely P attenuation that have been constrained by adequate data, identification of representative export coefficients for a given study is challenging. With better data (and in particular accurate source apportionment) it may be possible to combine simplistic sub-surface connectivity rules (e.g. based upon the hydrological properties of soils and geology) and likely retention coefficients (based upon soil geochemical properties) with estimates of tank losses.

Table 4.1 Per capita estimates of annual TP flux from WwTW.

P kg capita^{-1} year^{-1}	**Pre or post treatment**	**Source**
0.5	Flux to tank*	Sarac *et al*., (2001)
0.74 - 3.00	Flux to tank*	Endreny and Wood (2003)**
0.5 – 1.8	Flux to tank*	Kramer *et al*. (2006)
0.24 (TP)	Flux to water body	Hanrahan et al. (2001)
0.33	Flux to water body	Frost (1996)
0.4 (SRP)	Flux to water body	Foy and Lennox (2000)
0.26 (MRP)	Flux to water body	LLCMMS (2000)
0.44 (TP)	Flux to water body	Smith *et al*. (2005)
0.3 (TP)	Flux to water body	Carvalho et al. (2003)

** See also the per capita estimates for raw effluent to WwTW in Table 5.1, Section 5); ** From a review of 8 publications*

4.7 Conclusion

Septic tanks use the natural propensity of soils to retain P and historically the scientific community has shown this to generally be the case. However, it is now recognised that septic tanks have the potential to contribute a significant proportion of P to waterbodies, although the evidence base to support this is limited, uncertain and sometimes anecdotal (Dudley and May, 2007). The P flux from septic tanks may primarily be as a result of poor tank construction, maintenance and location (e.g. on poorly retentive (or fractured) soils and geology (Montgomery, 1988), close to a water body, close to or intersecting the groundwater table, or in some cases directly discharging to a water body. With little quantitative evidence of the magnitude of P fluxes (particularly at larger scales) most attempts to quantify them will be highly uncertain and often as a result in the use of literature values for export coefficients or retention coefficients.

At all scales, without more quantitative studies that apportion the relative magnitudes of P flux between sources there is likely to be little improvement in our confidence in estimates of P flux from septic tanks. Better quantitative information will allow simplistic rule-based approaches to estimate the likely variability in flux given certain catchment properties (such as the index-based system called for by Gold and Sims, 2000). Perhaps one of the most positive directions for future research is in the use of tracers, although more progress will need to be made for quantitative source apportionment (Dudley and May, 2007).

5 Wastewater Treatment Works P fluxes

5.1 Summary

The calculation of accurate point source P fluxes from Wastewater Treatment Works (WwTW) is crucial for source apportionment studies, which serve to isolate the diffuse P contribution to catchment fluxes estimated using EA GQA data. Isolation of diffuse contributions would subsequently help constrain models of diffuse P flux (see Section 3 and Section 11). The uncertainty associated with point source flux estimates is derived mainly from the type of data available for calculation; there is likely to be a gradual increase in confidence from methods that used generalised back-calculated export coefficients (per capita export coefficients) to those using measured flows and measured concentrations from WwTW outfalls (see the schematic representation in Figure 5.1). The latter may now be possible with the current availability of WwTW daily flow data coupled with P concentration data collected by the EA. Calculation of fluxes in this way is time-consuming and expensive, but would serve as a good estimate of point source P flux for a number of years unless upgrades in treatment type change P concentrations radically at individual WwTW. This new information will also allow estimation of P fluxes at higher resolution and in particular for periods of low flow during ecologically sensitive periods where the risk from P is likely to be greater (see Section 9).

5.2 Introduction

Estimation of Wastewater Treatment Works (WwTW) P fluxes, including an estimate of the associated uncertainty is critical, particularly for point source dominated catchments where the diffuse signal may be of the same order of magnitude as the uncertainty associated with the WwTW flux. The magnitude of the uncertainty for any given WwTW is dependant upon the data available to calculate fluxes and incidental factors such as storm overflows. There are two primary ways to estimate P flux: using an estimated number of people (or population equivalent – PE - which can include non-human sources such as from industry) and a per capita export coefficient or using the product of WwTW flow and P concentration estimates. There are a number of data-dependant methods to achieve these with differing degrees of confidence: these are discussed below, are reviewed in Carvalho *et al*. (2005) and are shown schematically in the pre-interview conceptual model presented in Figure 5.1.

5.3 Calculation of WwTW P fluxes

5.3.1 Data available for calculation of WwTW P flux

- Water Company estimated PE is available for some large WwTW but there is not a full record and data are fewer for small works (Expert 3p).
- Estimates of WwTW treatment type (primary-secondary-tertiary) for some large WwTW and some records for small works (Expert 3p).
- EA ***consented*** average daily dry weather flow (DWF: *the total daily flow value that is exceeded by 80% of the total daily flow values in any period of twelve months*).
- Water Company ***estimated*** average daily DWF: normally lower than that consented (see below).
- Water Company ***measured*** daily flows (measured daily – not an average daily flow).
- Measured concentrations (EA WIMS database: approximately monthly).

5.3.2 Methods for calculating point source fluxes

The methods for calculating point source fluxes listed below are arranged in an order thought to represent increasing confidence in estimates made. The confidence level for the flux from a specific WwTW will, however, vary depending upon specific factors, which are discussed in the sections below. The main methods considered here combine the following information:

1. A PE estimate combined with a treatment-dependent export coefficient.
2. A measure of DWF combined with some averaged concentration value (or statistics to include a confidence estimate).
3. An estimated daily flow and some averaged concentration value (or statistics to include a confidence estimate).
4. Measured daily flows and some averaged concentration value (or statistics to include a confidence estimate).
5. Measured daily flows and a flow-P concentration relationship determined by a closer analysis of the water company flow files and WIMS database concentrations. This would be very time-consuming unless an automised method could be used.

The experts expressed the opinion that, although time-consuming and costly, a good analysis of the water company flow data and WIMS concentration data for a broad range of consented discharges would be informative in itself as well as providing more reliable (less uncertain) estimates of point source fluxes (e.g. a comprehensive attempt at methods 4. and 5.). The point source flux estimates would be valid for a long time: although individual upgrades to treatment would add uncertainty.

Uncertainties associated with each data type:

Population Equivalents
Population Equivalents is a measure of the effective number of people served by a WwTW, which may include other sources such as from industry (but expressed as PE units). PE data is available from WIMS for some WwTW or can be estimated from census information (where an estimate of the unsewered population will need to be deducted) or from water company records of number of dwellings paying utility bills with an assumed occupancy rate. There can be significant differences between the actual PE connected to the WwTW and the *consented PE* or the *design PE* that is often reported (Carvalho *et al*., 2005). It's also difficult to account for transfers of sewage between catchments

Population Equivalents can be highly uncertain where there are large transient populations, such as tourists and students. This is a significant problem as tourist populations are likely to increase during the growing season when there are generally lower flows (i.e. less dilution of effluent) and rivers may be more sensitive to elevated P concentrations. The tourist population is very difficult to estimate as, although there may be estimates of number of 'tourist beds' available, there are no known records of actual bed occupation.

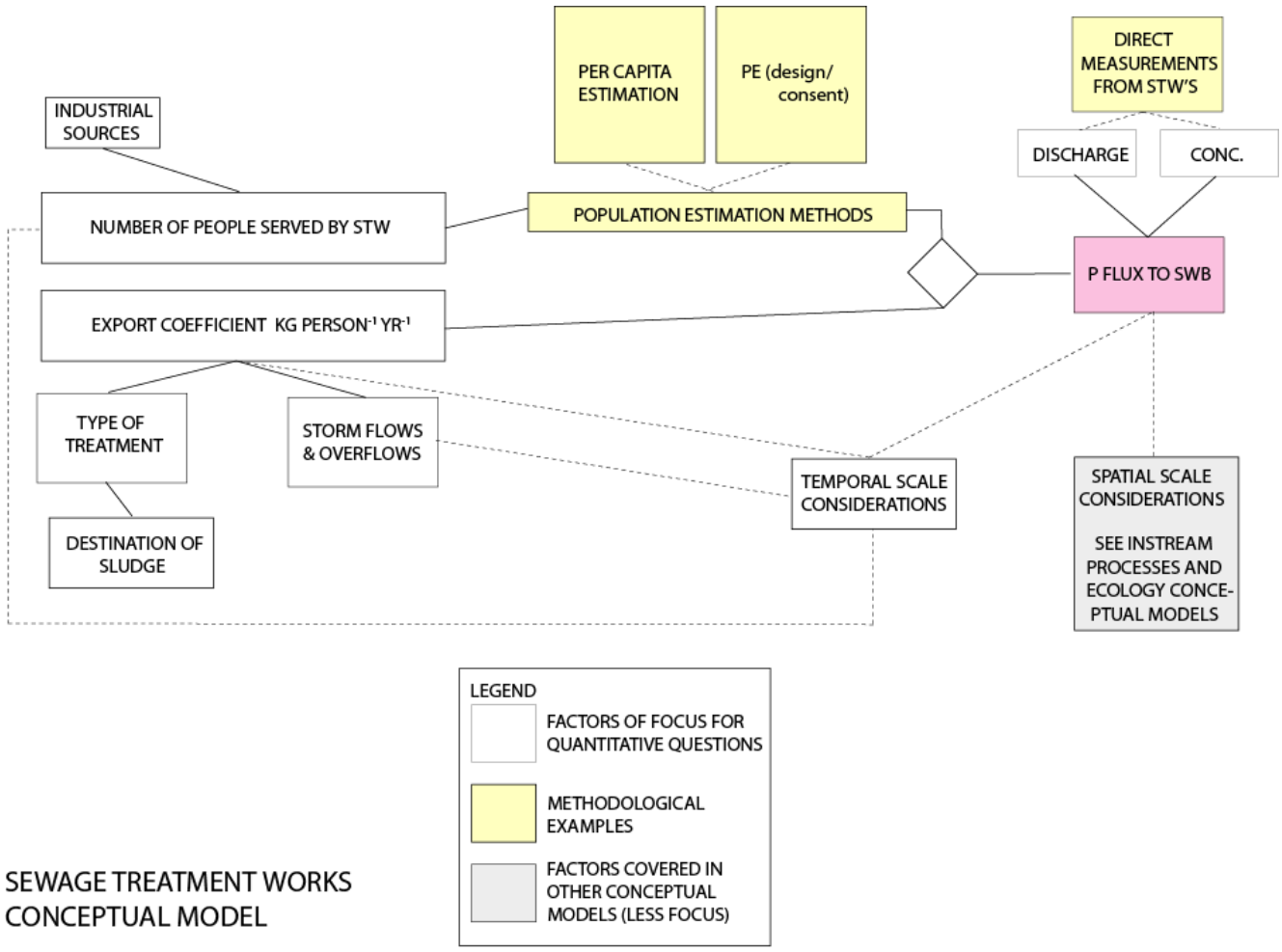


Figure 5.1– Interview phase conceptual model for estimation of P fluxes from WwTW.

Per Capita P Export Coefficients

Export coefficients (often estimated from a back-calculation of observed loads determined in particular catchment case studies) have been generated for P load per capita untreated and P load per capita treatment-specific. Literature on per capita export coefficients are sometimes vague, they are not always clear on what method was used to calculate them and what level of treatment was used at the WwTW. However, there seems to be some world-wide (assuming developed world) consensus on the approximate magnitude of the export coefficients, although there is still significant variation (see Table 5.1).

Table 5.1 Per capita estimates of annual TP flux from WwTW.

TP kg capita^{-1} year^{-1}	Pre or post treatment	Source
1.2	Pre treatment	Deevey and Harkness (1973)
0.76	Pre treatment	Gray (1984)
0.82 -1.03	Pre treatment	Foy *et al*. (1995)
1.0 –1.5	Pre treatment	Jones *et al*. (1979)
0.8	Pre treatment	Brylinsky (2004)
1.1	Post treatment**	Brigault and Ruban (2000)
0.62	Post treatment**	Morse (1993)
0.38	Post treatment**	Johnes (1996)
0.42	Post treatment***	White and Hammond (2007)

** includes primary and secondary treatment or was unspecified.
*** based on 62 large WwTW in the Severn Trent Region.

Where the type of treatment is taken into account, the efficiency of treatment is in general of lower uncertainty compared to other sources (Expert 2p). However, knowledge of the type of treatment is critical (Foy, 2007) as it can change the P content, compared to raw inputs, significantly (particularly for tertiary treatment which can reduce P loads by between 40 and 95%: Carvalho *et al*., 2005). Treatment itself is approximately constant, but there is a potential for incidental problems (which may change its efficiency) resulting from:

- Storm overflows;
- Engineering breakdown;
- Occasional non-compliance;
- Under capacity: e.g. under-sizing or out of date sizing of systems (where populations have increased or where tourism is not taken into account adequately).

Storm overflows are a particular problem, as they are not well known (e.g. a WwTW may or may not have a storm effluent storage facility and there is little or no information regarding this). Additionally, storm overflow outlets may not be co-located with the normal WwTW outfall.

WwTW flow estimates
The EA and water company DWF estimates are subject to high uncertainties; analysis of the similarity between the 2 datasets shows that for 40% of WFD catchments the two estimates are not within 50% of each other. The distribution of water company to EA flow ratios is shown in Figure 5.2 where it can be seen that water company estimated (not measured) average daily flows are primarily between 0.8 and 2.5 times the EA consented flow giving a potential for considerable bias in any flux calculation (EA consented flow data is the basis for calculations within the Agency SIMCAT model). Use of Water Company measured daily flow data is likely to drastically improve estimates of flow and allow higher resolution analysis of P fluxes for given periods (see WHS, 2008).

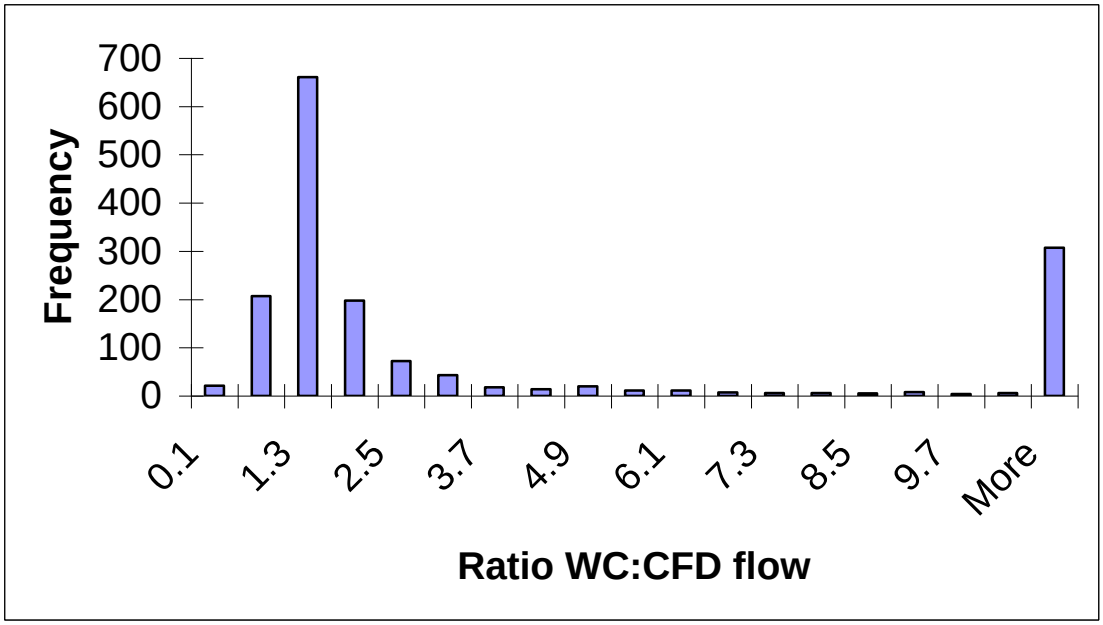


Figure 5.2 Distribution of the water company estimated (WC) (not measured) average daily flow to EA estimated dry weather flow (CFD) for approximately 1600 WwTW.

WwTW concentration estimates
The EA have approximately monthly samples from WwTW outfalls (approximately 3500) across England and Wales. As present, there is no link between the WwTW flows and measured concentrations. This being the case, flux estimates will be made using distributions or moments

of these distributions for given WwTW (methods 2, 3 and 4 above). An ongoing project (WHS, 2008) is, however, providing a link between measured WwTW flow data and P concentration measurements.

Small WwTW
For smaller WwTW (e.g. packet works), where the maintenance is the responsibility of the home owners connected, it was felt that where maintenance contracts lapse they may pose a significant problem: particularly as they are directly connected to the stream (Expert 1p). This is converse to most septic tank systems where, even with poor maintenance, there will still be the potential for significant attenuation by the soil (see Section 4).

Other uncertainties
There are uncertainties associated with the disposal destination of sludge from WwTW: this may however be taken into account if spatial datasets describing sludge application to land are included in terrestrial models as P inputs.

5.4 Conclusion

The calculation of accurate point source P fluxes is crucial for source apportionment studies, which serve to isolate the diffuse P contribution to catchment fluxes estimated using EA GQA data. Isolation of diffuse contributions would subsequently help constrain models of diffuse P flux (see Section 3) as calculation of diffuse loads by difference using highly uncertain point source flux estimates is problematic (particularly for catchments where the diffuse contribution is relatively small and hence could be smaller than the error associated with the point source estimate). The uncertainty associated with point source flux estimates is derived mainly from the type of data available for calculation; there is likely to be a gradual increase in confidence from methods that used generalised back-calculated export coefficients (per capita export coefficients) to those using measured flows and measured concentrations. The latter may now be possible with the current availability of WwTW daily flow data coupled with WIMS P concentration data. Calculation of fluxes in this way is time-consuming and expensive, but would serve as a good estimate of point source P flux for a number of years unless upgrades in treatment type change P concentrations radically at individual WwTW. This new information will also allow estimation of P fluxes at higher resolution and in particular for periods of low flow during ecologically sensitive periods where the risk from P is likely to be greater (see Section 9).

6 Instream Processing of Phosphorus

6.1 Summary

Instream processing of phosphorus is of particular importance for the Environment Agency's risk assessment as:

- many of the models currently used to predict diffuse P fluxes make estimates of P delivered to the stream and do not explicitly take into account instream processes: although as little data exists to evaluate models at the terrestrial-aquatic boundary, some instream processing may be implicit in catchment delivery models. Exceptions to this are the Kennet Model and INCA-P which model in-stream processes explicitly (Wade *et al*., 2007; Wade *et al*., 2001; Wade *et al*., 2002a);

- instream P measurements made as part of the EA's GQA monitoring strategy are often measured upstream or downstream of WwTW discharges and hence may not be representative of an entire WFD catchment or reach. Thus instream processing of P should be taken into account to put these observations into context with their proximity to upstream P sources.

Phosphorus fluxes from terrestrial systems, in soluble and/or particulate-attached forms, are likely to undergo numerous physical, chemical and biological transformations during downstream transport. These interacting processes act at many timescales from effectively instantaneous chemical reactions to the gradual and stepwise transfer of P and sediment during hydrological events. The significance of these processes, with regard to the risk from P of degraded ecological status, is primarily associated with the net retention of P, and more specifically the attenuation of instream bioavailable P concentrations during low flow periods in the growing season which may subsequently be released during less sensitive periods (e.g. autumn and winter; Svendsen *et al.*, 1995). The P retained during these ecologically sensitive periods may be flushed during autumn and winter storms and may hence have subsequent effects on lakes and transitional and costal waters. Thus the importance of the temporal dynamics of P processing is system-specific as all P fluxes are potentially bioavailable unless lost to a 'true sink' (i.e. attached sediment which is not resuspended or where desorption from the sediment does not occur, as in the sediment of some lakes). There are hence two primary considerations for instream processing for WFD risk analyses: the attenuation and release of growing season river P concentrations and the net effect of P release and retention over annual or longer timescales.

Estimation of the likely magnitude of P processing by instream is advantageous for risk analyses, particularly where there are no GQA monitoring locations in the WFD catchment of interest. Where GQA sampling is in place, and these data are the principal measures of stream P status, the net effect of all instream processes occurring upstream of a given sampling point are implicitly taken into account. Where GQA monitoring data exists for a WFD catchment, the primary concern relates to the representativeness of the GQA data (i.e. the location of the GQA sampling point with respect to the WFD catchment boundaries and its proximity to point source P emissions.

At WFD scales, the identification of attenuation/retention and release processes occurring, their relative magnitudes and temporal variation is highly uncertain as mass balance studies cannot identify which processes are operating and it is difficult to scale up from small scale studies that can (e.g. Wade *et al*., 2007). Furthermore, given the uncertainties associated with reach and catchment P budgets, estimated retention/release may represent error or methodological discrepancies rather than true fluxes (Moss *et al*., 1988). In the face of such difficulties, it may be the case that a more simplistic approach is taken to estimating the likely magnitude (and

direction) of instream P processing. Figure 6.1 incorporates the most important components that control P processing chosen by the experts and represents their simplified WFD catchment scale-dependent conceptual model. Central to the model is the role of flow regime as a control on P dilution (particularly from large point sources) and reach residence times: which have an effect on the time available for interaction between river water and sediment and the resupply of P for uptake by aquatic plants. This simple model is set in the context of the variability of the effects of the model components with seasonally differing flow regimes, light and temperature and, as discussed above, in context with the proximity of a given reach or monitoring point to point source input(s). The usefulness of this model (over and above an expert system) requires that a significant amount of reach/catchment scale data be collected with which to develop rules that define the likely magnitude of effect of each model component for a given system, or the likely effect of given river typologies.

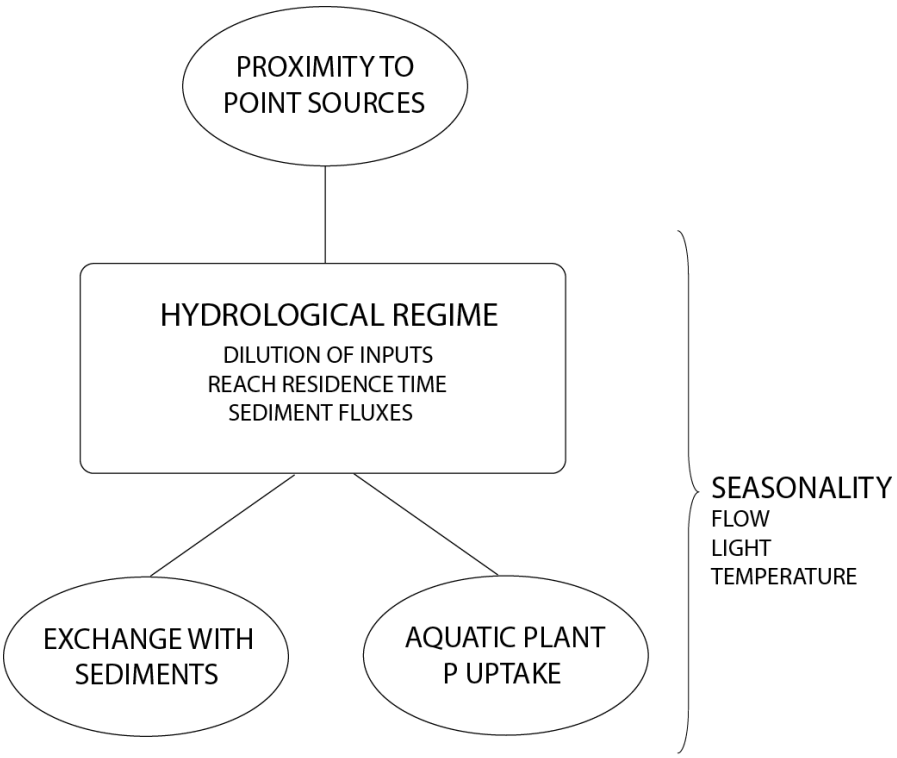


Figure 6.1 – Post-interview phase simplified conceptual model of instream P processes.

6.2 Introduction

Phosphorus fluxes into surface waters in a river basin are normally larger than those observed at the river mouth (Wassman and Olli, 2004). The flux of P is observed to be non-(mass) conservative and the difference is described as *retention*: a collective expression for biogeochemical and hydrological processes that decrease, decay, degrade, transform, either temporally or permanently, P in rivers or floodplains (Reddy *et al*., 1999). These processes alter the quantity and timing of phosphorus fluxes to downstream aquatic systems (Demars *et al*., 2005) as well as altering P form. Long term annual mass balance studies of P have highlighted the great variability of phosphorus retention at catchment scales (Meyer and Likens, 1979; Demars *et al*., 2005) but cannot identify the processes involved or their relative magnitude. Identification of discrete processes required direct measurement at research sites (Svendsen & Kronvang, 1993) or smaller scale fluvarium experiments (McDowell and Sharpley, 2003) but difficulties remain in upscaling such measurements to whole river systems. In addition, Moss *et al*., 1988 warn that, given the uncertainties associated with catchment P budgets, estimated retention may, in some cases, represent error or methodological discrepancies rather than true retention. The expert weightings discussed below are a scale-dependent attempt to quantify the likely effect of some of the processes that may effect P retention and instream concentrations.

It is worth noting that many literature studies of the retention/release of P in streambed sediments indicate that the sediments can behave as a sink for P over varying timescales and that some large rivers behave as 'true sinks', as would be the case in deep lakes where sediment gradually accrues over time. For the rivers of England and Wales, which are relatively short and have relatively short reach residence times, true sediment sinks may be of limited importance (although see the work of Demars *et al*., 2005 in the Norfolk Broads) and may perhaps be better described as temporary stores which may be flushed during high flow events (as shown for Danish streams by Svendsen and Kronvang, 1993). Of particular interest here, instream retention/release has been shown to buffer, to some extent, the impact of point sources of P (Moss *et al*.,1988; Haggard *et al*., 2001; Cooper *et al*., 2002; Marti *et al*., 2004) but that the net *uptake lengths*[φ] were much longer for nutrient enriched systems (Haggard *et al*., 2001; Marti *et al*., 2004; Haggard *et al*., 2005) compared to less impacted systems (Meyer and Likens, 1979) owing to P limitation. There are far fewer measurements in nutrient-poor systems but the work of D'Angelo *et al*. (1991), Meyer and Likens (1979) and Mulholland *et al*. (1985) investigated the role of organic matter, temperature, microbial activity and stream discharge in the P retention capacity of headwater streams.

In this section we consider the likely effect of instream processing of P: i.e. the instream sources and sinks which modify the observed P flux delivered from the terrestrial environment. Importantly, we will consider the temporal variation of source and sink terms with particular focus on the modification of instream P concentrations during critical periods: growing season low flow periods. Ideally, what is required are some simple rules that may allow us to determine the likely effect of instream processing on the terrestrial P fluxes estimated by measurement or modelling. In reality, this is problematic as, although many terrestrial P loss models nominally estimate P loss to the edge of the field/stream they will often have only been calibrated on poor edge of field data or on catchment outlet data: the latter meaning that the instream processing of P is 'absorbed' into other calibrated parameter values.

The link between stream concentrations and ecological effects in not considered here although all three experts expressed the concern that it is wrong to assume a simplistic link between P status and poor ecological quality as physical drivers such as hydrology and geomorphology can be very important: see Section 9 for a discussion.

6.3 Assumptions, definitions and caveats

The definition of importance for quantitative questions is: *the importance of a given component with respect to the modification of instream P concentrations (nominally SRP or TRP).*

The agreed typologies were: Generic upland and generic lowland which implicitly include the flashiness of the hydrological regime (e.g. as described by BFI).

There was an assumption that large point source emission (i.e. WwTW) locations and magnitudes are known for the purposes of this subject area.

Only one data scenario was considered during the quantitative phase of the interview: nationally available data.

6.4 Expert weightings for chosen model components

For model components chosen by the experts, where necessary, there is a definition in the relevant sections that follow. The experts chose only to distinguish the typologies: upland and lowland rivers with flashy and stable flow regimes as defined above. Composite (a combination of the 3 fuzzy distributions elicited from the experts: see Section 2) *importance* fuzzy weightings for the chosen components are presented in the left-hand pane of Figure 6.3 for flashy rivers and Figure 6.4 for stable rivers. Comparison of the importance and uncertainty weightings

[φ] *The downstream decline in P concentration relative to a conservative tracer (Webster and Ehrman, 1996).*

between typologies highlights that, although the experts chose to distinguish between flashy and stable rivers there was very little difference between the composite weightings given to individual components (apart from the role of P speciation). The experts showed consensus and a high degree of certainty (indicated by the narrow distribution) that hydrological dilution, proximity to point source, reach residence time and seasonality effects are critical components. From the qualitative answers elicited this was primarily as a result of effects in relation to point sources which were thought to dominate many catchments (see the discussion below). These ratings have served to identify the components of the simplified conceptual model presented in Figure 6.5 below and also

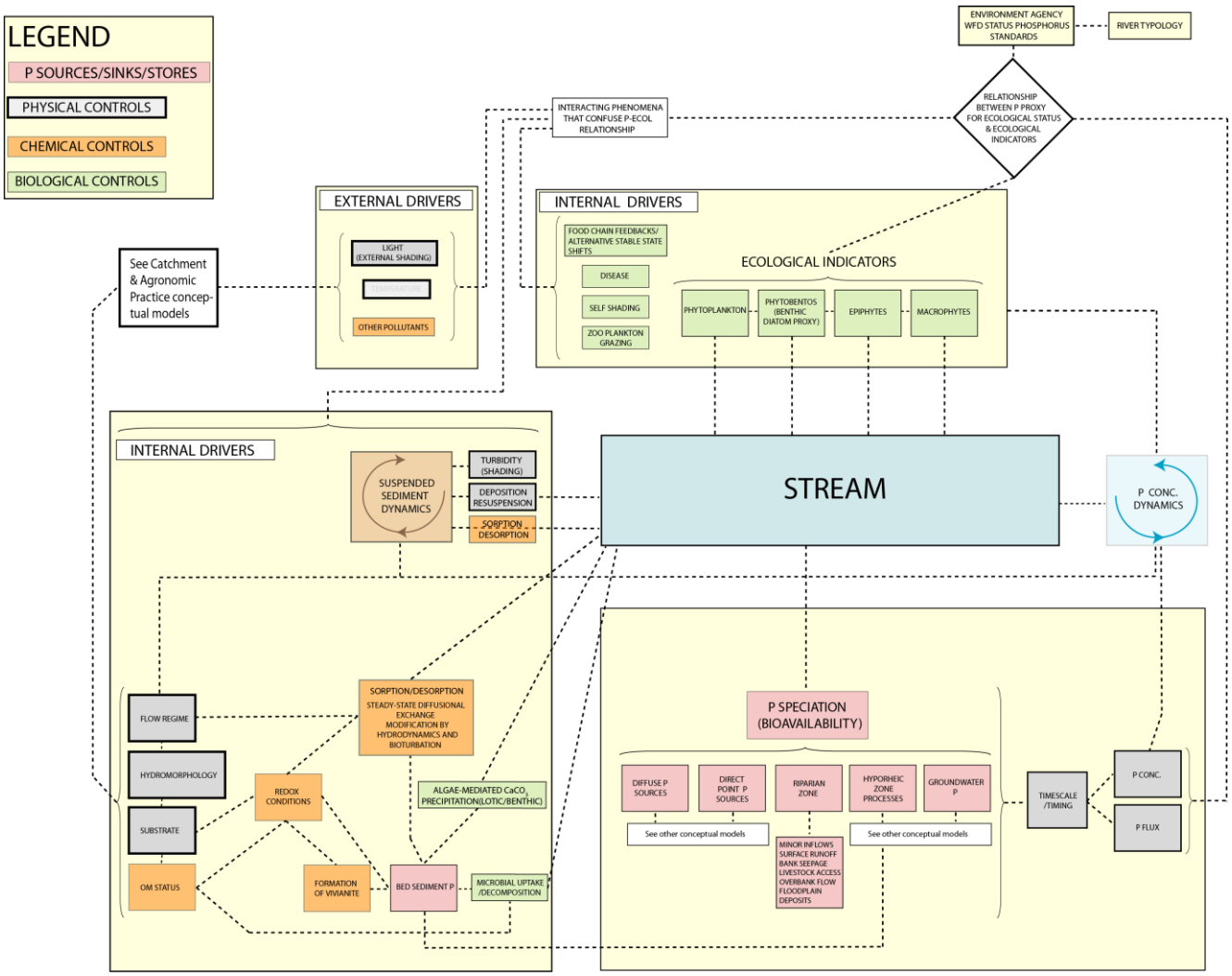


Figure 6.2 – Pre-interview phase conceptual model for instream processes.

to identify which are associated with the highest levels of uncertainty (i.e. plant uptake, exchange with sediments and P speciation). The combination of importance rating and uncertainty rating is an implicit measure of research prioritisation, as presented below. However, depending upon the sources of the uncertainty it may not be practical to prioritise certain areas of research in terms of cost-benefit or to re-frame question at the scale of interest, rather than small-scale detailed science. For example, the role of plant uptake and P speciation were given high rankings for uncertainties associated with our scientific understanding such that experiments to enhance our understanding may be required. However, they were also ranked as being uncertain with respect to spatial and temporal heterogeneity, which may mean that even a large number of small-scale experiments may not improve our estimates of their role at WFD reach scales. In this situation it may be best to invest in reach-scale experiments with

enough measurements to estimate the roles of the primary factors (e.g. House and Warwick, 1998). Similarly, the role of exchange with the sediments was thought to be well understood, but that it is difficult to predict in time and space without a very large amount of data owing to heterogeneities. In this case similar reach scale studies may help resolve the net effect of sediment exchange at the spatial and temporal scales of interest.

In the sections that follow, the underpinning scientific basis (published in the scientific literature) for each of the primary components will be briefly reviewed and assessed for suitability for incorporation into national scale risk analyses. Suitability will be assessed based upon whether or not it is feasible to upscale observations made to WFD reach/catchment scale given the information currently available at catchment scale (i.e. a Tier 1 risk assessment). Although not explicitly quantified by experts with fuzzy weightings, the potential benefit to risk analyses of higher tier data collection will be discussed, with particular emphasis on: the type and resolution of data that would be required to significantly reduce risk analysis uncertainty.

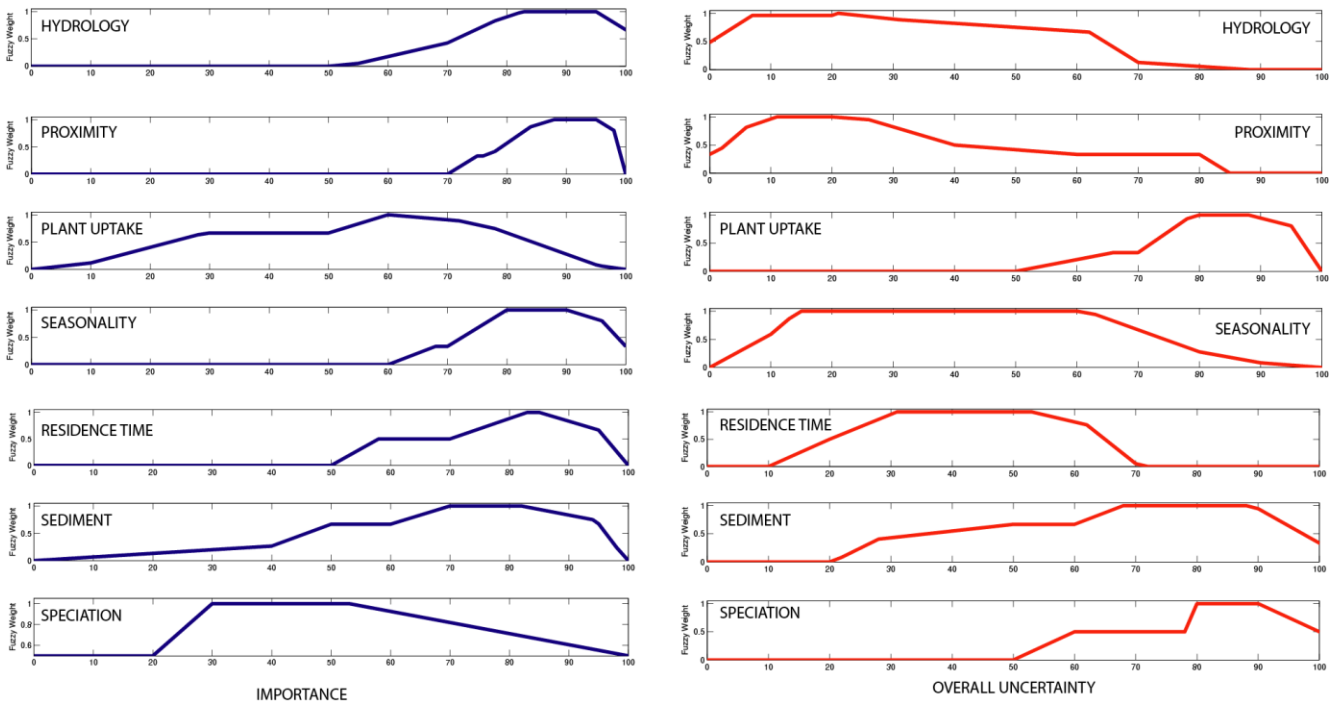


Figure 6.3. Flashy Rivers: Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

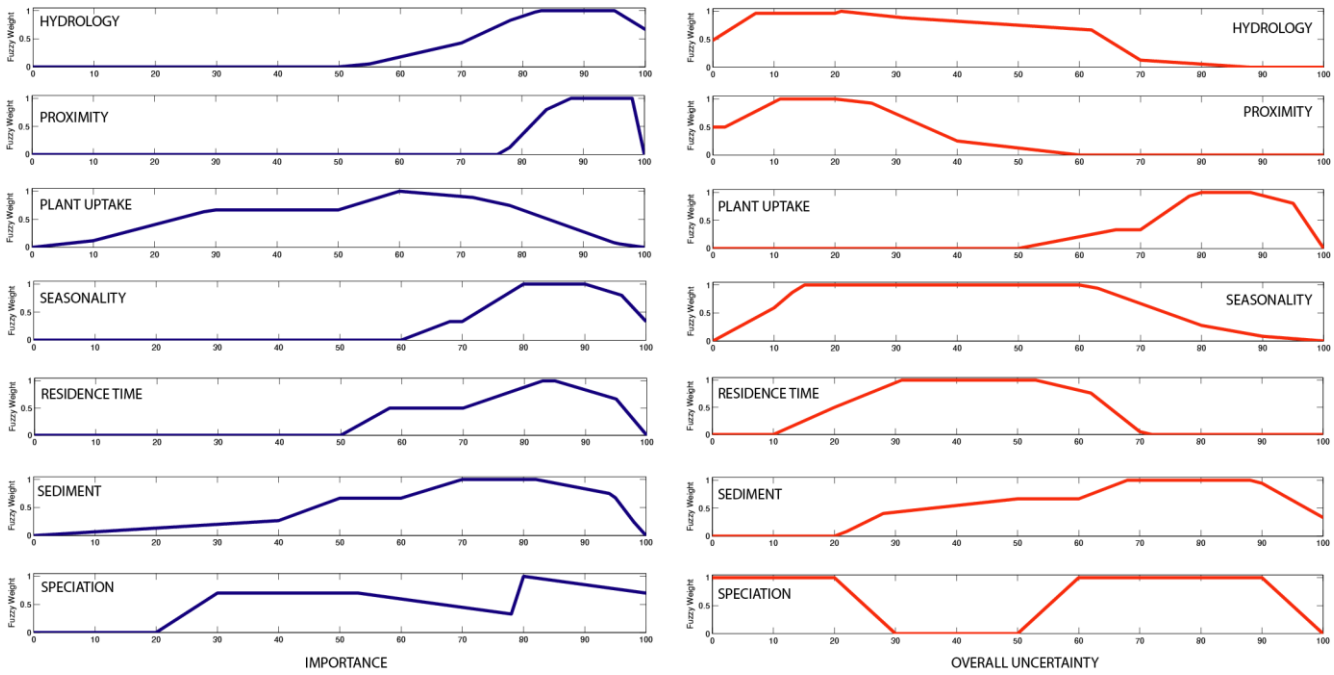


Figure 6.4. Stable Rivers: Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

6.5 Discussion of chosen components

6.5.1 Seasonality - of light, temperature, aquatic plant uptake and flow

The seasonality of light, temperature, biological uptake and flow are to some extent predictable and control the temporal distribution, and redistribution, of nutrients and biota. All three experts believed that seasonality should be taken into account for any P risk assessment as annual average measures can be misleading. As well as seasonal nutrient redistribution, the relative importance of external P sources is distinctly seasonal with a higher flux of diffuse P being delivered during periods of high discharge and a lower dilution of point source emissions during periods of low discharge (this is discussed further in Sections 3 and 11). In particular, for point source dominated rivers, the coincidence of low flow (reduced dilution) periods with the growing season increases their importance relative to the case where average fluxes are considered. Indeed, this importance weighting may be even higher owing to the bioavailability of P forms emitted by large point sources.

Seasonality is a master variable here as it affects all of the other components chosen by the experts. It is critical that seasonality be taken into account for P concentration in rivers, although annual average fluxes are still required where rivers feed lakes and for transitional and costal waters.

6.5.2 Proximity to point sources

A measurement of P concentration at any given point on a river is affected by all P inputs and P processing upstream of that point. In particular, if measuring downstream of a point source (e.g. a sewage treatment works), and assuming that measurement are made past the zone of

incomplete mixing[℘], there is often a rapid decline in P concentration (Jarvie *et al*., 2006) as the water interacts with sediments and biota (House and Dennison, 1997). It is important that proximity to source is taken into account to determine the significance (representativeness) of measured concentrations (e.g. GQA measurements) for a given WFD reach or catchment. Without knowledge of reach specific P retention/uptake it is required that some scale-dependent generic rules for P uptake and uptake length are developed that take account of the spatial and temporal variability observed (e.g. see Meyer and Likens, 1979; Marti and Sabater, 1996; Haggard *et al*., 2001; Marti *et al*., 2004; Haggard *et al*., 2005) and interactions in multiple reaches upstream of any given point (e.g. where the Mersey catchment reaches its estuary there are approximately 150 WwTW inputs upstream at distances of up to 80km).

At Tier 1 level, there are two scenarios where proximity to source needs to be taken into account: where GQA data are available and where P fluxes/concentrations are estimated. In the former case, where sufficient GQA measurements exist, the need to estimate retention/release of P may be overridden (as all upstream processes are implicitly included an any given P concentration measure) and the relevance of GQA measurements reverts to their representativeness for a given reach or catchment. In the latter case, uncertain estimates of point and diffuse P fluxes will be available from observations (see Sections 3, 4 and 5) and modelling (see Section 11); in this case P retention/release will need to be estimated from the results of smaller scale studies or from relationships derived at larger scales where data exists (e.g. see Behrendt and Opitz, 1999). This approach may be confounded by the large uncertainties associated with diffuse and point source P flux estimates and estimates of P flux at GQA locations.

6.5.3 Hydrology - conservative dilution of P signal and control of residence time

Hydrological discharge is crucial in the dilution of P entering the system (e.g. see House and Dennison, 1997), and in controlling water residence times that affect biogeochemical interactions, although this is a much simpler picture for point compared to diffuse sources. In general, our hydrometric data is very good as is our ability to predict flow patterns and magnitudes. Dilution can be reasonably well-estimated but residence time will be less certain because of the need to estimate reach geometry and the effects of backwaters and dead-zones in the calculation of *effective* residence times. The uncertainties associated with hydrological dilution and residence times are, however, relatively low compared to those affecting other important components. As well as these natural effects, there are anthropogenic impacts on the hydrological flow regime such as water abstractions, reservoirs and water transfers that impact flow regulation.

Hydrological variations also affect the cross sectional area (or reach volume) to bed area ratio and reach depositional regime that both contribute to the magnitude of sediment-water interaction.

6.5.4 Sediment - water interaction and P exchange capacity of sediments

There are relatively few studies of the role of instream processes that partially control the concentration of P fractions in rivers (House, 2003). Mass-balance studies can determine the overall effect of instream processes and determine periods of system loss or storage of P (e.g. House and Warwick, 1998; Bowes and House, 2001) but cannot fully identify the exact mechanisms in operation as there are a number of physical, physiochemical and biochemical mechanisms involved (Evans and Johnes, 2004; Evans *et al*., 2004). More detailed experiments which identify temporally variable chemical gradients and quantify the release and uptake of sediments are required (e.g. see House and Dennison, 1997). For example, Jarvie *et al*. (2006)

[℘] *EA GQA site location criteria requires that samples are taken after the zone of incomplete mixing. This zone is determined by modelling for larger WwTW (either using the Fisher equations or a US EPA model: CORMIX) or by sampling at a point at least 10 times river width downstream of the outfall for small WwTW.*

showed, using detailed measurements of sediment P equilibria and uptake into biofilms, the important role of bed sediments under certain conditions; they showed that immediately after P-stripping at a local WwTW bed sediment SRP-release was responsible for a 30 μg-P l^{-1} rise in reach SRP concentrations and it took approximately 6 months before a new equilibrium was reached under relatively low and stable flow conditions. The primary mechanisms for exchange of P considered in this study are:

- sorption-desorption reactions with bed (and suspended) sediments;
- Precipitation-dissolution reactions: e.g. the co-precipitation of phosphate with calcite in the water column and photic zone of bed sediments and the formation of vivianite in bed sediments of eutrophic systems;
- reach sediment fluxes: the erosion/depositional regime that controls sediment and particulate P fluxes and the resultant composition and structure of the bed sediment at any given time.

The type and magnitude of effect of these processes will depend upon factors such as: pH, redox potential, sediment geochemistry and organic matter content, biological activity, flow velocities and residence times, the relative concentration of P in the water column, of stream sediment (and interstitial waters) and suspended sediment.

6.5.5 Sorption-desorption reactions

Sorption-desorption reactions can be described in terms of an isotherm, which relates the amount of P adsorbed to solid phase substrates to given water column concentrations. Isotherms can take different forms such as the Freundlich and Langmuir nonlinear approximations or more simple partition coefficients (K_d) where linearity can be assumed for a known range of concentrations. As sorption reactions are driven by concentration gradients (which can be highly variable) between the stream water or sediment pore water and sorbing surfaces, quantifying the magnitude of fluxes is challenging. The challenges include quantifying the complex effects at the sediment-water boundary (e.g. redox conditions, bioturbation and hydrodynamic effects), prediction of the contact time between water and sediment (Meyer 1979; Bencala 1983; Munn and Meyer 1990) and estimating the sediment P status (for which Equilibrium Phosphorus Concentrations[Π] (EPC_0) are not routinely measured). If available, EPC_0 measurements may be useful in predicting the magnitude of the uptake or release of soluble reactive phosphorus, although there is evidence that their use becomes less reliable as the solution concentration approaches the EPC_0 (House and Dennison, 1997). One way to estimate K_d and EPC_0 at the national scale is by obtaining experimentally derived values from a significant number of reaches, within functional typologies (that take into account sediment physico-chemical properties such as particle size distribution: House and Dennison, 1997, 1998) such that estimates for other locations can be derived from national scale data on catchment characteristics (Expert 3r).

However, these values for sorption capacities are based upon laboratory studies where sediments are continuously and fully mixed with P-enriched water and where natural factors that effect the magnitude of contact (such as hydrostatic pressure, diffusion gradients and bioturbation) are not considered. Thus it is difficult to use these sorption data in isolation to predict real sediment sorption in dynamic stream systems (Lijklema,1993; Reddy *et al*., 1999), although they may be useful for a tier 1 screening approach (Expert 1hz). There are also implications in using lumped (in time and space) modelling representations of complex non-linear processes with averaged data (particularly where the non-linearities have been observed under laboratory conditions: e.g. the use of adsorption isotherms with concentration gradients estimated over monthly, seasonal or annual scales); problems arise as there is an inconsistency between the average of a heterogeneous nonlinear process and a similar nonlinear representation of the average heterogeneous behaviour (Neal, 1996). If these type of relationships are to be used successfully, a scale dependent approach will be required (Addiscott and Mirza, 1998b).

[Π] *Sediment Equilibrium Phosphorus Concentrations provide information upon whether sediments will take up or release SRP when placed in contact with a freshwater of known SRP concentration (House and Denison, 1998, 2000).*

However, given the problems discussed above, it may be more appropriate to derive even more simplistic rules about the likelihood of sediments being a source or a sink during different periods: e.g. periods of high or low flow during or outside of the growing season. Experts consulted considered the role of bed sediments to be potentially important (particularly as a sink during growing season low flows when stream P concentrations are likely to be higher) but that quantifying the magnitude of fluxes and their effect on instream P concentrations in time and space to be difficult; these are very complex processes and can only be approximated by deterministic computer models (see, for example, Wade *et al*., 2002a).

6.5.6 Precipitation reactions

Precipitation reactions such as the co-precipitation of phosphate with calcite can occur in the water column and the photic zone of biofilms (House *et al*.,1989) and bed sediments and is primarily (although not exclusively, House and Dennison, 1997) driven by photosynthetically induced changes in pH owing to the reduction of CO_2 or HCO_3 (Otsuki and Wetzel, 1972). This mechanism is hence particularly important during the growing season and can be considered to be a self cleansing mechanism (House, 2003). Co-precipitation may however be inhibited at high phosphate concentrations (Reddy, 1977), although this is not a simple relationship (Neal *et al*., 2002). In eutrophic systems, where pore water SRP concentrations are sufficiently high, the formation of vivianite can occur although the mechanism for its formation is uncertain (House, 2003). It is however associated with discharge from WwTW and fine sediments where there is sufficient organic matter for anoxic conditions to develop. Precipitation reactions can be relatively permanent, depending upon conditions; they can be readily reversible given significant changes in sediment solution pH, redox potential (Expert 2r; Diaz *et al*., 1994; Gold and Sims, 2000).

6.5.7 Sediment fluxes

The amount of P in the sediments at any time is not simply a product of the biological and chemical fluxes occurring in the reach. There can be potentially very significant reach sediment fluxes at hydrological event timescales and upwards (Bowes *et al*., 2003). The continual deposition and resuspension of bed material can therefore alter the ability of a reach to modify the instream P signal, P form (speciation) and timing of P flux through a river system. These fluxes are primarily controlled by hydrological regime, catchment characteristics and river geomorphology, so may be predictable to some degree.

6.5.8 Biological processing by aquatic plants

Plant uptake during the growing season can reduce water column P concentrations and has been observed to do this rapidly downstream of WWTW discharges (particularly during low flows: Tanner and Anderson, 1997; Jarvie *et al*., 2002b) but is, of course, one of the manifestations of eutrophic conditions. Water column P reduction is primarily a result of uptake by periphyton (Reddy *et al*., 1999) as most macrophytes utilise P from sediment porewaters. The uptake by macrophytes from pore waters can, nevertheless, be significant and create P concentration gradients. Over longer timescales these are two-way mechanisms as decomposition of plant matter will release P (House and Dennison, 1997), the fate of which depends upon timing; the P released may be quickly used by other organisms or flushed from the system during autumn storms (Reddy *et al*., 1999).

As for exchange with sediments, quantification of biological uptake (and decomposition) is challenging. From model simulations, the biological uptake of P has been shown to be secondary to point and diffuse P inputs and hydrological dilution (Expert 3r). Jarvie *et al*. (2006) found that uptake by biofilms accounted for only a very small percentage of in-stream P-uptake and that bed sediment exchanges had a more important control on baseflow SRP concentrations and loads. Biological processing of P may be more important in the uplands

where P is limiting or close to limiting. For example, House *et al*. (2001) suggest that uptake of P by macrophytes, from the bed and water column, is a relatively minor effect in nutrient-rich waters but is potentially of higher importance in nutrient-poor rivers.

Plants can also play a physical role by reducing water velocities, promoting sedimentation of particulate-P.

6.5.9 Phosphorus speciation

Speciation is important and in general we know that P released from point sources is more bioavailable than that from diffuse sources. There are some difficulties in our knowledge of the consistency between operational definitions (such as SRP) and true bioavailability which will vary depending upon species (that can only be determined by specific bioassays). Expert 1r suggested that bioavailable (sometimes SRP is used as a proxy but there are limitations to this measure: e.g. as part of the SRP measure is colloidal-P which differs in bioavailability and reactivity to phosphate-P) is the primary determinand and that a certain fraction of TP is made bioavailable by instream processes (as all fractions are potentially bioavailable) and some bioavailable P is made unavailable (as discussed below) but that these transformations are difficult to quantify. There are particular problems with organic P speciation (Expert 2r). The importance of bioavailable P is system-dependent: i.e. it will differ between P-rich or P-poor systems. P-poor systems may be more sensitive to bioavailable P inputs whereas in lowland systems, which already have high levels of P, it may be less important. However, for P-poor upland systems there are very little data and the P concentrations are often close to or below the limit of detection (e.g. most of the data from the Plynlimon catchments in Wales: Expert 2r); thus in P-poor situations, much of our knowledge is not based upon strong evidence. If only measuring SRP, we may be missing the organic-associated P component: this is not well studied and a priority for future research.

6.6 A simplified conceptual model of instream phosphorus processing

At WFD catchment/reach scales and at a Tier 1 level it is not practical to predict the interacting effect of each of the components described above and particularly their temporal dynamics. It may however be possible to derive some simple and generalised rules for different river systems that describe the likely level of P retention/release and P concentration on a seasonal basis, using nationally available data to describe the components of the simplified conceptual model presented in Figure 6.5. As discussed above, reach mass flux studies can determine overall behaviour but cannot attribute their relative magnitude of effect. Thus, initially any model based upon these components (at the scales of interest here) will be based upon expert opinion until sufficient observations become available from studies at appropriate scales. At higher tier level however, where sufficient data exists for model evaluation, it may be possible to apply more complex models successfully, although the data required can be extensive. For example, the application of the Kennet model to the upper reaches of that river demonstrated that the following data are required: flow, (total and/or soluble reactive) phosphorus concentrations, solar radiation and air temperature (Wade *et al*., 2001). Other data, namely macrophyte and epiphyte biomass, sorption co-efficient and equilibrium phosphorus concentrations and streamwater boron concentrations are also useful (Expert 3r).

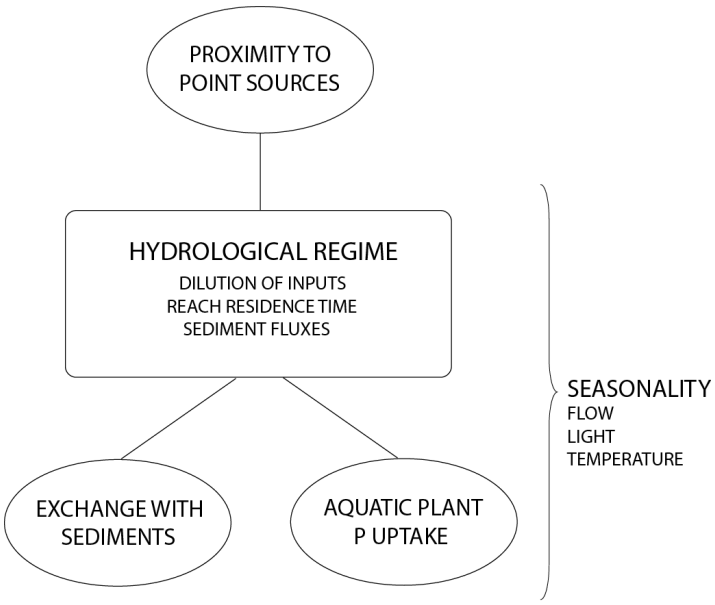


Figure 6.5 – Post-interview phase simplified conceptual model of instream P processes.

6.7 Research Priorities

Studies that may help determine the relative magnitude of the effect of individual instream processes for different rivers or reaches (perhaps expressed as typologies) or determine larger scale behaviour include:

- Accurate quantification of point source inputs (see also Sections 4 and 5).
- Determination how important the bed sediments are at larger scales (e.g. WFD reach scales). Once this has been done for a sufficient number of reaches, it may be possible compare them with catchment characteristics to identify useful relationships.
- An expansion of sediment exchange research to more upland systems (Expert 3).
- Understanding the role of organic P in the uplands.
- Understanding the role of other pollutants and their effect on ecology (e.g. herbicides and pesticides: see also Sections 9 and 10).
- Understanding the role of reservoirs, lakes, river transfers and canals which maintain low flows, and input of nutrients and algal innocula at times when ecology is most at risk.
- Longer-term studies (over many years) are needed to resolve what is really happening: as there is also the question of the timescale of observable response.
- Better use of the knowledge that already exists within the EA. This could be a desk study aimed at EA local knowledge before other studies are planned.
- More detailed modelling/data collection in a few catchments/reaches.

Research priorities that are implicit from the elicited fuzzy distributions (Figures 6.3 and 6.4 above) are those components which were rated as highly important and highly uncertain. This is expressed simplistically in Figures 6.6 by a weighting calculated from the product of the ratings (the centroid of each fuzzy distribution) for each chosen component. Figure 6.6 shows that there are no distinct research priorities that stand out above the other components, although plant uptake and exchange with sediment have the two highest weightings.

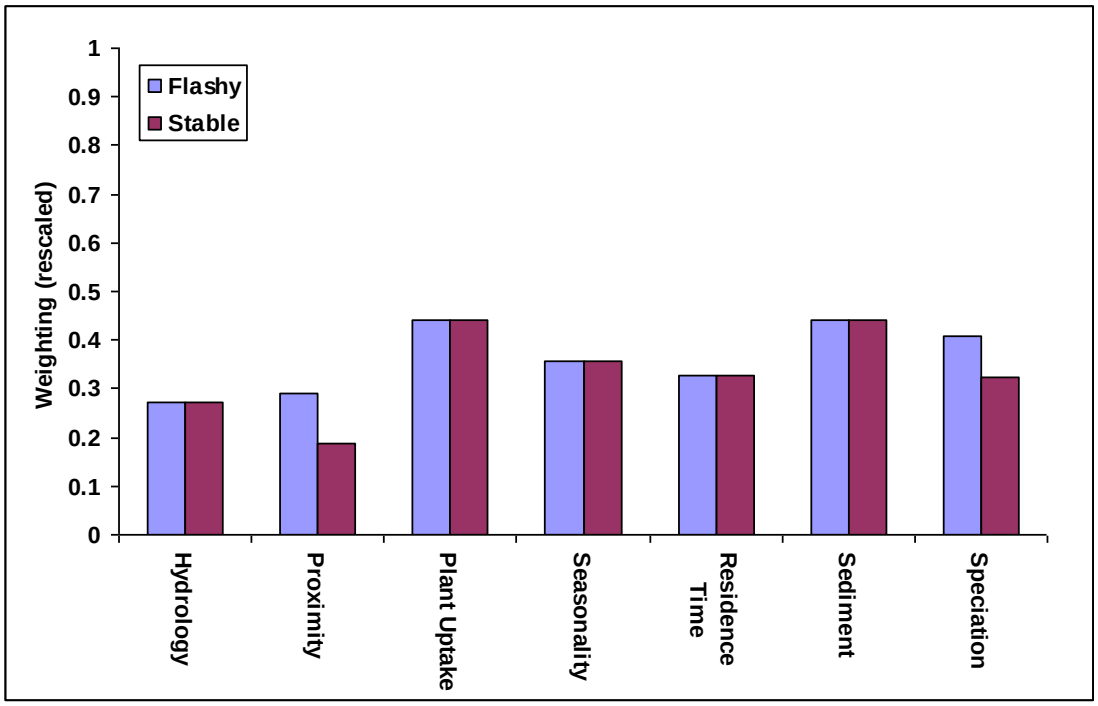


Figure 6.6 – Ratings for Instream Processes model components with regard to research priority estimated using the product of the centroid of the importance and overall uncertainty fuzzy distributions.

6.8 Conclusions

Instream processing of phosphorus is of particular importance for the Environment Agency's risk assessment as many of the models currently used to predict diffuse P fluxes (particularly at Tier 1 level) make estimates of P delivered to the stream and do not explicitly take into account instream processes. At higher tier levels more complex models such as INCA-P have been used which explicitly account for instream processing, but which have high data requirements. Additionally, instream P measurements made as part of the EA's GQA monitoring strategy need to be put into context within WFD catchments or reaches with respect to their proximity to upstream sources.

Phosphorus fluxes to rivers are likely to undergo numerous physical, chemical and biological transformations during downstream transport. These interacting processes act at many timescales from effectively instantaneous chemical reactions to the gradual and stepwise transfer of P and sediment during hydrological events. The significance of these processes, with regard to the risk from P of degraded ecological status, is primarily associated with the net retention, or release, of bioavailable P concentrations during the growing season: although the resuspension of P-rich sediments during the autumn and winter may be important for lakes and transitional waters. Thus the importance of the temporal dynamics of P processing is system-specific as all P fluxes are potentially bioavailable unless lost to a true sink. There are hence two primary considerations for instream processing: The attenuation/release of growing season river P concentrations and net effect of these processes over annual or longer timescales.

At WFD scales, the identification of retention/release processes, their relative magnitudes and their temporal variation is highly uncertain as resolution of individual effects is difficult and as with terrestrial P losses our knowledge of scale-specific representations or methods of upscaling from detailed studies is limited. What is needed are studies which allow the development of scale-dependent rules or scaling theories. This may be made possible by some of the suggestions for future research.

The usefulness of the simplified model presented here (over and above an expert system) requires that a significant amount of reach/catchment scale data be collected with which to

develop rules that define the likely magnitude of effect of each model component for a given system, or system typology.

7 Hyporheic Zone Processing of Phosphorus

7.1 Summary

The functional significance of the hyporheic zone (HZ) with respect to EA P risk analyses primarily relates to its role in the modification of surface water P concentrations and groundwater P concentrations entering surface waters: i.e. does the HZ mitigate the impacts of polluted GW discharge or contribute to instream water quality improvement? In terms of ecological impacts, it is desirable to be able to estimate whether the HZ of a given WFD river reach or catchment will behave as a P source or a P sink during ecologically sensitive periods (i.e. the growing season). Similarly to the broader discussion of instream processing in Section 6, high P concentrations during growing season, which normally coincides with baseflow periods when the dilution of point source P emissions is normally at its lowest, are of primary concern: i.e. is there a net retention, or release, of P in the HZ during these periods which attenuates P concentrations (e.g. see Garay *et al*., 2001). Estimation of the likely magnitude of P retention/release, or instream P concentration attenuation would be advantageous for risk analyses, particularly where there are no GQA monitoring locations in the WFD catchment of interest. The possibility of improving the quality of both surface and groundwaters in this way makes the HZ potentially very important (Hancock *et al*., 2005).

The primary factors that determine how the HZ is likely to function (as identified by the experts consulted) are those that control the water residence time in alluvial sediments, the sediment exchange potential and physically- and biogeochemically-driven controls on chemical gradients within the sediment. Identification of these factors as components of a simplified scale-dependent conceptual model of HZ function was made during the interviews, after consideration of processes observed at a number of scales (e.g. those occurring at microscale on particle surfaces and at sub-reach scale such as the flow through *pool-riffle* sequences). The simplified conceptual model developed is presented schematically in Figure 7.1 and broadly relates to 4 main areas which may vary significantly on a seasonal basis: hydraulic gradient and hydraulic conductivity controls on water exchanges; P concentration gradients; sorption and precipitation/dissolution of P in HZ sediments and biological uptake.

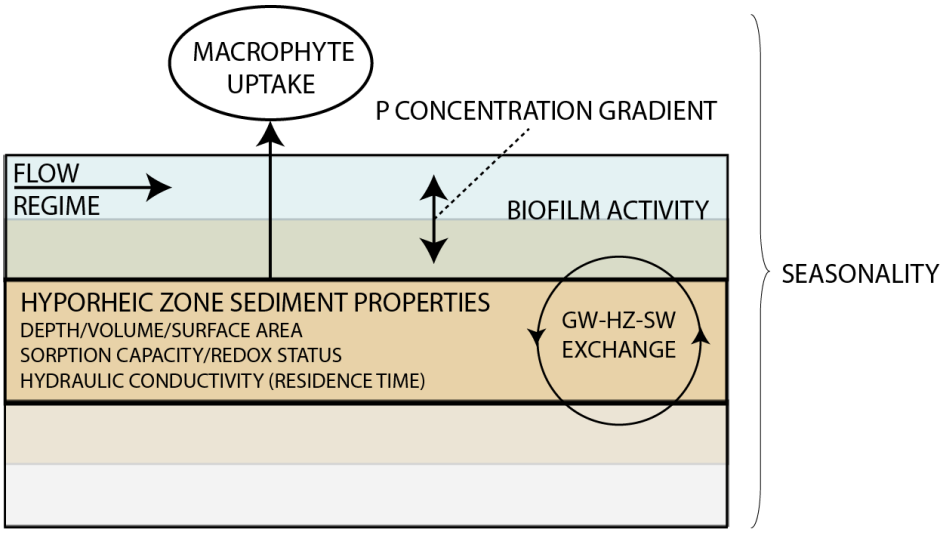


Figure 7.1 – A simplified conceptual model of hyporheic zone function at WFD reach and catchment scales.

The predictability of these primary model components at WFD scales will be highly uncertain and will require us to determine the net effect of dynamic and heterogeneous HZ functioning for periods of interest: e.g. during the growing season and as an annual average.

Recommendations for data collection and further research

- An increase in reach-scale studies to determine the net functioning of the HZ over various timescales.
- Good source apportionment studies are required to quantify fluxes through the HZ. Tracer studies may be the best option for determination of source.
- Better knowledge of redox conditions within sediments is critical, but may only be practical for higher tiered approaches.
- Better fractionation of P forms in samples.

Investigation of the usefulness of riverbed visual surveys as proxies for HZ conditions: this may include some elements from the River Habitat Survey (Raven *et al.*, 1997).

7.2 Introduction

Many different definitions of the hyporheic zone (HZ) exist (e.g. see the reviews of Brunke and Gonser, 1997 and Smith, 2005) and vary between disciplines (e.g. ecologists sometimes describe stream bed sediments as HZs, where surface waters (SW) do not interact with any 'true' deeper groundwater (GW) component: Expert 2hz. For the purposes of this project the HZ is defined as:

> *The water-saturated transitional zone between streams and groundwater in which there is bi-directional exchange and mixing of water between channel and subsurface.*

Within this definition, groundwater is a generic term for water that moves through deeper aquifers, alluvial aquifers, riparian and parafluvial zones (e.g. see Figures 7.2 and 7.3).

The HZ is transitional zone between the surface stream and groundwater and can be important geochemically and ecologically. Exchanges of oxygenated water, nutrients, and organic matter occur in response to variations in discharge, river geomorphology and bed sediment hydraulic conductivity. This gives rise to temporally dynamic gradients at all scales. Much of the literature research is concerned with relatively small scale HZ flow pathways driven by surface water advection, such as the flow through *pool-riffle sequences* (Figure 7.3b) or through deposits underlying stream meanders (Figure 7.3c), and the gradients in dissolved O_2, organic matter, microbial activity and nutrients that it promotes. The generalised conceptual model describes initial adsorption of P in aerobic sediments on the upstream side of the HZ structure and a subsequent release of P along a redox-driven gradient towards (but probably not fully) anaerobic at the lee side of the HZ structure. As these concepts are upscaled (in time and space) it is the net fluxes of P within these structures that become important. Larger scale conceptual models exist (such as the hyporheic corridor concept that describes HZ functioning at sediment, reach, Garay *et al*. (2001), and catchment scales, Stanford and Ward (1993)) but may still fail to account for larger scale hydrogeological flow regimes and geochemical processes (Smith, 2005). Whichever scale is of interest, the functional significance of the HZ is primarily driven by the degree of interaction between surface water bodies and groundwaters and sediment physical and geochemical properties.

At the scale of interest for this study (i.e. WFD catchment reaches), the functioning of the HZ can be conceptualised in terms of the net effect of the many small scale processes occurring. These processes are spatially and temporally heterogeneous and we currently do not

understand well their effect at larger scales: how we deal with this heterogeneity and uncertainty is key (Expert 2hz). It may, however, be possible to predict how heterogeneity varies regionally (e.g. there may be simplistic upland and lowland typologies where the amount and size distribution of bed sediment and degree of water exchange are significantly different) and seasonally (e.g. under different flow regimes). The experts agreed that it was meaningful to try and quantify the effect of the HZ at WFD scales, providing it is done in an adaptive and iterative manner. Indeed, significant progress has been made in estimating the nature of HZs across England and Wales from nationally available data (see Booker *et al*., 2006). What is ultimately required for WFD risk analysis purposes is an estimate of the potential (and magnitude of effect) for the HZ to behave as a source, attenuator[¥] or sink for P over given timescales (see Figure 7.4 for a schematic representation).

7.3 Assumptions, definitions and caveats

During the interview phase where experts identified important model components, importance is defined with respect to: the likely magnitude of effect of a given component in modifying the stream P signal at the scale of interest.

HZ typologies per se were not chosen. The experts did however consider HZ functioning during:

1 - '*annual average'* conditions.
2 - '*growing season average'* conditions.

Only one data scenario was considered during the quantitative phase of the interview: that available from national databases.

7.4 Expert weightings for chosen model components

For model components chosen by the experts, where necessary, there is a definition in the relevant sections that follow. The experts chose only to distinguish typologies that related to net HZ function under *annual average* and *growing season average* conditions. Composite (a combination of the 3 fuzzy distributions elicited from the experts: as detailed in Section 2) *importance* and *overall uncertainty* fuzzy weightings for the chosen components are presented in the left- and right-hand pane respectively of Figure 7.5 for annual average and Figure 7.6 for growing season average. There was a high degree of consensus between the experts for both importance and uncertainty ratings with the main differences being the breadth of distributions elicited. Comparison of the importance and uncertainty weightings between typologies highlights that, although the experts chose to distinguish between annual average and growing season functionality, when combined, there were generally only small differences between the weightings given; the importance of biofilm activity was however given a significantly higher weighting during the growing season and the components GW/SW exchange and sediment depth/volume were weighted as marginally more important during the growing season. Consistent with the instream processes chapter, the sources of uncertainty for the HZ components chosen were derived more from spatial and temporal heterogeneities, rather than scientific understanding per se. These heterogeneities coupled with low data availability make our scientific understanding difficult to apply.

[¥] *Used here in strict signal-processing terms to mean the reduction in amplitude of a signal rather than the more general definition of: a reduction or weakening of a given property.*

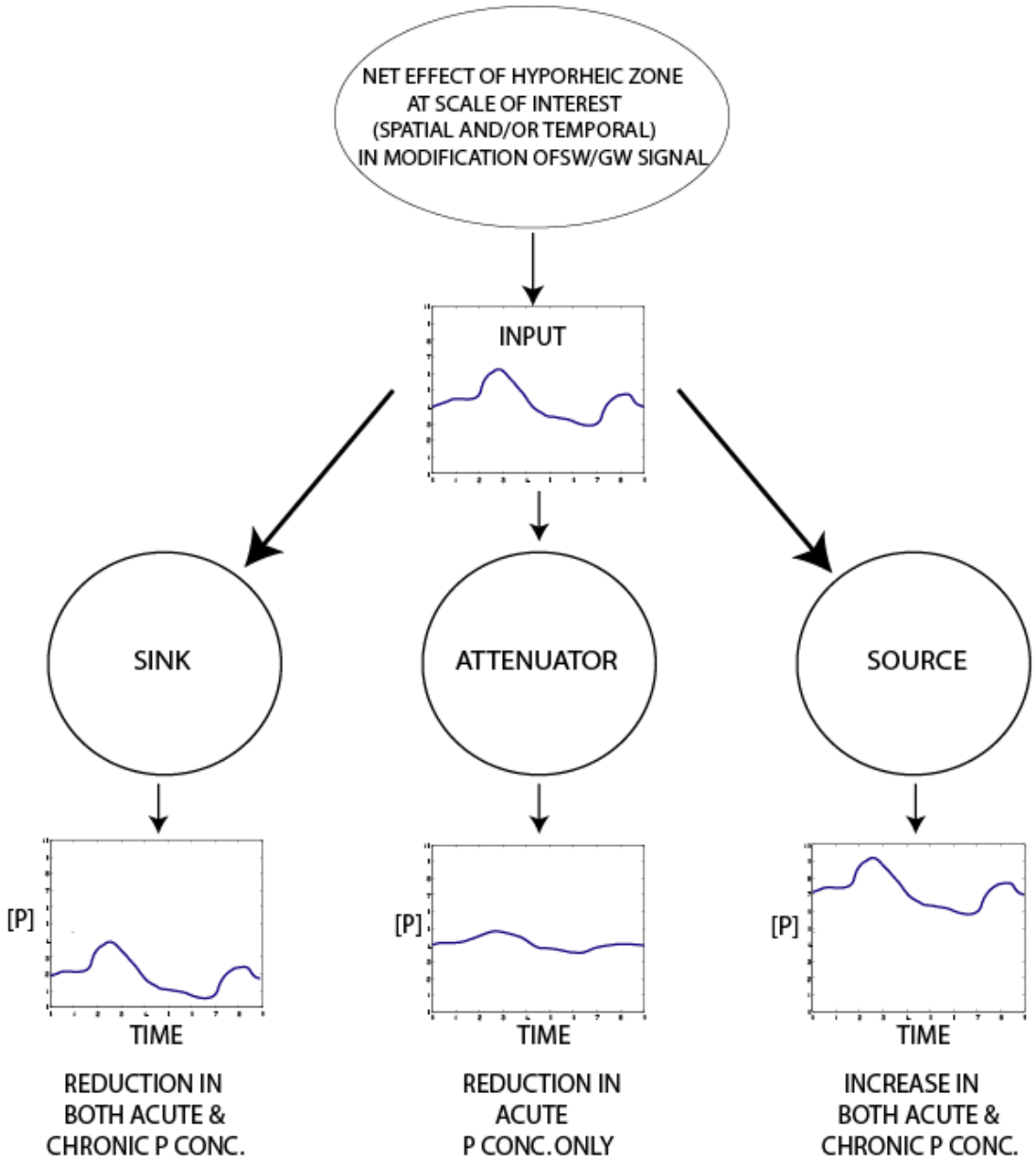


Figure 7.2 - A schematic representation of possible hyporheic zone modifications of the stream or groundwater P signal (timescale independent). The signal shows generalised dynamics that may relate to any temporal scale and the HZ modification relates to its net effect at any spatial scale. The key point of the diagram is the question: what is the functional significance of the HZ at the scale of interest (e.g. is acting as a source (increasing P concentrations), sink (decreasing P concentrations) or signal attenuator (reducing the amplitude of the P signal).

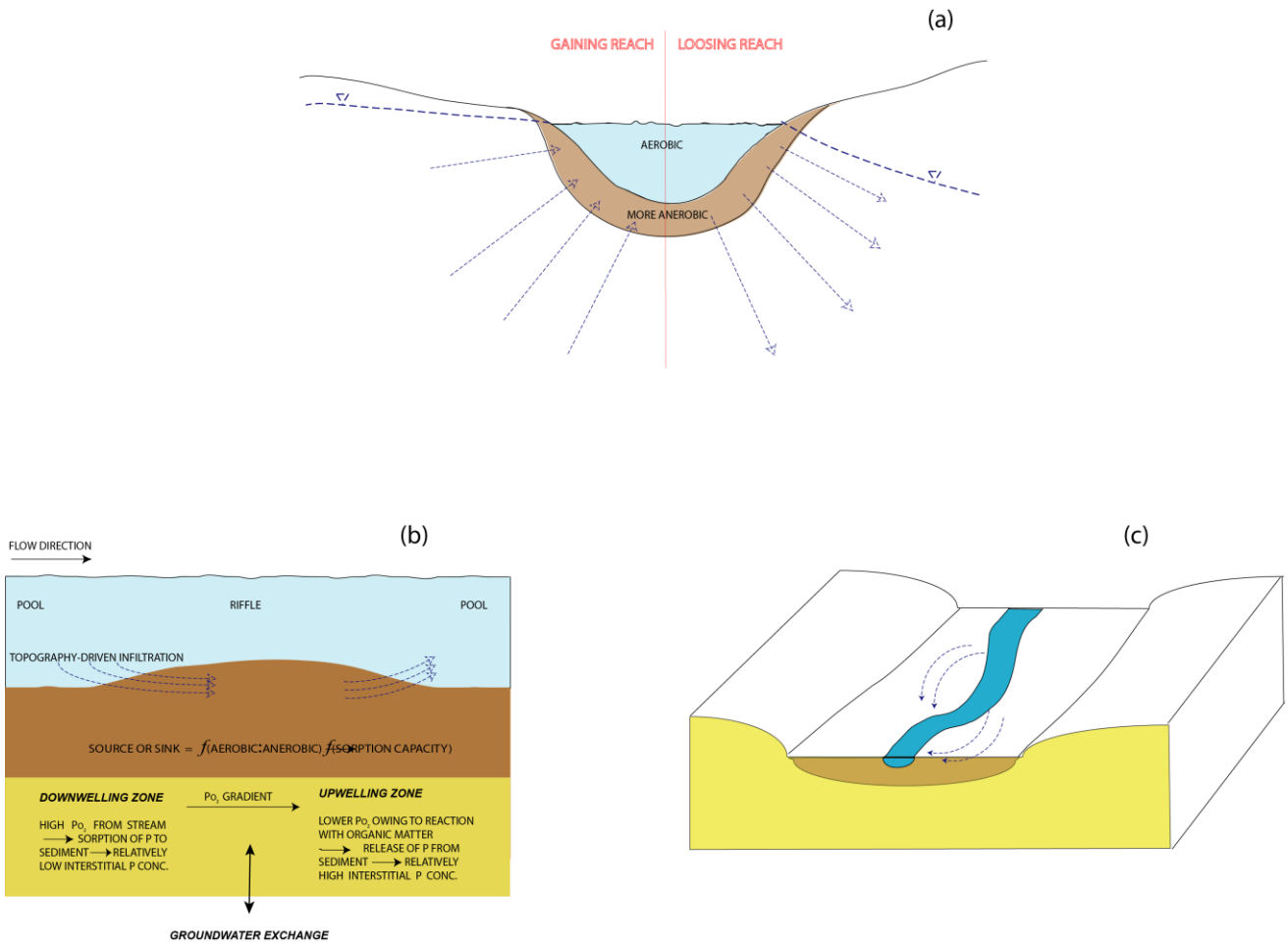


Figure 7.3 - Simplified diagrams of hyporheic zone exchanges and flow pathways: (a) A cross section of a stream showing both gain and loss of groundwater; (b) The topographically-driven flow through a pool-riffle sequence resulting in upwelling and down welling zones and biogeochemical gradients along the flow path (c) Flow through stream meander sediment bars resulting in similar flow path properties shown in (b).

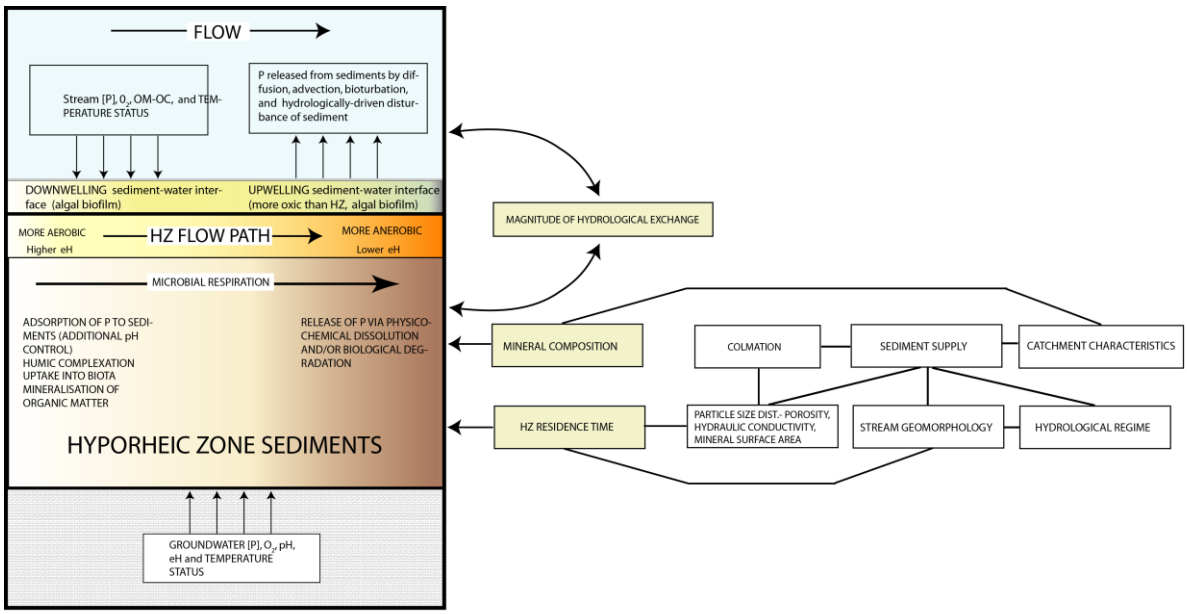


Figure 7.4 - Generalised schematic diagram of HZ functioning where SW/GW exchanges represent generic upwelling and downwelling zones (vertical or horizontal) resulting from flow through pool-riffle sequences, gravel bars, meanders etc. and the GW component represents that in parafluvial zones, alluvial aquifers and 'true' aquifers.

The importance weightings indicate the likely magnitude of effect of given components on instream P concentrations. In the sections that follow, the underpinning scientific basis of the primary components will be briefly reviewed and assessed for suitability for incorporation into national scale risk analyses. Suitability will be assessed based upon whether or not it is feasible to upscale observations made to WFD reach/catchment scale given the information currently available (i.e. a Tier 1 risk assessment).

7.5 Discussion of chosen components

The effect of **seasonality** is implicit in the distinction between HZ function under annual average and a growing season average conditions and was expressed by the difference in fuzzy weightings given to the model components: e.g. see the weightings for biofilm activity, GW/SW exchange and sediment depth/volume. Seasonality was also highlighted during the qualitative phase of the interviews, during general discussion whilst considering components. As for many other instream processes (Section 6), one crucial role of the HZ may be the potential to attenuate (see Figure 7.2) growing season P concentrations (particularly during low flows where point source emissions are subject to lower dilution: Garay *et al*., 2001). It is possible that the HZ may act as a P sink during these crucial periods and a source when flows are higher and where they are associated with lower P concentrations (Expert 2hz).

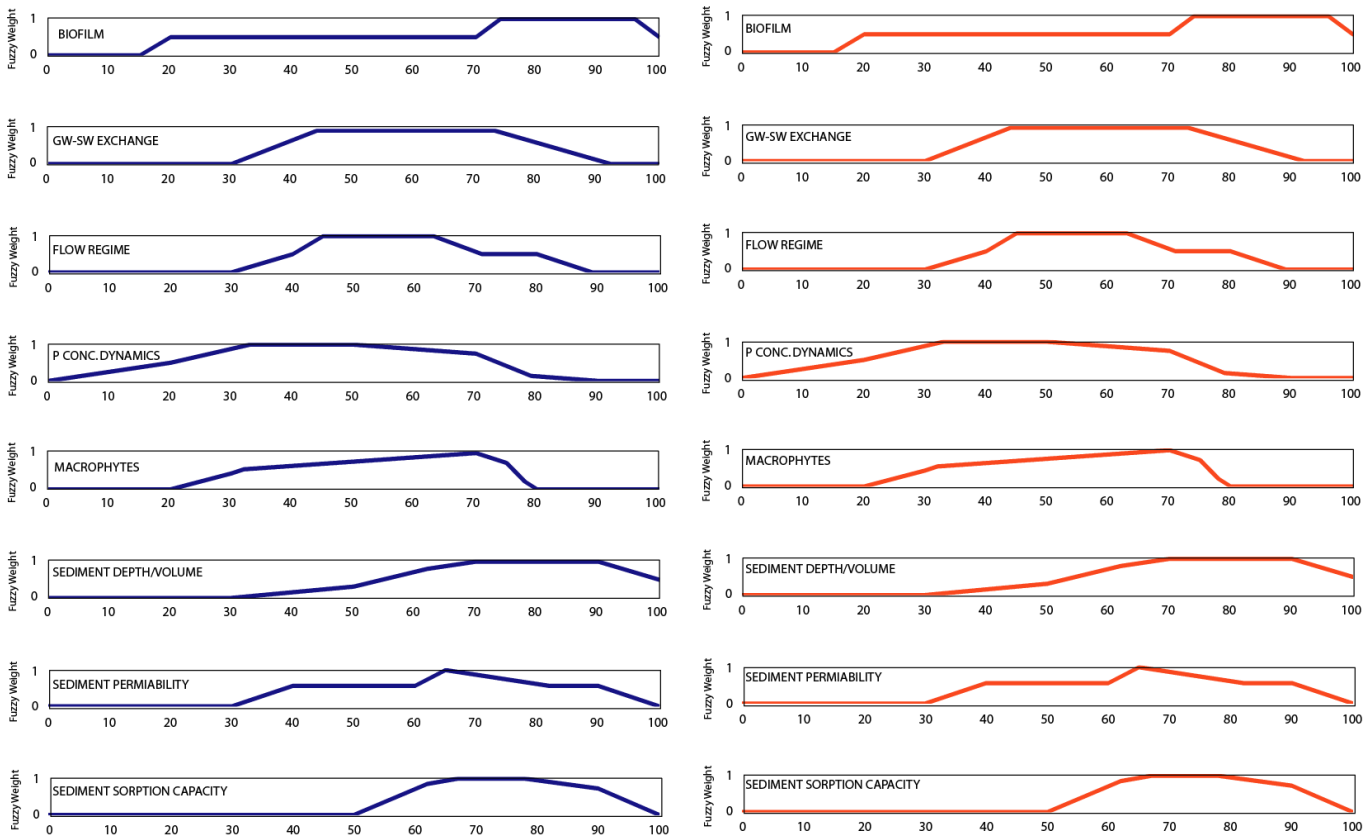


Figure 7.5 – Hyporheic zone function (annual average) Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

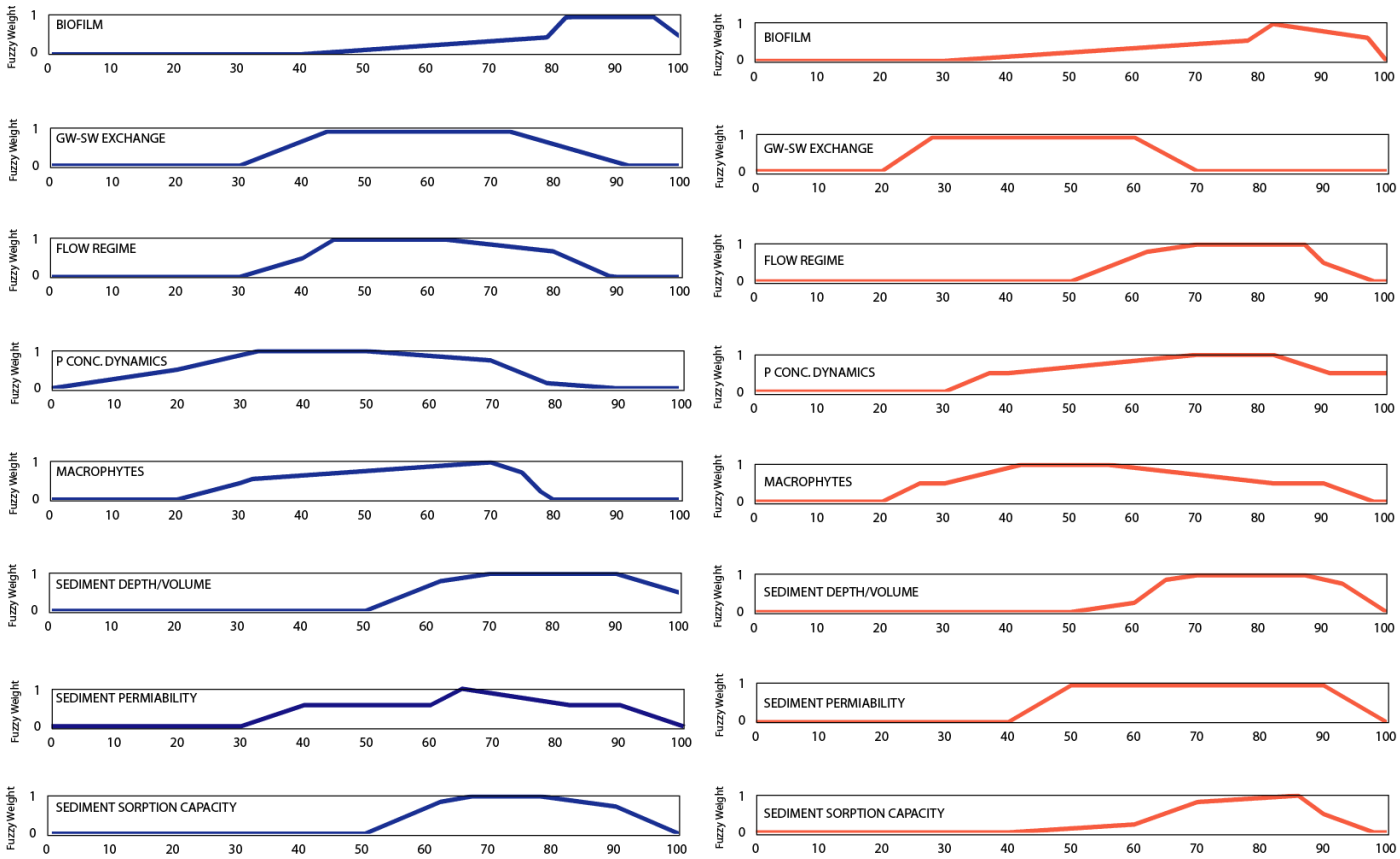


Figure 7.6 – Hyporheic zone function (growing season average) Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

7.5.1 Exchange between groundwater and surface water body

The degree of exchange between GW and SW controls chemical gradients and promotes physical mixing and is driven by many factors such as hydraulic gradients, sediment characteristics and geomorphological features (Dahm *et al*., 1998; Garay *et al*., 2001; Malard *et al*., 2002). At small scales, the direct measurement of HZ water flux is difficult and the scale of heterogeneities observed in the hyporheic zone often coincides with the scale of measurement, resulting in significant uncertainty (Smith 2005). Larger scale techniques such as tracer tests and reach-scale mass-balance observations can provide a more integrated measure of likely flow for a given period, but may not elucidate average behaviour over seasonal or annual scales. It may be possible to use more novel techniques such as infra-red imaging from aircraft (using temperature as a tracer: Faux, *et al*., 2001, Environment Agency, 2007c) to get a qualitative estimate of HZ fluxes at different times (and flow conditions) throughout the year. At the national scale the degree of exchange may be estimated by simplistic parameters such as BFI, although BFI may only help predict the degree of connection with a GW system rather than exchange per se. Without direct measurements it may be possible to estimate exchange from general knowledge of the river flow regime, river stage height, localised riparian head, sediment thickness and valley gradient: rapid river appraisal methodologies that geomorphologists use may be helpful in achieving this (Expert 2hz).

In simplistic terms, the more bed sediment (depth/volume) the higher the likelihood of increased water residence times and hence contact time for biogeochemical reactions to take place. Residence times will also be determined by the **saturated hydraulic conductivity** of the sediment which will in turn be determined by factors such as: sediment particle size distribution;

the presence of biofilms (Findlay and Sobczak, 2000) and colmation (the deposition of fine-grained sediments forming a low hydraulic conductivity layer). Colmation may be the most widespread process, with a regionally significant impact in the UK (Smith, 2005).

At larger scales, and ignoring localised geomorphological controls, sediment thickness may be approximated from catchment properties and stream power (a function of discharge and channel slope). For example, Booker *et al*. (2006) estimated a relative depth of fine sediment for surface waterbodies in England and Wales using a balance between stream power and the catchment sediment supply. Similarly, Booker *et al.* (2006) have estimated sediment hydraulic conductivity by combining the hydraulic conductivity of drift deposits with estimated catchment fine sediment supply.

In sediments with low hydraulic conductivities, chemically reducing conditions are more likely as the ingress of O_2 is lessened and longer water residence times allows microbial respiration to deplete O_2 supplies further. Microbial respiration is primarily controlled by sediment organic carbon which is generally significantly higher in the HZ compared to groundwater bodies (organic carbon content may be estimated from the GBASE BGS database or derived from catchment properties: Expert 1hz). The sediment redox conditions are critical and determine whether or not P sorbs or desorbs onto sediment surfaces and also partially controls precipitation reactions (e.g. see Surridge *et al*., 2005 and Section 6). HZ redox conditions may be approximated, where measurements exist, using a combination of GW and SW O_2 concentrations weighted by the BFI (Booker *et al*., 2006). In GW dominated catchments, GW geochemistry can also have a profound influence on P behaviour by controlling the pH: general base status may be a good proxy for generic types of HZ (Expert 2hz).

7.5.2 Sediment sorption capacity and phosphorus precipitation

Sediment sorption capacity and phosphorus precipitation are a function of sediment and pore water physico-chemical properties, pH and redox potential and is discussed in Section 6 (see also Smith and Lerner, 2007).

7.5.3 Biofilm mass productivity

The capacity for P uptake in biofilms was thought to be important, although the experts were not biologists and thought that it may be difficult to estimate at larger scales (see also Section 9). Garay *et al*. (2001) showed, however, that a combination of periphytic biofilm and HZ sediments retained approximately 22% of instream TP in a study on a large reach in France and that the biofilm compartment was a temporary sink during critical periods.

7.5.4 Macrophyte uptake from hyporheic zone sediments

Macrophytes with roots in the substrate will take up P from porewaters and will be seasonally variable (see the discussion in Section 9).

7.5.5 Stream phosphorus concentration dynamics

The chemical gradients within the HZ will be partly driven by variations in stream P concentrations. The variability can be significant and can vary both with and independently of stream flow depending upon circumstances (e.g. the relationships between P and flow exhibited by point source or diffuse source dominated reaches).

7.6 A Simplified Conceptual Model of Instream HZ processing

The primary factors that determine how the HZ is likely to function (as identified by the experts consulted) are those that control the water residence time in alluvial sediments, the sediment exchange potential and physical- and biogeochemical-driven controls on chemical gradients within the sediment. Identification of these factors as components of a simplified WFD scale conceptual model of the HZ was made during the interviews and after consideration of processes observed at smaller scales (e.g. those occurring at microscale on particle surfaces and at sub-reach scale such as the flow through *pool-riffle* sequences). The simplified model is presented schematically in Figure 7.7.

Identification of these components for WFD reaches/catchments nationally depends upon the available data and has been considered by Booker *et al*. (2006). Table 7.1 shows the most likely data sources for identifying the likely significance of model components which may be combined to give an overall attenuation potential. As for other subject areas, meaningful combination of weightings for model components (over and above a purely expert system) depends upon adequate scale-dependent data that can resolves differences in HZ function between different HZs or functional classes of HZ.

Table 7.1 - data sources for simplified HZ model components

Model Component	Data source	Available seasonally
GW/SW exchange	BFI	N/A
Sediment depth/volume	Catchment properties and stream power**	N/A
Sediment Hydraulic conductivity	Drift/catchment properties ** EA Sediment Risk Analysis	N/A
Sediment sorption capacity	Sediment Geochemistry**	N/A
Flow regime	Gauged Flow statistics (Q5, Q50, Q95, etc.); LOWFLOWS 2000; BFI	YES
Stream P conc. dynamics	GQA P-Q relationships (see Section 9 and Fig. 9.5) and model approximations	YES
Macrophyte uptake	Limited (see Section 9)	Approximation
Biofilm P uptake	Limited (see Section 9)	Approximation

** From Booker *et al*. (2007).

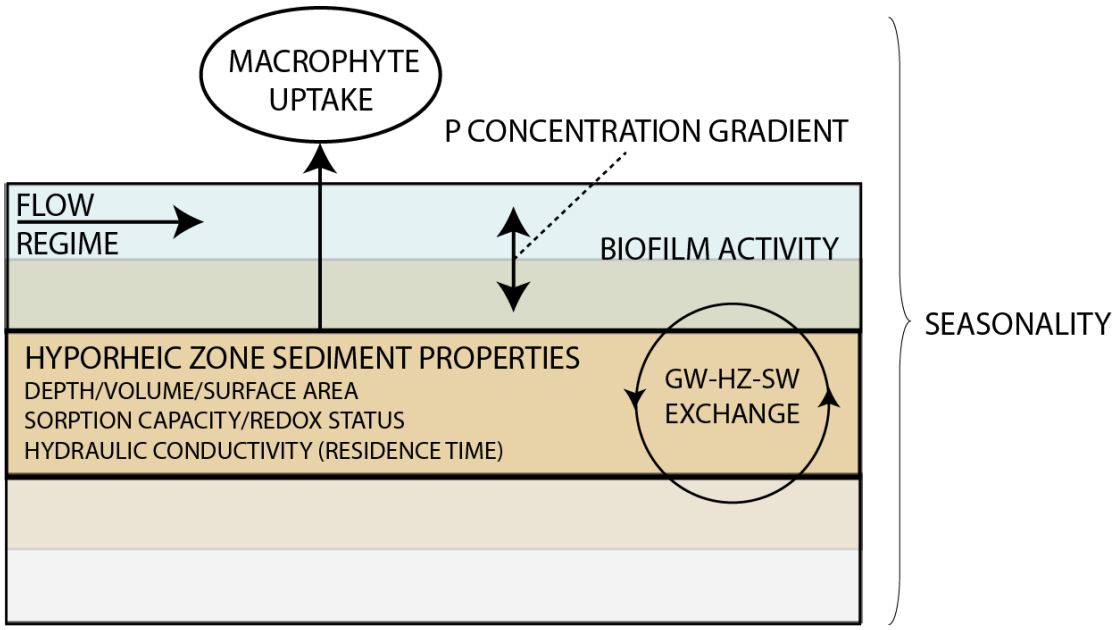


Figure 7.7 – A simplified conceptual model of hyporheic zone function at WFD reach and catchment scales.

7.7 Research Priorities

Explicit research priorities identified by the experts include (see also the recommendations of Smith *et al*., 2008):

- An increase in reach-scale studies to determine the net functioning of the HZ over various timescales.
- Good source apportionment studies are required to quantify fluxes through the HZ. Tracer studies may be the best option for determination of source.
- Better knowledge of redox conditions within sediments is critical, but may not be practical at Tier 1 level.
- Better speciation of P samples.
- Investigation of the usefulness of HZ visual surveys: this may include some elements from the River Habitat Survey.

Research priorities that are implicit from the elicited fuzzy distributions (Figures 7.5 and 7.6 above) are those components which were rated as highly important and highly uncertain. This is expressed simplistically in Figure 7.8 by a weighting calculated from the product of the ratings (the centroid of each fuzzy distribution) for each chosen component. These figures show that identification of the likely magnitude of interaction between ground and surface waters in HZ sediments have been highlighted as has work on identifying the role of biofilms.

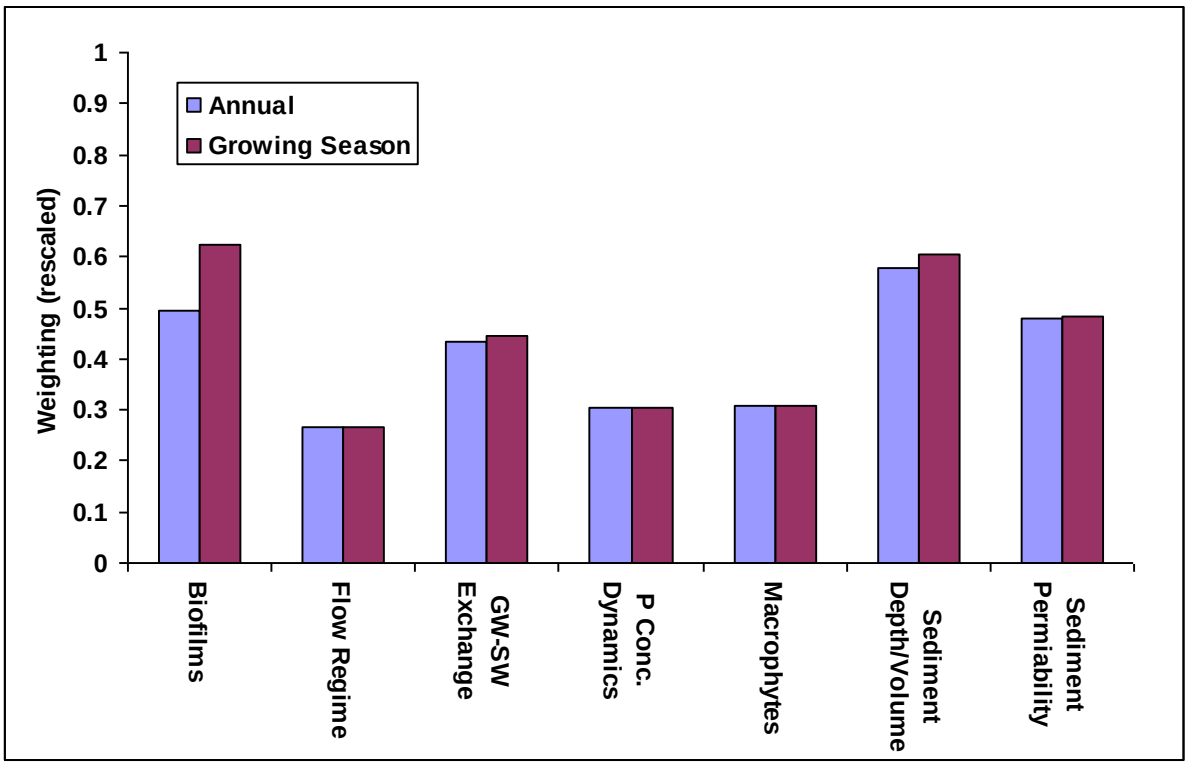


Figure 7.8 – Ratings for Hyporheic Zone model components with regard to research priority estimated using the product of the centriod of the importance and overall uncertainty fuzzy distributions.

7.8 Conclusions

The functional significance of the hyporheic zone (HZ) with respect to EA P risk analyses primarily relates to its role in the modification of surface and groundwater P signals. In terms of ecological impacts, it is desirable to be able to estimate whether the HZ of a given river reach or WFD catchment is behaving as a net P source or P sink over the duration of the growing season. Similarly to the broader discussion on instream processing (Section 6), high P concentrations during growing season low flow periods, when the dilution of point source P emissions is normally at its lowest, are of primary concern: i.e. a seasonal P retention/release. Estimation of the likely magnitude of P concentration attenuation (of both surface and groundwaters) would be advantageous for risk analyses, particularly where there are no GQA monitoring locations in the WFD catchment of interest. Where GQA sampling is in place, and these data are the principal measures of stream P status, the net effect of all instream processes occurring upstream of a given sampling point are implicitly taken into account. Similarly to the case for general instream processing (Section 6), and where GQA monitoring data exists for a WFD catchment, the primary concern relates to the representativeness of the GQA data (i.e. the location of the GQA sampling point with respect to the WFD catchment boundaries and its proximity to point source P emissions.

8 Phosphorus in Groundwaters and Contributions to Surface Water Receptors

8.1 Summary

As there are no drinking water standards for P and there are no measures of direct ecological effects in groundwater (GW), surface waterbodies are the ultimate receptors for groundwater P (EA, 2007). Hence, in this section, groundwater bodies are relevant as part of the risk pathway to surface water receptors. Principal sources of P in groundwaters (over and above naturally occurring background levels) are derived from diffuse agricultural sources, point agricultural sources (e.g. leakage from slurry tanks: Gooddy *et al*., 2002) and quasi-point sources: septic tanks and sewer leakage. Both point and diffuse derived P also enter groundwaters during recharge from rivers. The flux of P to groundwater is controlled by the degree of attenuation in soil and drift deposits. The subsequent flux of groundwater P into surface waters is primarily dependent upon further attenuation within the aquifer, the relative significance of the aquifer, the degree of connectivity between the groundwater and surface water bodies and modifications to P fluxes within wetlands and hyporheic zones. The degree of uncertainty associated with predictions of the likely magnitude of effect of groundwater P on surface waters is determined by our limited observational data and system heterogeneity. There are, however, simplistic rules which govern the degree of risk which can be used at national scales. The two extremes of high and low risk of P entering groundwater bodies are described by the following scenarios:

- *Low Risk* – where there are thick soil deposits and thick, low hydraulic conductivity drift deposits overlying a porous media aquifer (dominated by matrix flow) with a deep water table and a high P sorption potential (Figure 8.2 (a)).

- *High Risk* – where there are thin soil deposits with no drift deposits overlying fractured or karstic geology with a shallow water table and a low P sorption potential (Figure 8.2 (b)).

At WFD catchment scales the primary controls on P pressure to surface waters from groundwaters are:

- the P pressure on the groundwater and
- the degree of connectivity between the groundwater and surface water.

Where groundwater monitoring data exists, it can be used in conjunction with base flow index (as a measure if the degree of connectivity) to estimate receptor P pressure (e.g. as EA, 2007). Where no monitoring data exist, P pressure on the groundwater can be estimated by pressure from point and diffuse sources coupled with soil, drift and aquifer property information (as indices of the likelihood of P transfer to, and within, the groundwater body) and combined with base flow index in a similar way. For higher tiered approaches more detailed analyses may be appropriate where sufficient data exists for evaluation: e.g. by combining information on the model components shown in the simplified conceptual model developed during the elicitation (see Figure 8.5).

The following research priorities were identified:

- Tracer studies (particularly at large scales) may help elucidate flow pathways and P sources.
- There should be a focus on the area where GW and surface waterbodies meet (the HZ and wetlands) as focus on processes closer to the receptor (stream) allows integrated

effects to be seen. These are ‘bottlenecks’ in the system so determination of the magnitude of their effect is a priority.

- Targeted GW P monitoring with appropriate limits of detection.
- Research into pathways for P into groundwaters: e.g. quantification of fluxes from anaerobic plumes from septic tanks and unlined farm slurry pits (Nicholson and Brewer,1997; Gooddy *et al*., 2001; Gooddy *et al*., 2002). See also the discussion in Section 4.

8.2 Introduction

Estimation of the likely impact of GW P fluxes on surface waters requires knowledge of GW body P concentration and the degree to which the GW body contributes to the surface water receptor. For England and Wales GW P concentration data are sparse, and there are problems relating to the limits of detection:- measurements are frequently below the limit of detection, and the limit of detection changes over time. Where no data exists, P pressure for GW bodies must be estimated using other methods to represent the P pressure to the terrestrial system and the likelihood of P entering, and not being retained by, the GW body. The understanding of P movement through groundwater is in its infancy and is reviewed in detail by (Holman *et al*., 2008). It is known, however, that diffuse P transport will be controlled by the hydrological and geochemical properties of the soil, drift and aquifer itself. The pre-interview phase conceptual model of the system developed by the experts is presented in Figure 8.1.

For the likelihood of transfer of diffuse P sources to groundwater there are also two ‘extreme case’ scenario models of the soil-drift-aquifer system:

1) *Low Risk* - Thick soils where there is a high probability of adsorption to mineral surfaces over a low permeability drift deposit. These are underlain by a porous media aquifer (e.g. Permo-Triassic sandstone) where there is no colloidal/particulate transport and sorption capacity. Generally, in this extreme case, there will be a deep water table. Hence this is a pathway with a high potential for rock-water interaction where P is adsorbed: the efficiency of the adsorption will depend upon the naturally occurring background P levels of the minerals themselves (Figure 8.2 (a)).

2) *High Risk* - thin soils, no drift deposits, over limestone or chalk where a karstic or fracture flow system gives high connectivity to the GW system and where there are low sorption capacities (Figure 8.2 (b)).

There is also an additional model which may modify the low risk case: where the thick drift may be fractured (e.g. fractured till), there is a risk that P could be transported down to the semi-confined aquifer where anaerobic conditions may exist, enhancing P transport in the aquifer.

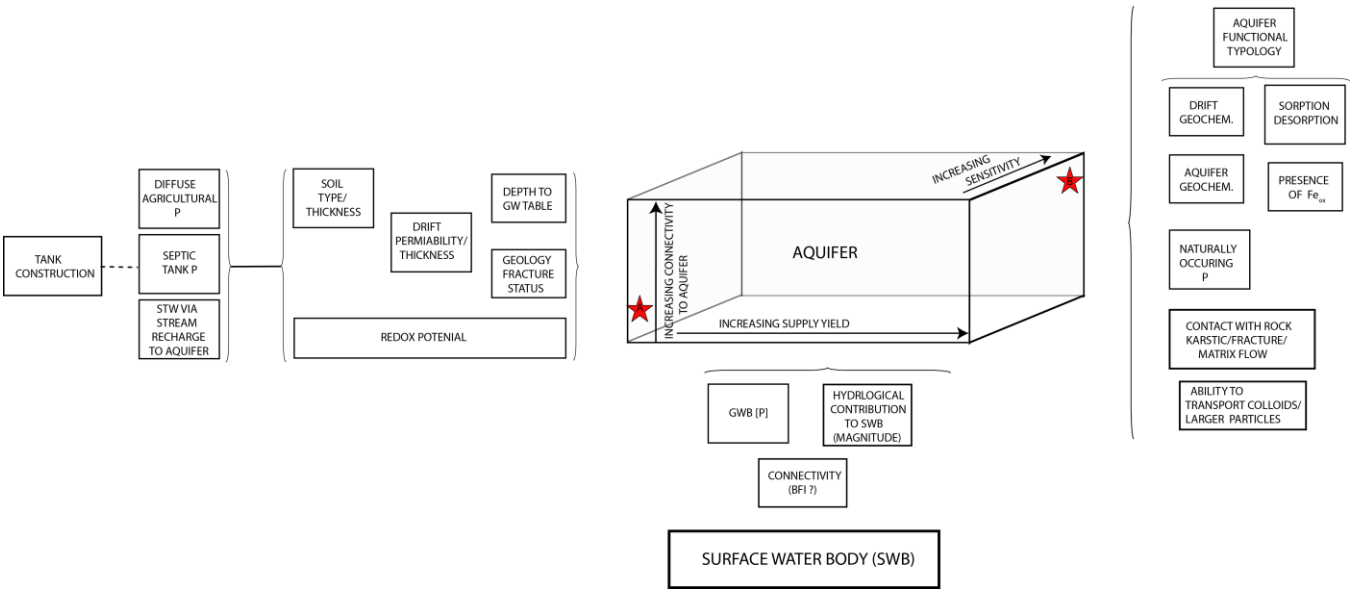


Figure 8.1 - Generalised schematic diagram of Terrestrial system-GW-SW exchanges. Note that additionally hyporheic zone processes can effect the flux of P to the SW. Cases A and B within the red stars also represent low and high risk cases where in addition to risk of P entering the waterbody, the risk associated with SW sensitivity and aquifer significance (yield) is represented.

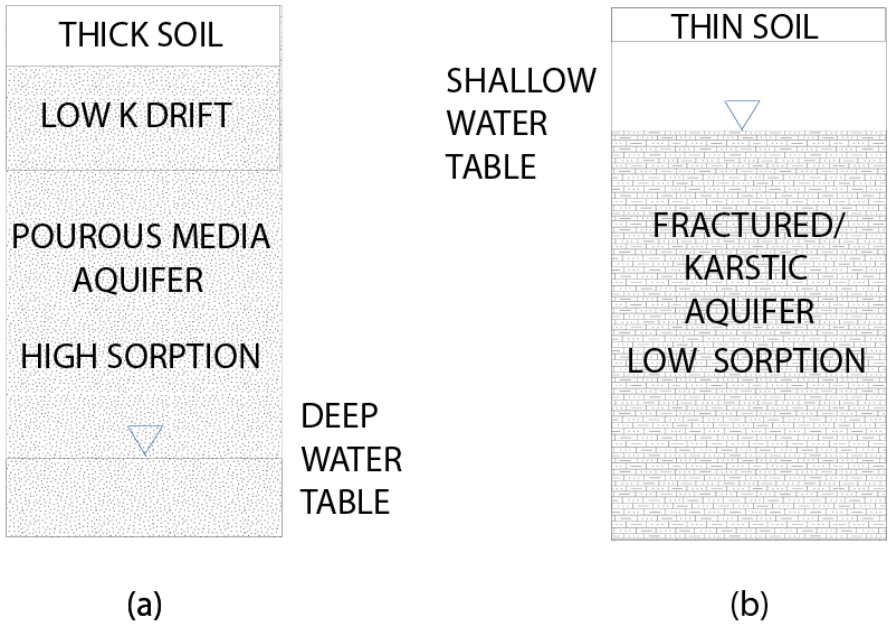


Figure 8.2 - (a) Low and (b) high risk of P flux to groundwater controlled by extremes of soil-geological situation (c.f. the conceptual models of Kilroy et al., 2001 and Holman et al., 2008)

8.3 Assumptions, definitions and caveats

- As there are no drinking water standards for P, surface waterbodies are the ultimate receptors for groundwater P.
- All nationally available data on GW P concentrations and catchment properties are available.

8.4 Chosen model components

Only one expert familiar with phosphorus in groundwater completed the fuzzy weighting exercise during the interview and the results for importance and overall uncertainty are presented in Figure 8.3. The discussion of these components in the next section does however include the opinions of two experts. Figure 8.3 highlights that the physical factors controlling the connectivity of the aquifer to the surface water body (such as the degree of surface water-groundwater interaction and fracture flow dominate the components chosen and those with high ratings together with the significance of the aquifer. Point sources of P are also seen as being significantly more important than diffuse sources. In most cases components given high importance ratings are also rated as uncertain making them priorities for research (e.g. degree of surface water-groundwater interaction, presence/absence of fracture flow and presence of point source contamination. In most cases the sources of the uncertainty were identified as resulting system heterogeneity and lack of data rather than scientific understanding. The model components identified in Figure 8.3 are discussed in the next section.

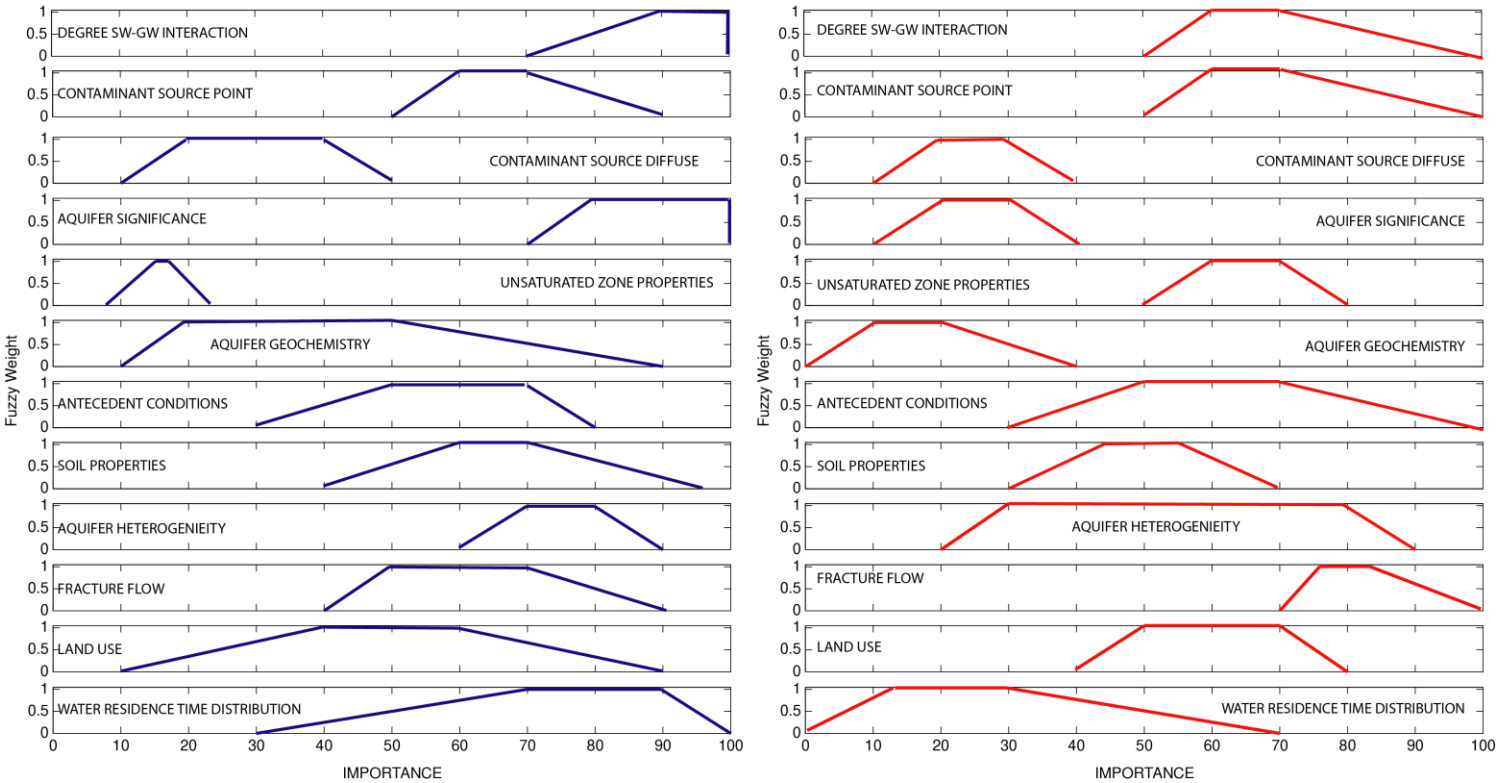


Figure 8.3 – Groundwater conceptual model components Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

8.5 Discussion of Model Components

8.5.1 Land Use (point and diffuse sources)

For the purposes of this section, land use describes a generic measure of likely P loading pressure to a GW body (e.g. as used for diffuse P by EA, 2007). However, the source type is an important factor and experts considered distributed point sources (e.g. septic tanks, slurry tanks etc.) to be a particularly important pathway for P to GW. Enhanced P transport to groundwater can occur where anaerobic conditions exist in pollutant plumes below these point sources, which results in sorption to the soil being less efficient (Expert 1g; Expert 2g; Gooddy *et al*., 2001; see also the discussion of septic tanks in Section 4). This problem may be exacerbated by poor construction or unsuitable location of septic and slurry systems whereby much of the soil is by-passed and the plume enters the soil unsaturated zone directly. A similar problem is thought to occur where there is leakage from urban sewers (Holman *et al*., 2008)

8.5.2 Soil properties – physical/geochemical

The soil physical and geochemical properties control hydrological pathways within the soil, have an influence on redox conditions and control P sorption-desorption reactions and P transport to GW (as for aquifer properties described below). Many of these properties are relatively constant in time but others are more dynamic: e.g. soil moisture conditions which play a role in controlling hydrological connectivity to the GW. In extreme cases soil cracking can change connectivity dramatically (Expert 2g). See also the discussion in Section 3.

8.5.3 Aquifer significance

Aquifer significance is important in terms of its supply yield, which is determined by its physical size and hydraulic conductivity/transmissivity.

8.5.4 Aquifer Properties

Presence/absence of significant fracture flow
Fracture flow is critically important as it allows turbulent flow to transport colloidal- and particulate-attached P to groundwater via a pathway where there is low rock-water interaction. Crucial here is the connectivity between fractures rather than fracture presence per se (Expert 2g), which together with aquifer heterogeneity controls the water residence time distribution from the surface system to the receiving surface water body.

8.5.5 Aquifer geochemistry

Aquifer geochemistry controls sorption of P onto aquifer solids that act as a retention mechanism within the aquifer. Enhanced adsorption occurs where aquifer geochemistry is high in Fe, Al and Mg oxides. See also the discussion of sorption-desorption reactions in Section 6. Aquifer geochemistry can also affect the behaviour of P when GW mixes with SW as it influences the amount of P that may be lost to stream bed sediments via Fe and Ca co-precipitation (Griffioen, 2006). See also the discussion in Section 6. ‘Natural sources’ of P in groundwater (e.g. Phosphatic chalks in particular: Expert 2) account for a significant input into the surface water system (e.g. Jarvis, 1992). This background or 'baseline' is difficult to quantify but is important when considering amelioration strategies.

8.5.6 Unsaturated zone properties

See soil and aquifer properties above.

8.5.7 Degree of groundwater surface water interaction

Ultimately, the 'bottleneck' in the system is the degree of GW/SW interaction. Once P has reached groundwater the speciation will not change significantly in UK aquifers (although the HZ - including riparian wetlands - can have an impact: see Section 7). The basic descriptor of groundwater- or non groundwater-dominated reaches/catchments can be classified using the HOST BFI estimate and may be particularly useful for nationwide approaches where no other data exists (e.g. EA, 2007). In statistical analyses comparing variables with measured GW P concentrations, EA (2007) found that only BFI showed a fit to observed data (variables considered were: BFI, soil leaching/groundwater vulnerability, HOST soil classification, absence of low permeability drift, and effective rainfall) but explained only a small part of the observed variation.

8.6 Observed groundwater phosphorus data

EA (2007) reported that Phosphate groundwater monitoring data were available for over 1000 monitoring locations (for the period 1998 to 2003). However, (Holman *et al*., 2008) identified that a significant number of samples were below the limit of detection and that detection limits were generally not consistent between data sets Moreover, and more importantly, the limit of detection can be higher than the P thresholds for surface waters bodies (Holman *et al*., 2008). These issues need to be taken into account within any risk analysis (e.g. see the treatment of data in EA, 2007)

8.7 Simplified groundwater conceptual model

Presented in Figure 8.4 is a schematic representation of the components identified by the experts to be included in a risk analysis conceptual model. The components chosen describe factors that control the likelihood of elevated P concentrations in GW and subsequent connection to surface waterbodies and can be aggregated to describe the following:

- Magnitude and type of P source.
- Physical and biogeochemical controls on P transfer to GW.
- Connectivity of GW to SW.

The estimation of P sources is discussed in Sections 3, 4 and 5 and are measures of potential P pressure to GW if used in conjunction with estimates of likelihood of transfer to waterbody (e.g. those estimated using soil properties). The final primary control, connectivity to SW, was thought to be most pragmatically described by BFI, which can be estimated nationally using the HOST database.

The conceptual model for diffuse pollutants reported by the EA (2007) uses two of the three primary controls listed above. A P pressure measure for the terrestrial system (using land use and livestock numbers) is coupled with the BFI to estimate GW P pressure. EA (2007) also used a parallel model using GW monitoring data and BFI to estimate an independent GW P pressure index which was subsequently combined with the first index described to give a composite P pressure index where monitoring data was available. In these analyses, EA (2007) gave a higher weighting to the indices calculated using the groundwater monitoring data. As a result of the significant uncertainties associated with these methods, EA (2007) assigned a low confidence to their pressure estimates, especially where they solely estimated P pressure in the absence of monitoring data.

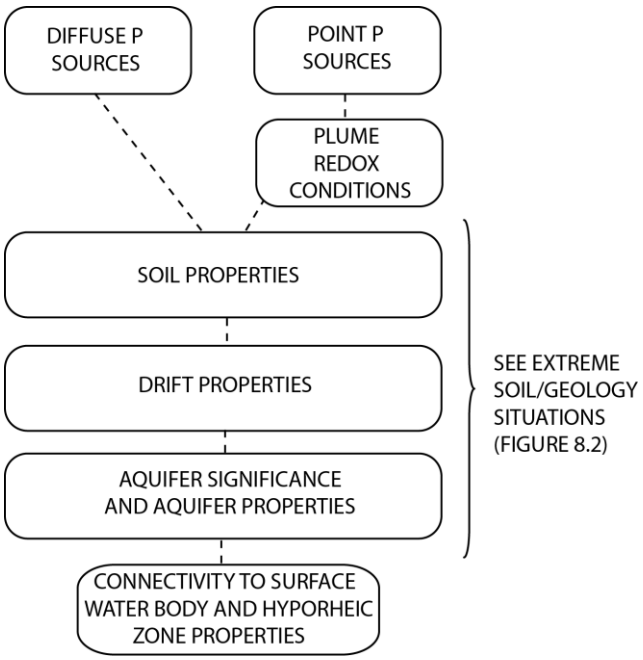


Figure 8.4 – Simplified conceptual model of primary factors controlling the risk to surface waters from elevated groundwater P concentrations.

8.8 Research Priorities

- There should be a focus on the area where GW and surface waterbodies meet (the HZ and wetlands) as focus on processes closer to the receptor (stream) allows integrated effects to be seen. These are 'bottlenecks' in the system so determination of the magnitude of their effect is a priority.
- Targeted GW P monitoring with appropriate limits of detection.
- Research into pathways for P into groundwaters: e.g. quantification of fluxes from anaerobic plumes from septic tanks and unlined farm slurry pits (Nicholson and Brewer,1997; Gooddy *et al*., 2001). See also the discussion in Section 4.

See also the recommendations of Holman *et al*. (2008).

8.9 Conclusions

As there are no drinking water standards for P and there are no measures of direct ecological effects in groundwater (GW), surface waterbodies are the ultimate receptors for groundwater P (EA, 2007). Principal sources of P in groundwaters (over and above naturally occurring and poorly defined background levels) are derived from diffuse agricultural sources, point agricultural sources (e.g. leakage from slurry tanks) and quasi-point sources: septic tanks and sewer leakage. Both point and diffuse derived P also enter groundwaters during recharge from rivers. The flux of P to groundwater is controlled by the degree of attenuation in soil and drift deposits. The subsequent flux of groundwater P into surface waters is primarily dependent upon further attenuation within the aquifer, the degree of connectivity between the groundwater and surface water bodies and modifications to P fluxes within wetlands and hyporheic zones. The degree of uncertainty associated with predictions of the likely magnitude of effect of groundwater P on surface waters is determined primarily by our limited observational data and system heterogeneity.

At WFD catchment scales, the primary controls on P pressure to surface waters from groundwaters can be described by the P pressure on the groundwater and the degree of

connectivity between the groundwater and surface water. Where groundwater monitoring data exists, it can be used in conjunction with base flow index (as a measure if the degree of connectivity) to estimate receptor P pressure. Where no monitoring data exist, P pressure on the groundwater can be estimated by pressure from point and diffuse sources coupled with soil, drift and aquifer property information (as indices of likelihood of P transfer to and within the groundwater body) and combined with base flow index in a similar way.

If calculated on a long-term average basis, these types of estimate give an averaged measure of groundwater interaction/contribution to surface water receptors; in reality groundwater contribution can be highly variable with higher base flow contributions during ecologically sensitive summer low flow periods. It is hence desirable that the importance of seasonality is taken into account.

9 Elevated Phosphorus Levels and Ecological Status in Rivers

9.1 Summary

The link between elevated P concentrations and ecological status is less clear for rivers than it is for lakes, primarily owing to shorter residence times and the influence of other factors such as disturbance by high flow velocities. The experts consulted generally agreed upon the way that eutrophication affects the abundance and composition of the aquatic plant community and on the factors that confuse simplistic P-ecological indicator relationships. The primary complicating components identified were the effects of flow regime (velocity, return period and duration of disturbance), grazing of biofilms and the bioavailability of P species; the experts were not fully clear on the significance of the role of alkalinity in changing the response of the ecological community to given levels of P and thought that the WFD SRP concentration standards adopted by UKTAG (which are set significantly higher for high alkalinity waters) may not be precautionary enough.

Given the complexities and uncertainties of the system it may be more realistic to use a more regime-based (see discussion below) representation of what is an acceptable river P concentration. A move away from simple threshold water quality standards may be desirable: distributions of acceptable P concentrations over space and time within a river network may be an improvement on threshold-type standards, which may not adequately distinguish between natural ecosystem dynamics and unacceptable anthropogenic change. A simplified conceptual model has been proposed (Figure 9.1) that may allow the focussed collection of data suitable for the development of a simplistic predictive tool under a regime-based philosophy. The creation of such a tool may however require a significant amount of data to be collected, the amount of which may only be known as progress towards these goals is made.

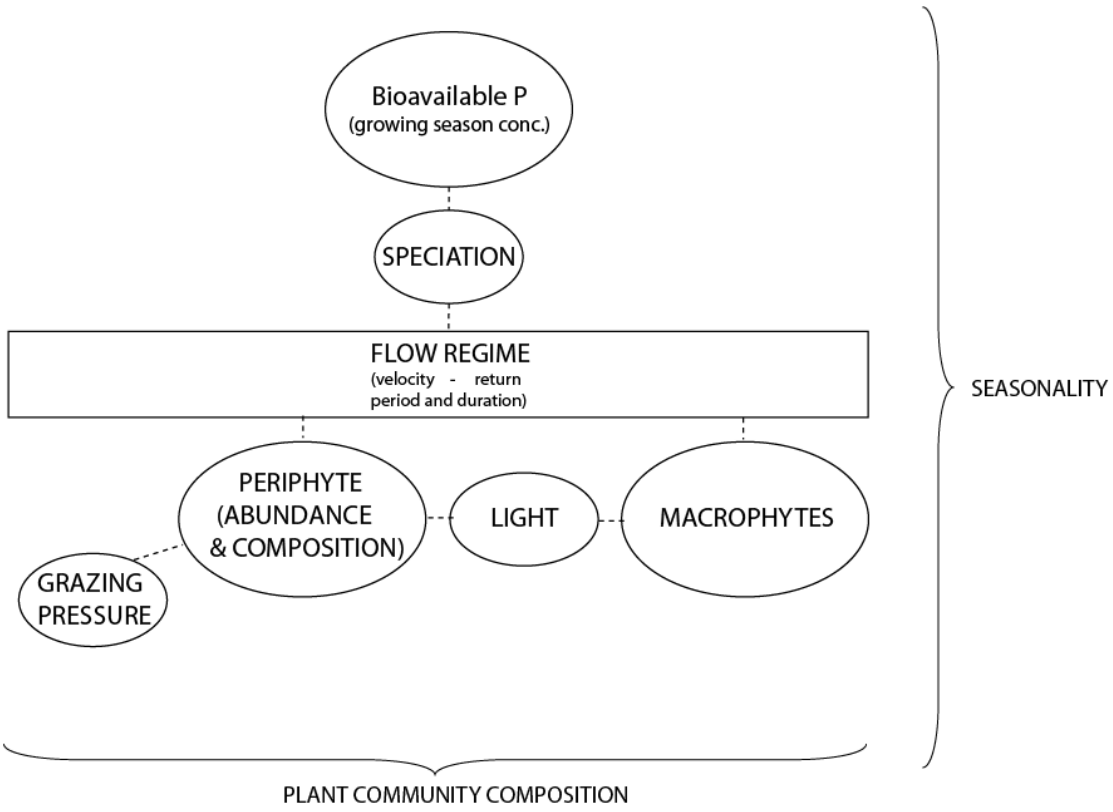


Figure 9.1 - Simplified post-interview conceptual for short residence-time rivers

9.2 Introduction

In this section the effects of elevated P concentrations on river ecology is considered. Only relatively short-residence-time rivers are considered here; rivers that have the tendency to function like lakes with significant phytoplankton communities are deemed to be covered by short-residence time lakes in Section 10. Consequently, we consider the degradation of the aquatic plant community composition which is deemed to be the primary indicator of eutrophic conditions. Elevated P levels are thought to be a primary driver of increased periphytic algal growth which reduces the available light to macrophyte leaves. This not only shifts the plant community composition from macrophyte-dominated towards periphyte-dominated but also towards macrophyte species tolerant of low light and eutrophic conditions (Mainstone and Parr, 2002; Hilton *et al*., 2006).

Hilton *et al*., 2006 developed a conceptual model of eutrophication development summarised in the following observable stages:

1. healthy stands of macrophytes, dominated by submerged species (**oligotrophic**);
2. healthy macrophyte stands with an increasing relative abundance of light efficient and/or near surface or floating-leaved and emergent plants with slight occurrence of epiphytic algal cover on the leaves and some benthic algae visible (**mesotrophic**);
3. macrophytes appear to be under severe stress with increasingly heavy epiphyte cover. Dominance by near surface or floating-leaved and emergent plants.
4. significant benthic algal cover (**eutrophic**);
5. very sparse or no macrophyte stands leaving heavy cladophora and/or benthic algal cover (**hypereutrophic**);

In the following sections other factors which may influence aquatic plant responses to elevated P levels are also considered and discussed in relation to the conceptual models developed by the experts (Figures 9.2 and 9.7). Although aquatic plants have been chosen to be the primary

indicators of ecological degradation by increased P concentrations, it must be noted that fauna may also be affected although there is no explicit coverage of these effects in this report. Effects can occur directly owing to algal response to enrichment (e.g. changes to bed sediment physico-chemical condition caused by epilithic algae and subsequent affects on fish egg survival), or indirectly by impacts on the higher plant community (e.g. loss of refugia).

9.3 Assumptions, definitions and caveats

Experts were not restricted to previously identified ecological indicators or river typologies, e.g. the use of benthic diatoms within altitude/alkalinity typologies as shown in Table 9.1 (Duncan *et al*., 2006) or those identified by the EA Rivers Task Team (0.1 mg l^{-1} SRP for calcareous waters and 0.04 mg l^{-1} SRP for non-calcareous waters).

- Experts were asked to comment on the ecology of rivers of relatively short residence-time, rather than rivers which may behave in a similar way to a lake.
- The importance of a chosen conceptual model component relates to its importance with respect to influencing the level of risk to ecology from elevated P levels or its importance as an ecological indicator.
- We are only considering a lumped external and/or bed sediment P and not individual P sources (e.g. STWs, septic tanks, diffuse etc.).

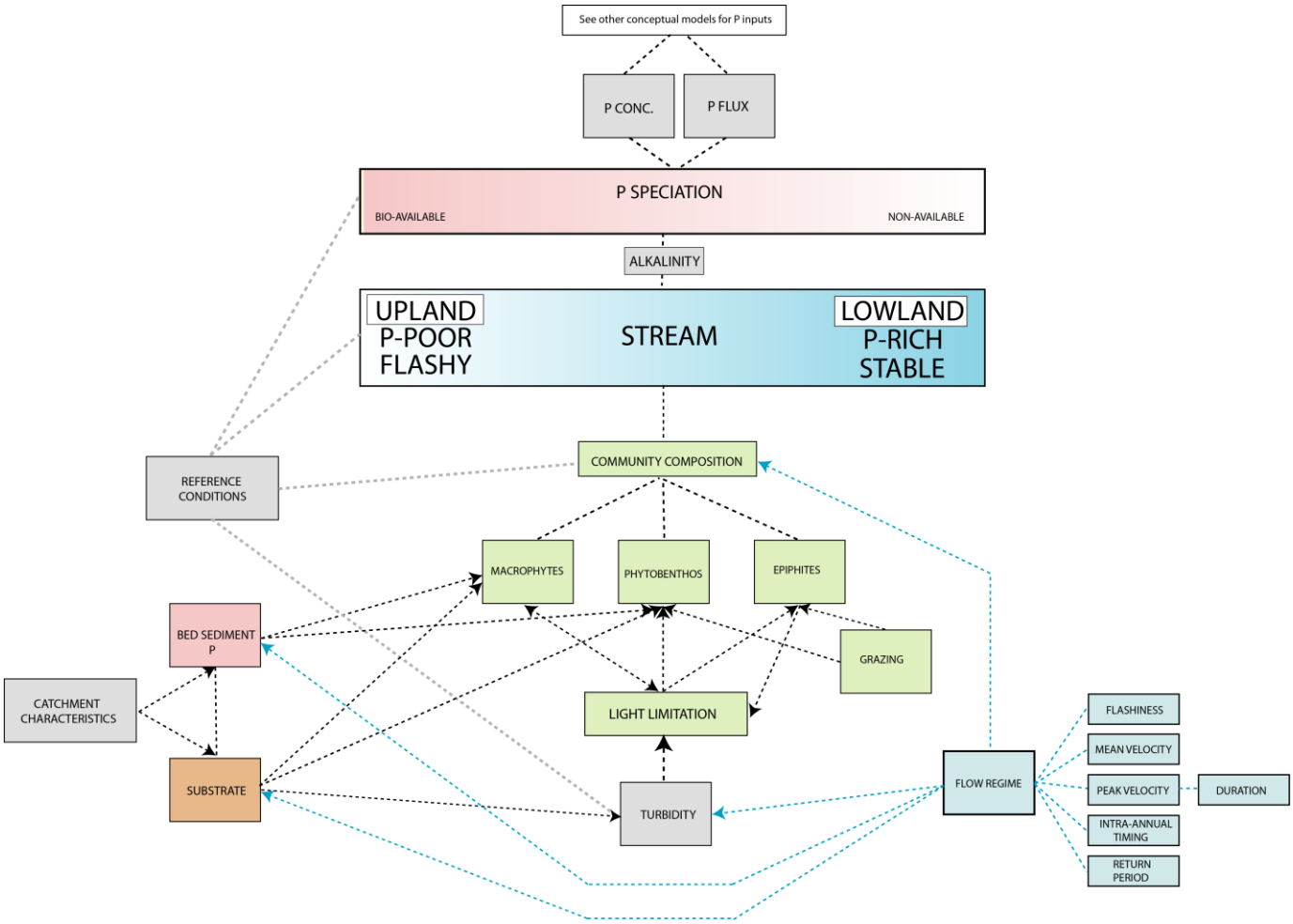


Figure 9.2 – Pre-interview phase conceptual model for short residence-time rivers

Table 9.1 - River typology classifications 1-4 used by Duncan et al. (2006)

	Alkalinity	
Altitude	< 50 mg l^{-1} $CaCO_3$	> 50 mg l^{-1} $CaCO_3$
< 80 m	1 (30; 50)**	3 (50; 120)**
> 80 m	2 (20; 40)**	4 (50; 120)**

** Values in parenthesis are soluble reactive P (SRP) regulatory standards for the HIGH/GOOD and GOOD/MODERATE WFD boundaries in $\mu g\ l^{-1}$.

9.4 Expert weightings for chosen model components

Presented in Figures 9.3 and 9.4 are composite (a combination of the distributions elicited from the experts: see Section 2) for *flashy* rivers (i.e. characterised by rapid and wide variations in flow) and stable flow rivers respectively. *Importance* weightings are presented in the left-hand pane and *overall uncertainty* weightings are presented in the right-hand pane. In general there was a high degree of consensus between experts on both the importance and overall uncertainty of components. Comparison between the two figures, separated on the basis of flow regime, indicates that, although the experts chose to identify the flashy and stable flow typologies, they did not rank the components significantly differently between the two: with the exception that the importance of external P load was thought to be higher for rivers with more stable flow conditions. It is also worth noting that three of the seven components relate to P supply: the relative importance of P load and P concentration and the importance of P speciation (here used to differentiate between immediately bioavailable and unavailable forms). The other chosen components are: macrophytes, absolute algal abundance (periphyton), flow regime and grazing of periphyton. These chosen components, the interactions between their effects and how his relates to assessing the risk from P to short-retention time river ecology is discussed below.

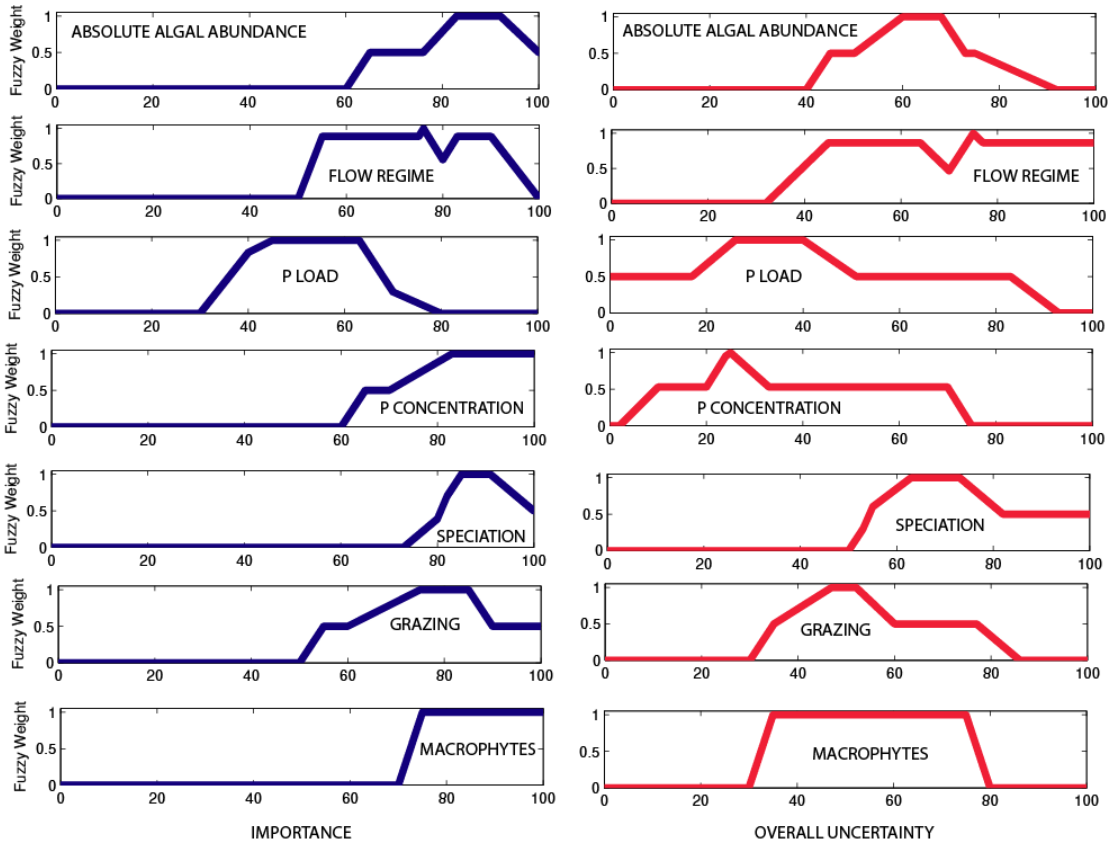


Figure 9.3. Flashy Flow Rivers Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane)

to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

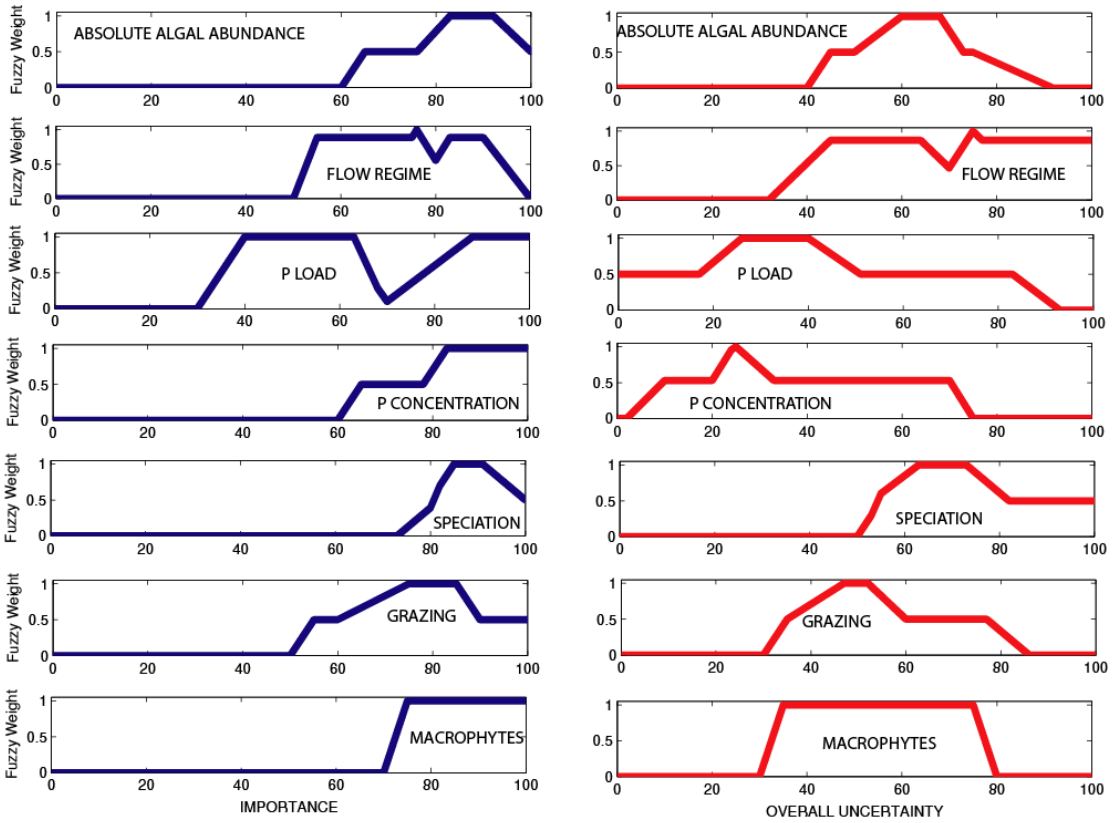


Figure 9.4. Stable Flow Rivers Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts.

9.5 Phosphorus-ecological indicator relationships

The most simplistic way of considering the problem of nutrient impacts on river ecology is to determine P concentration-ecological indicator(s)-biomass relationships such as those investigated by Bowes *et al*. (2007) for periphyton. This type of approach is analogous to the Vollenweider model use for lakes (Vollenweider & Kerekes, 1982: see also Section 10) which relates phosphorus concentration and phytoplankton abundance (expressed as the concentration of chlorophyll *a*). Within the Vollenweider model this simple relationship was corrected for different lakes' responses by lake depth/residence-time. In reality, there are many other factors which determine a given lake's response to a given P concentration. As shown for lakes, there have been studies indicating that a significant number of temperate rivers are limited or co-limited by nitrogen (Francoeur, 2001). For the short residence-time rivers considered here, the situation is complicated further by the variability of river flow regimes and the direct and indirect effects that flow dynamics have on aquatic plants. Direct effects relate primarily to the physical effects of high velocities (such as scour and uprooting) and indirect effects include the variation of nutrient concentrations with flow and substrate stability. Hence it is probably unwise to assume a priori that P is the primary controlling factor of autotrophic state in any given stream (Dodds, 2006).

In the following sections there is the basic assumption that relatively simple relationships between P concentration (e.g. some averaged measure) and ecological response (defined by the abundance and composition of the aquatic plants defined in Table 9.2) exist. The uncertainties in these simple relationships are assumed to result from the principal complicating

factors identified by the experts during the elicitation. These factors, together, can be considered the essential components of a river typology that is suited to meet the needs of the WFD. The typology need not represent discrete categories. These factors, of course, may interact significantly with one another making it difficult to resolve their individual effects even in data-rich situations. Interactions between these factors may contribute to the hysteretic relationships between P status and ecological status which are described as alternative stable states (Beisner *et al*., 2003; Scholten *et al*., 2003). These more complex dynamics are thought to be less pronounced for short residence-time river environments (Hilton *et al*., 2006) as a result of more frequent disturbances that 'reset' the system.

The primary components chosen by the experts to represent these complicating factors are shown in Table 9.2. Hilton *et al*., 2006 provide a comprehensive review of the role of these factors, as a basis of developing a conceptual model of UK river function with respect to eutrophication. The review does not, however, explicitly consider the uncertainty associated with making risk predictions on the basis of their combined effects, which is the focus here.

The EA LEAFPACS project (Willby *et al*. 2006; Penning *et al*., 2008) is currently developing a predictive system to assess the ecological status of rivers and lakes using macrophytes. LEAFPACS describes species as responding positively or negatively to eutrophication pressure, and defines a Lake Trophic Rank (LTR) score as an average of the scores of all species, weighted by abundance. It combines species-level values into a whole lake index (based solely on the species composition). The LTR should thus respond to a change in macrophyte community caused by a change in eutrophication pressure (Penning *et al*., 2008).

Table 9.2 Primary components chosen to represent P pressure and its likely ecological effect

Nutrient Pressures	**Quasi-fixed characteristic**	**Ecology**
P load (TP)	Flow regime (flashiness)	Absolute algal abundance (periphyton)
P speciation	*Alkalinity*[℘]	Macrophytes
P concentration (SRP & TP)		Biofilm grazing
Bed sediment P		

[℘] *There was not a clear consensus on the role and importance of alkalinity – see the discussion below.*

9.5.1 Flow regime control on ecological indicators

Although not explicitly in order of importance, Allan (1995) suggests that, for periphyton, scour by high stream velocities, light and nutrients are the primary controlling factors of biomass and that the effect of substrate type and grazing of periphyton can be important in some situations. Which of these factors will be most dominant can be localised: e.g. as a result of shading by woodland or spatial variations in flow velocity (e.g. dead zones or pools). In some cases, P-ecological response relationships can be dominated by river flow regime (Bunn and Arthington, 2002); flow regime control has been shown for epiphytes (Biggs, 1995; Murdock *et al.*, 2004) benthic algae (Biggs, 2000) and macrophytes (Riis and Biggs, 2003; Flynn *et al*., 2002; Wade *et al*., 2002b; Westwood *et al.*, 2007). In particular, the macrophyte species which have tended to dominate change as a result of the epiphytic film presence (i.e. a shift in the macrophyte community to species which grow higher in the water column to obtain more light). These species are at greater risk of up-rooting as they have a higher drag during hydrological events.

The effect of flow regime varies at the hydrological event scale as well as seasonally and may be best represented on the basis of the frequency (return period) of events of given magnitude

(velocity); Biggs *et al*. (1995) found a strong relationship between the 80th percentile of flow velocity and annual average periphyton biomass and (Lohman *et al*., 1992) showed that at enriched sites there was more rapid accrual of chlorophyll between disturbances. These may be useful parameters which can be assessed for UK growing season (nominally March-September in the UK) conditions: i.e. during ecologically sensitive periods (Hilton *et al*., 2006: Mainstone and Parr, 2002).

9.5.2 Biofilm Grazing

Grazing of biofilms by invertebrates can significantly affect the observed standing crop and community composition (Sumner and McIntire, 1982; Rosemond *et al*., 1993). Mulholand *et al*. (1991) showed using controlled laboratory streams that, in streams with a grazing population, in general the periphyton biomass was lower, the resilience (i.e. ability to return to a pre-disturbed state) of the periphyton community was lower but that the structure of the periphyton community was more resistant to flow disturbance. The latter resulted from a modification of the community structure to more prostrate species which supported previous work by Power and Stewart (1987). Steinman *et al*. (1991) state that grazing is most likely to be the primary control on periphyton biomass in physically stable systems. This was supported by Boulêtreau *et al*. (2006) who found that without inclusion of heterotrophic processes into their biofilm model accurate simulation of biofilm dynamics during summer low flow periods was poor. However, Welch *et al*. (1992) suggested that grazer densities would need to be very high to significantly reduce periphyton accrual in nutrient enriched systems.

Predation of invertebrates (and further top-down control with increasing trophic level) is a control of grazing potential (as is disturbance by high flow velocities: Poff and Ward, 1989) but is not explicitly considered here.

9.5.3 Light (as controlled by plant community composition)

The aquatic plant community composition is partly controlled by the competition for light: in particular, competition between epiphytes and macrophytes. Epiphytic algal communities have the direct effect of reducing the amount of light reaching macrophytes which can also have the indirect effect of changing the macrophyte community composition. The latter can change the community to light-seeking species that may be more susceptible to uprooting by high velocity flows (Hilton *et al*., 2006). Light limitation can be of particular importance where phosphorus concentrations are not limiting (Jarvie *et al*., 2002).

9.5.4 Phosphorus pressure and speciation: a bioavailable P measure

The flux of P that passes through a river reach is often partitioned between the soluble and particulate associated P forms. In practice the distinction is determined operationally from the design and implementation of chemical monitoring protocols. Soluble and PP are determined from fractions retained or not by passing samples through a 0.45 μm filter). The fraction of this P that is, or will eventually become available to, plant biomass is difficult to predict, is related to water and sediment residence-times and plant-specific uptake strategies etc. In general, filtered reactive P (often simplified to soluble reactive P) is used as an approximation of immediately bioavailable P. For these reasons, and specifically for rivers, it may ultimately not be important to quantify the flux of P through a reach, but to focus on the likely exposure concentration of bioavailable P in waters (e.g. as some time-averaged concentration during the growing season) and sediments. From a predictive point of view, and at the national scale, this may best be achieved by analysis of long-term P concentrations measured at GQA locations. The data collected at these locations, although only monthly, is in some cases monitored over a long enough period to establish with reasonable confidence the relationship between P concentration and flow magnitude (Figure 9.5) which would then allow a more confident estimate of growing season P concentrations. In general, the weakness associated with these data primarily

results from the stratified quasi-monthly sampling regime giving measurements which are inevitably weighted towards periods of low flow. This, however, is more of a weakness when trying to estimate a P flux. It is less problematic when trying to estimate the likely distribution of P concentrations that will occur during the growing season: when flows will generally be at their lowest, flow regimes will normally be more stable and reach residence-times will generally be longer. Thus, for rivers, a growing season average bioavailable P concentration (or a distribution of concentrations) in the water column is desirable as a measure of P pressure: this is possible using a subset of the GQA data (e.g. see Figure 9.6; note that EA GQA data is generally unfiltered reactive P rather than filtered reactive P). This conclusion is supported by Hilton *et al*. (2006) who state that a median growing season SRP concentration may well be a better indicator of the likely level of nuisance eutrophication than the annual TP load.

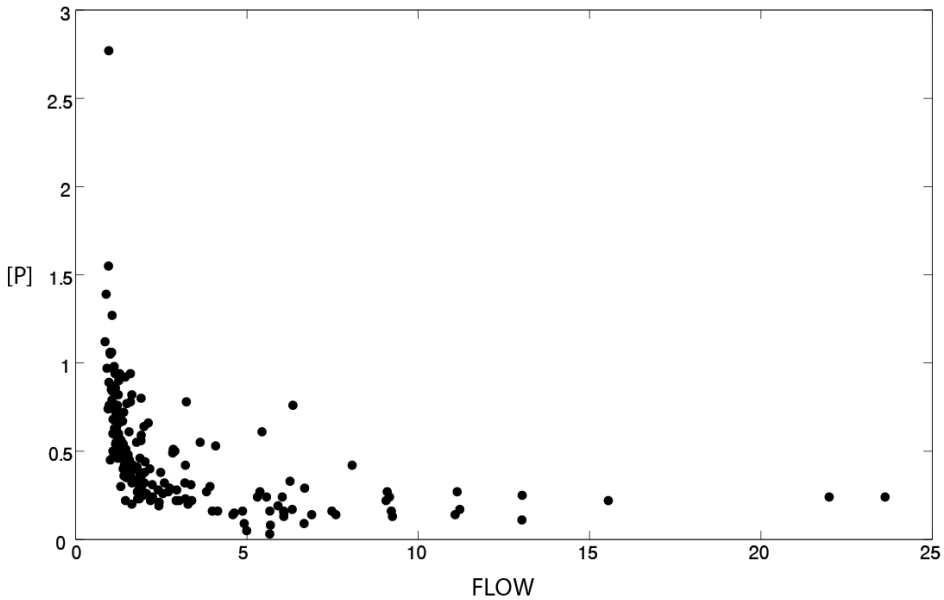


Figure 9.5 – An example of a point source dominated river reach showing dilution at higher flows (GQA sample site 88003554 unfiltered RP in mg l^{-1}: 1990-2005).

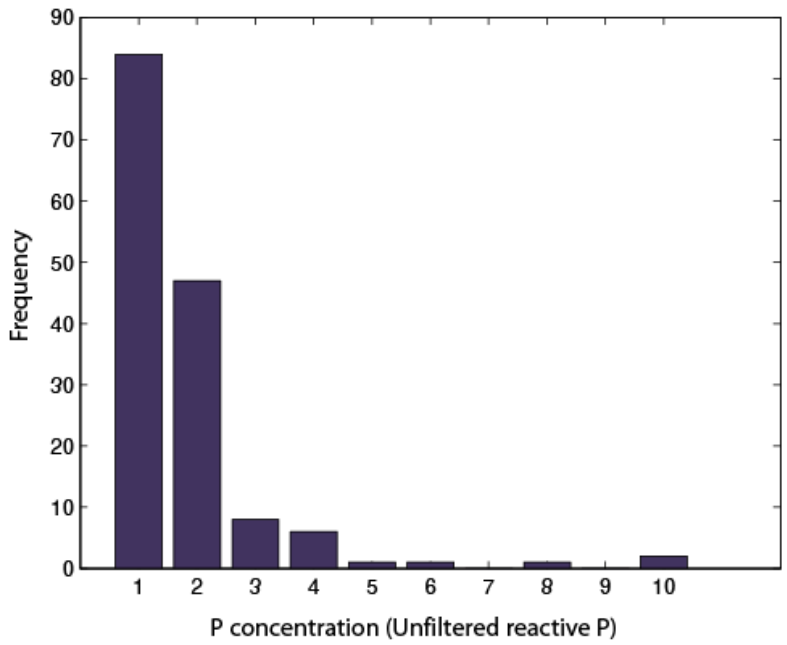


Figure 9.6 – An example of a distribution of growing season TRP concentrations (mg l^{-1}) at one GQA location for the period 1999-2005.

9.5.5 Bed Sediment P

The P associated with instream bed sediments that can be used by macrophytes, and to a lesser extent some periphytic species, is more difficult to assess (see Section 6) as routine measurements of sediment and sediment pore water P concentrations are not made at non-research sites. It may be possible to obtain an uncertain estimate of likely bed sediment P sources by estimating likely reach scale depositional regimes (Sections 6 and 7).

9.5.6 Ecological indicators

Given that water column P concentration observations will be our main indicator of the **potential** for eutrophic effects, any risk analysis will need to relate a level of P concentration to likely ecological effects. As noted above, UKTAG (2006) have assigned P concentration targets for WFD designations (see Table 9.1) using the community composition of benthic diatoms as ecological indicators of P status. This methodology uses the Diatoms for Assessing River Ecological Status (DARES: http://craticula.ncl.ac.uk/Dares/dares_project.htm), which considers the balance between nutrient tolerant and nutrient intolerant diatom taxa to give a Trophic Diatom Index. This index, however, only considers the diatom community composition, not the absolute abundance which is a WFD requirement resulting in a weakness in the method (Expert 1re). It also does not consider other important indicators, such as macrophytes and periphytic species, specified by the experts as important. Hence, It is a good indicator of elevated P concentrations but not necessarily a good indicator of overall ecological status. According to Experts 1re and 2re, good ecological status is better described by overall aquatic plant community composition (as controlled by the many factors discussed above). Importantly, ecological status should be measured by the deviation from an expected (i.e. the *reference state* defined for WFD purposes) plant community composition given other controlling factors such as flow regime, bed substrate, catchment geology, elevated N etc. (i.e. potential typologies).

9.5.7 The role of Alkalinity

The overall effects of alkalinity in rivers is not clear, the chemistry is not fully defined and causes and effects can be confused (Expert 3m). There are however some mechanisms which have been reasonably well-studied and reported. Alkalinity is known to have a function in controlling the exchange of P from sediments, where high alkalinities reduce P sorption and increase desorption of P bound to sediments (Reynolds, 2003). Alkalinity can also play a role in reducing instream P concentrations via the co-precipitation of P with calcite (as discussed in Section 6) but this may be primarily an algal-mediated process and limited in the water column. Thus, high alkalinities may increase or decrease water column PP concentrations.

Loss of P from the water column by precipitation is not a reason to allow higher residual P concentrations as high alkalinity does not necessarily mean that higher water column P concentrations are unavailable to biota. The experts consulted were unaware of any published evidence (apart from the UKTAG analysis of diatom communities discussed below) to indicate that higher SRP target concentrations should be set for high alkalinity waters; Expert 2re suggested that the SRP concentration targets presented in Table 9.1 above are not precautionary enough and Expert 1re thought that the standard for high alkalinity waters (120 $\mu g\ l^{-1}$) for the moderate/good WFD boundary may be more representative for the moderate to poor status boundary. Similarly, more representative targets for good/moderate and high/good would be 50 $\mu g\ l^{-1}$ and 30 $\mu g\ l^{-1}$ respectively. There is, however, some emerging evidence that P may be limiting below approximately 90 $\mu g\ l^{-1}$ (SRP) and that above this value other limiting factors (such as dissolved inorganic carbon and light) take over (Bowes *et al.,* 2007).

As well as controlling P speciation and dynamics, alkalinity can also affect aquatic plant community species composition where there is competition for dissolved organic carbon. Many submerged aquatic species are able to use bicarbonate as an alternative carbon source when dissolved CO_2 limits growth (Maberly and Spence, 1983; Reynolds, 1984; Jones *et al*., 2002). In high alkalinity waters with correspondingly high pH there will be lower levels of dissolved CO_2; where P status is also high and productivity is high there will be more competition for dissolved CO_2, high rates of photosynthesis will result in a rise in pH and limitation by dissolved CO_2 will be exacerbated giving a competitive advantage to bicarbonate utilizing species.

9.6 A simplified conceptual model riverine ecological response

Even a simplistic representation of the interactions between P concentration, flow regime, epiphytic growth and macrophyte composition/abundance (e.g. Figure 9.7) is not easy to represent at WFD surface water catchment scales (i.e. it is difficult to attribute a likely magnitude of effect of elevated P concentrations on these indicators given the complex nature of the system). The conceptual model presented by Hilton *et al*. (2006) also shows a schematic layout of this type of model. There are, however, very few quantitative data presently available with which to evaluate its structure. Thus for EA risk analysis purposes it is proposed that an even simpler model be developed using the primary components identified during the elicitation and currently available information. The components of the simplified model are:

- ***P concentration (and speciation):*** The likely distribution of growing season bioavailable P concentrations (i.e. estimated from EA GQA data).

- ***Flow Regime:*** A measure(s) of seasonal flow regime (e.g. flashiness, return period of high velocity events etc.: see Clausen and Biggs (1997) and Clausen and Biggs (2000) for a discussion of which flow variables are best suited to representing disturbance to periphyton and macrophytes).

- ***Grazing Potential –*** (there are some family-level data available from RIVPACS)

Flow regime and grazing potential in this case represent a river's sensitivity to elevated P concentrations.

- ***Aquatic Plant Community Composition:*** Macrophyte and periphyton abundance and composition data (There are currently little data available for these variables).

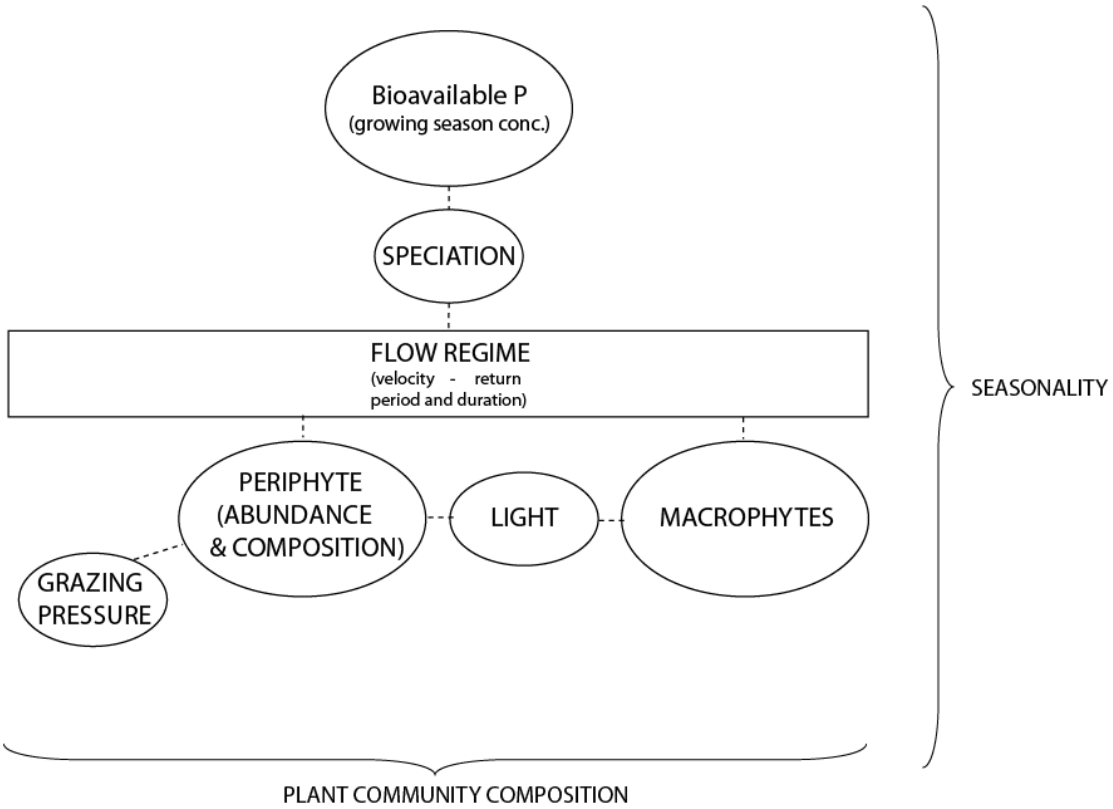


Figure 9.7 - Simplified post-interview conceptual of short residence-time rivers

9.7 Regime-based river water quality standards for Phosphorus

Given the complexity of the system and the uncertainties associated with linking a given magnitude of detrimental effect to a given P concentration it may be more realistic to use a more regime-based representation of what is an acceptable P concentration. A move away from simple threshold water quality standards has been proposed by Poole *et al.* (2004) who call for a system whereby desirable distributions of conditions over space and time within a stream network are specified. Poole *et al.* argue that threshold standards may not adequately distinguish between natural ecosystem dynamics and unacceptable anthropogenic change. Rather than solely the magnitude of a water quality determinand above a threshold being seen as a failure, the degradation of good water quality is likely to manifest itself as changes in magnitude, frequency, timing, location and spatial extent. For these reasons comparisons of likely distributions of water quality parameters to distributions of 'acceptable' water quality parameters may be a better option. This type of approach is consistent with a more probabilistic approach such as that of Dodds (2006) who developed frequency distributions in order to estimate the probability of a given chlorophyll level for a specified nutrient concentrations (using TP and TN). Dodds (2006) did not, however, consider the interactions with other important variables (such as those identified here) which may be required to explain differences in system responses. Biggs *et al*. (1998) did take into account other factors and used the primary variables of flood disturbance, nutrient resource supply and invertebrate grazing to develop a habitat matrix conceptual model for stream periphyton. Their matrix allowed prediction of a biomass gradient from *low* in frequently disturbed low nutrient habitats to

high in infrequently disturbed nutrient-enriched habitats. This type of approach could be used for the aquatic plant community considered important by the experts in this study and adapted for conditions found in England and Wales and extended to incorporate the regime-based approach of Poole *et al.* (2004). If this was to be attempted, it would be necessary to collect data on the matrix variables for river typologies as a research priority.

9.8 Research Priorities

It seems clear from the above discussion that the primary research priority is to identify the likely magnitude of effect of the most important variables identified for this study at WFD reach scales, for a representative and structured sample of such reaches, such that progress can be made on producing a predictive tool that takes into account the natural variability of river systems: this is discussed further in Section 11. From the discussion above the priorities highlighted are the estimation of growing season bioavailable P concentrations, quantifying the effect of flow regime on aquatic plant communities for UK rivers (or river typologies) and the quantification of the effect of biofilm grazing and estimating grazing potential for UK rivers (or river typologies).

Within the variables identified, further priorities are implicit from the elicited fuzzy distributions (Figures 9.3 and 9.4 above): i.e. those components which were rated as highly important and highly uncertain. This is expressed simplistically in Figure 9.8 by a weighting calculated from the product of the ratings (the centroid of each fuzzy distribution) for each chosen component. These figures show that the roles of P speciation, flow regime, macrophytes and grazing pressure are research priorities: algal abundance has a high weighting owing to its overall importance as an ecological indicator. Further research into these highlighted variables is consistent with the aim of creating a predictive tool. Ideally, a predictive tool would include the ability to predict the interacting effects of theses variables but the data required to be able to resolve these relationships may be too great and will only be known through initial research initiatives.

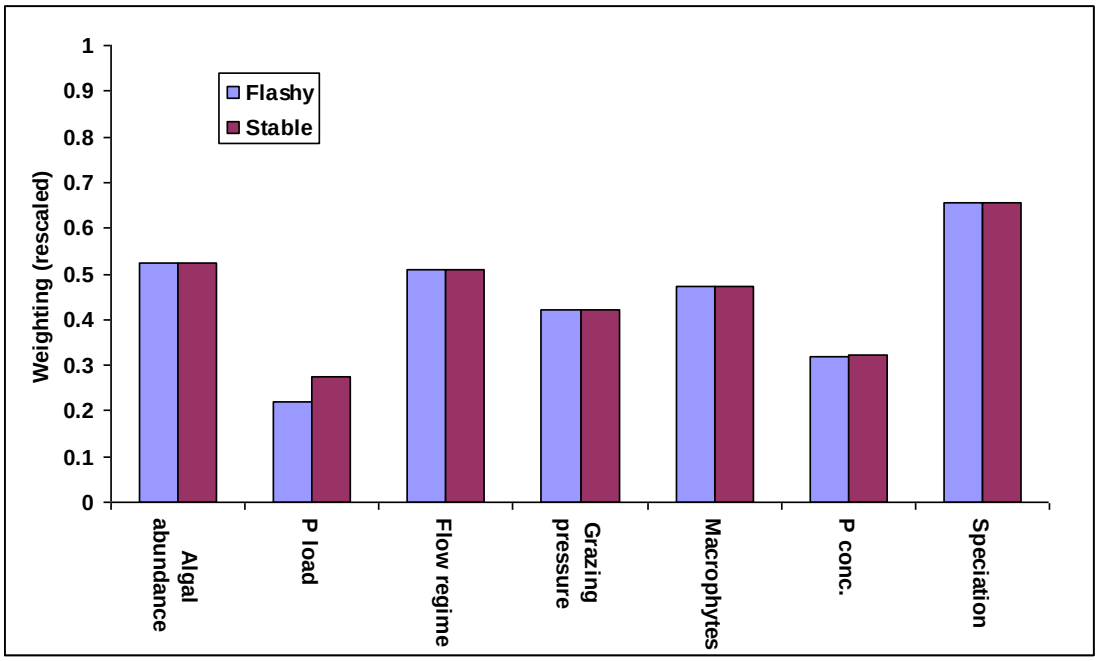


Figure 9.8 – Ratings for the river ecology model components with regard to research priority estimated using the product of the centroid of the importance and overall uncertainty fuzzy distributions.

9.9 Conclusions

The link between elevated P concentrations and ecological status is less clear for rivers than it is for lakes, primarily owing to shorter residence times and the influence of disturbance by high flow velocities. The experts consulted generally agreed on the effects that eutrophication had on the abundance and composition of the aquatic plant community and on the factors that lead to complicated P-ecological indicator relationships. The primary complicating components identified were the effects of flow regime (velocity and return period and duration of disturbance), grazing of biofilms and the bioavailability of P species; the experts were not fully clear on the significance of the role of alkalinity in changing the response of the ecological community to given levels of P and thought that the WFD SRP concentration standards adopted by UKTAG (which are set significantly higher for high alkalinity waters) may not be precautionary enough.

Given the complexities and uncertainties of the system it may be more realistic to use a more regime-based representation of what is an acceptable target P concentration. A move away from simple threshold water quality standards may be desirable: distributions of acceptable P concentrations (and ecological indicators) over space and time within a river network may be an improvement on threshold-type standards, which may not adequately distinguish between natural ecosystem dynamics and unacceptable anthropogenic change. A simplified conceptual model has been proposed that may allow the focussed collection of data suitable for the development of a simplistic predictive tool under a regime-based philosophy. The creation of such a tool may however require a significant amount of data to be collected the amount of which may only be known as progress towards these goals is made.

10 Elevated Phosphorus Levels and Ecological Status in Lakes

10.1 Summary

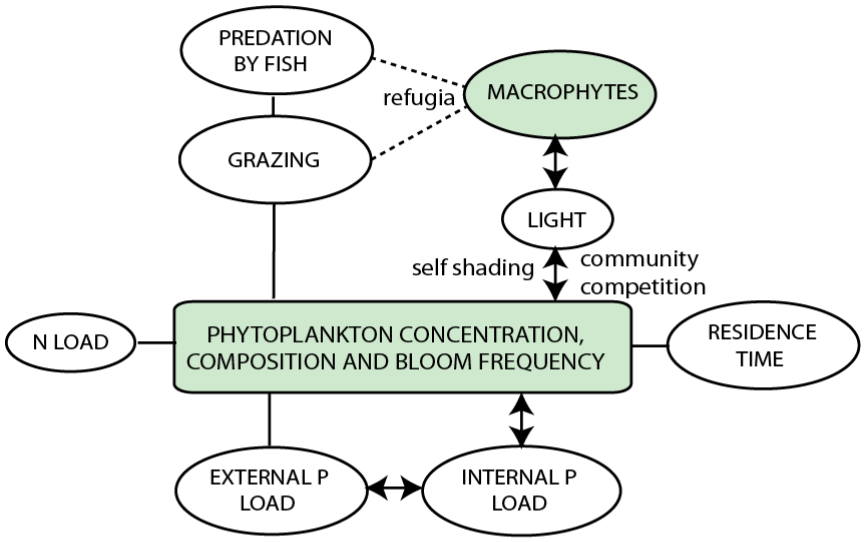
This chapter investigates the uncertainty in assessing the risk to lake ecology from elevated P status. The discussion focuses on the factors that confuse the P-ecological status relationship where ecological status is described by total chlorophyll *a* (Ch*a*) and macrophyte status. There is significant variability in the relationships between P status and these indicators making dose-response type relationships for WFD use uncertain. The experts consulted identified conceptual model components critical for inclusion in EA WFD risk analyses, although data will not be available for all components at a Tier 1 level. These components, listed below, have been used to create the Tier 1 and higher tier conceptual models presented for shallow and deep lake typologies in Figures 10.1 and 10.2 respectively.

- External P load;
- Internal P load (within lake P cycling);
- Flushing Rate (residence time);
- P speciation (bioavailability);
- Other nutrients;
- Light (availability/limitation);
- Sedimentation (of phytoplankton cells and zooplankton faeces);
- Mixing of algae between the aphotic and photic zones;
- Macrophytes (interaction with other plant species);
- Grazing (of phytoplankton by zooplankton);
- Fish (predation of zooplankton, bioturbation).

These components interact within complex, dynamic and stochastic lake system processes which can shift to alternate stable ecological states at a given P status. Thus, given the uncertainties associated with subsequent WFD designations it may be beneficial to use a more holistic approach to determining lake ecological status, perhaps including regime-based representations that allow for natural system variability rather than dose-response relationships. Risk analyses methodologies are discussed further in Section 11.

SHALLOW LAKES

TIER 2



TIER 1

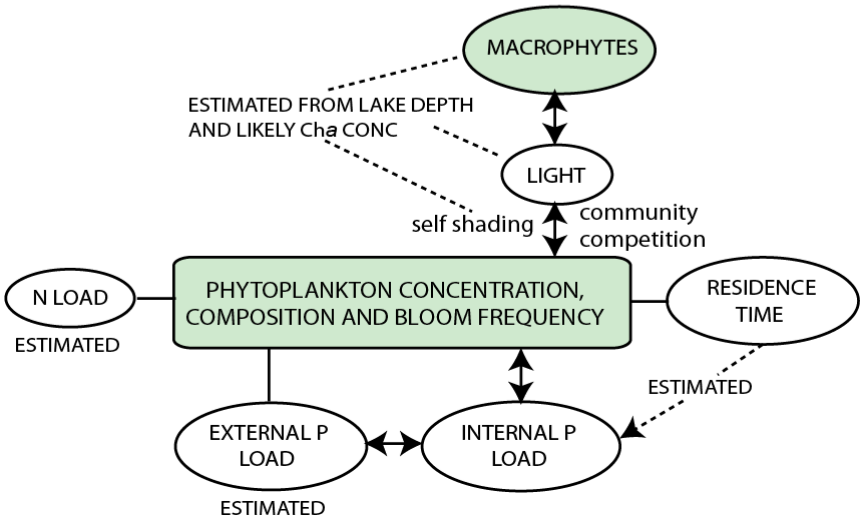


Figure 10.1 – Simplified conceptual model for shallow lakes including the primary components that determine the ecological response to changes in P status. Tiers 1 and 2 refer to the two different data scenarios: nationally available data and derived (modelled) datasets and higher resolution lake-specific observations

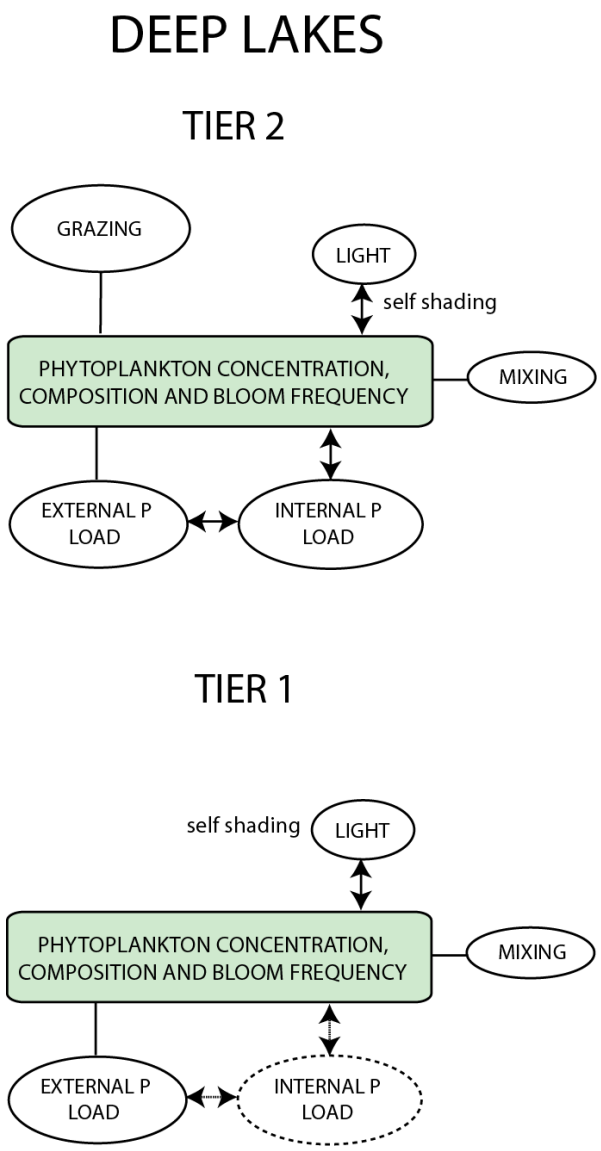


Figure 10.2 – Simplified conceptual model for deep lakes including the primary components that determine the ecological response to changes in P status. Tiers 1 and 2 refer to the two different data scenarios: nationally available data and derived (modelled) datasets and higher resolution lake-specific observations

10.2 Introduction

A risk of reduced ecological status from increased lake P status implies a relationship between the two. Phosphorus is often regarded as the master variable in determining the ecological status of lakes within a given biogegraphical area (such as Europe), but the relationship is not simple (e.g. see Dodds, 2006 and Elser *et al.*, 2007). Lack of a well-defined relationship makes implementation of threshold-type measures of acceptable P status (i.e. for each of the WFD designation boundaries) problematic and increases the chances of misclassification. A question that remains to be fully answered is whether or not a lake with elevated (from a reference condition) P status (i.e. *eutrophic*), but that does not show the signs of eutrophication (i.e. not *eutrophied*: Schoulten *et al.,* 2005), can be regarded as in good ecological status (e.g. see the approach set out in Environment Agency, 2007b). Pragmatically, and where only an estimated lake P status exists, it is likely to be the case that a designation must be made on this basis, but that the designation will be uncertain, compared to the case where in-lake P and ecological observations are available. This uncertainty is reflected to some degree by the confidence weightings given to designations under EA protocols (Environment Agency, 2007b). This type

of approach which defines ecological status upon threshold-type chemical standards has been criticized for not being able to take into account natural variability (Poole *et al.*, 2004), both within and between lakes, and not being holistic enough for such complex systems (Moss, 2007).

In this section, the focus is on the principal factors that complicate the P status-ecological status relationship and hence makes the designation of ecological status uncertain, where only modelled estimates of P status exist. We also consider the reduction in uncertainty where lake specific information is available (see the higher tier data scenario example that follows in this section). The discussion is based upon the opinions elicited during the interview phase of the project where experts were asked to consider the problems associated with assessing the risk to lake ecology from P: specifically to answer the following questions:

- what is the risk to lake ecology from elevated P concentrations?
- what are the primary indicators of ecological degradation resulting from elevated P concentrations?
- what is the relationship between ecological indicators and P concentration/load and what are the complicating factors that lead to uncertainty in assessing the risk from P pressures?
- what are the difficulties and uncertainties associated with using a P threshold-based water quality standard?
- how can the uncertainties associated with this type of risk analysis be reduced?

10.3 Assumptions, definitions and caveats

- The most appropriate indicators of lake ecological status (with respect to P) are the abundance of total phytoplankton chlorophyll *a* (and community composition) and the abundance and community composition of macrophytes.

- Experts were asked to comment on the ecology of lakes of relatively long residence-time, rather than lakes which may act in a similar way to a river. The operational definition used is defined as:

  Lakes where there is high potential for a significant phytoplankton community to develop, but not necessarily for the whole year.

- Experts were not restricted to previously identified ecological indicators or lake typologies (e.g. phytoplankton abundance within lake depth/alkalinity typologies as shown in Table 10.1 and Table 10.2: Carvalho *et. al.,* 2005 and Environment Agency, 2007b respectively). After the interview phase of the project it was agreed that the primary ecological indicators are phytoplankton abundance (Ch*a*), phytoplankton community composition and macrophyte abundance and composition. Initially the typologies considered were: very shallow, shallow, deep (and Alkalinity ranges for each). After two of the three experts chose to comment on a split between shallow and deep lakes, it was agreed to use solely shallow (nominally 0-3m mean depth) and deep (nominally > 3m mean depth) lakes and not to consider alkalinity explicitly (see discussion below).

Table 10.1 Provisional lake typology for Great Britain (after Carvalho et. al., 2005)

	Peaty	**Low-alkalinity**	**Medium alkalinity**	**High alkalinity**
Alkalinity (µequiv.l^{-1})	<0	<200	200-2000	>2000
Shallow (0-3 m)	1S	2S	3S	4S
Deep (>3 m)	1D	2D	3D	4D

Table 10.2. Typologies for EA lake P risk analysis (after Environment Agency, 2007b)

Lake Type		Reference TP for Risk Assessment (μg l^{-1})
Peat	Deeper (>3m)	8
	Shallow (≤3m)	15
Low Alkalinity	Deeper (>3m)	7
	Shallow (≤3m)	11
Moderate Alkalinity	Deeper (>3m)	11
	Shallow (≤3m)	19
High Alkalinity	Deeper (>3m)	29
	Shallow (≤3m)	32
Marl	Deeper (>3m)	11 see foot note[1]
	Shallow (≤3m)	19 see foot note
Brackish	Deeper (>3m)	29 see foot note
	Shallow (≤3m)	38

[1] *Insufficient data prevented Carvalho et al (2005) from providing a value. Marl lakes have been assumed to take the same standard as Moderate Alkalinity Lakes, Deep Brackish Lakes the same value as Deep High Alkalinity lakes.*

10.3.1 Definitions used within the elicitation interview

- The importance of a component relates to its importance with respect to the risk to lake ecology from elevated P levels or as an indicator of ecological status.

- Data Scenarios used:
  - Data 1 - Export Coefficient type model calculated from readily available national scale meteorological and catchment data, using the OECD regression (OECD, 1982) to convert P load to P concentration: using Meteorological Office and other readily available national data.
  - Data 2 – e.g. phytoplankton abundance and composition, P conc. temperature, etc. collected from a buoy located in the centre of the lake, 4 times per year at regular intervals. In addition secchi depth, a standard macrophyte and fish survey (5 yearly interval), integrated depth O_2 and temperature measurements would also be available.

- Only 'lumped' external or internal P loads were considered: not individual P sources (e.g. STWs, septic tanks, diffuse etc.). The components of riparian and external P not measurable from primary river inflows were also considered.

10.4 Ecological response to elevated lake P levels

For the purposes of this discussion, we will start from the assumption that indicators of lake ecological status are directly related to lake P status. The experts consulted for this study considered phytoplankton to be the primary indicator (Carvalho *et al*., 2005 also chose phytoplankton as it is the most sensitive indicator of changing nutrient conditions) but that macrophytes become important for shallow lakes. One of the most well known and much used predictors of trophic status is the Vollenweider regression: a regression between annual average lake TP concentrations (normalised by residence time and mean lake depth) and annual average Ch*a* concentrations (OECD, 1982; Vollenweider and Kerekes, 1980;

Vollenweider and Kerekes, 1982). This interlake regression shows a strong relationship, but also significant variability (approximately an order of magnitude range around the regression line using a 99% confidence interval). If this range is presented as trophic classifications (Figure 10.3 and Table 10.3), the magnitude of class overlap is a stark indicator of the challenge for robust lake classification based upon lake P status. Vollenwieder (1976) warned that this type of relationship is only illustrative and should not be used too literally: uncritical use as a predictive tool without considerations of its limitations is problematic: P status is a reasonable general indicator of trophic status but is not, in itself a reliable causal predictor of eutrophication problems (Reynolds, 1992) or reference state. For 131 UK lakes, Carvalho *et al*. (2005) found a similar magnitude of variability in the TP-Cha relationship (see Figure 10.4) and also found that, using analysis of variance tests, only lake depth (from the typological descriptors considered) was significant in explaining any of the observed variation. There are many factors that may confound this relationship, such as daphnid grazing of algae (Moss, et. al., 1996; Reynolds 1998) and limitation by light or other nutrients (Elser *et al.*, 2007). Some of the primary factors affecting this relationship are presented in the conceptual model developed for the interview phase of this study (Figure 10.5). The components chosen from the conceptual model (by the 3 experts) as the most likely source of the observed variability are discussed in more detail below.

There are historical reasons for P being the primary nutrient linked to trophic state: (i.e. the experiments in the Canadian Shield region, where there are very low background P concentrations: Schindler, 1974). The findings from the Canadian experiments work well for naturally oligotrophic systems (e.g. the English Lake District) but do not extrapolate well to lowland lakes (e.g. the Midland Meres and Norfolk Broads). In general, the simplistic P-ecological status relationship is less valid the shallower the lake and the warmer the climate; for example, Moss (1989) found that the Vollenweider regression was too simple to explain the algal dynamics of the Norfolk Broads (see also the analysis of (Elser *et al.*, 2007) which showed that, globally, limitation and co-limitation of primary production by N is as common as that of solely P limitation).

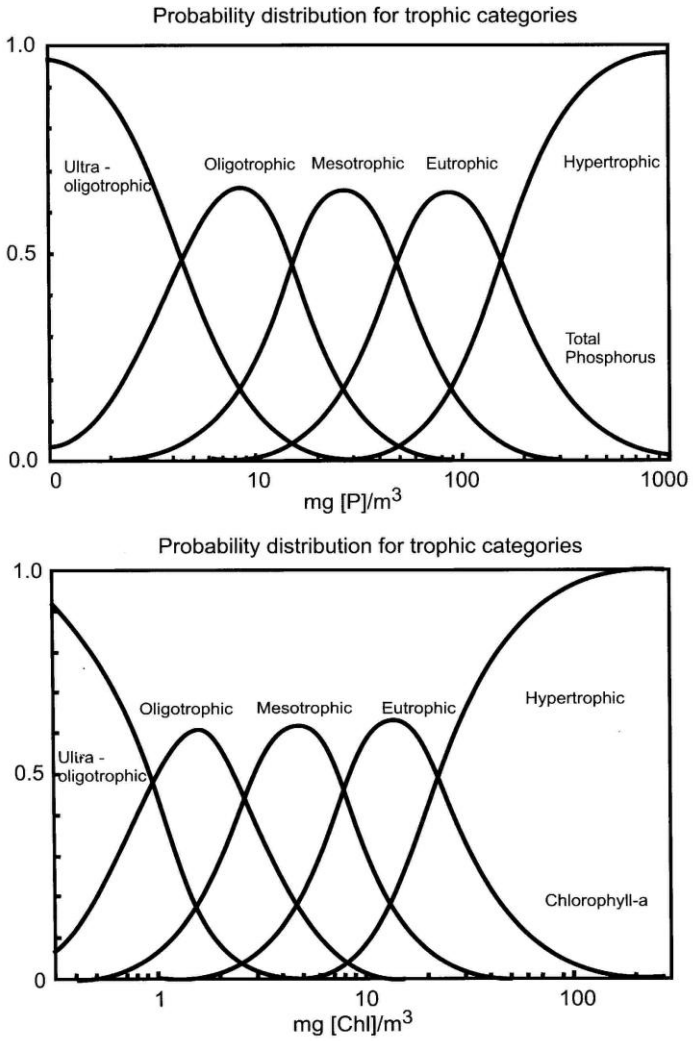


Figure 10.3 – Classification of lakes on the basis of annual average TP load (OECD, 1982)

Table 10.3 Trophic categories of lakes adopted by Reynolds et al. (2002)

Code	Category	Max. biomass Carbon (mg C/m^3)	Max. chloro-phyll (mg /m^3)	TP (mg P/m^3)	Period when MRP < 3 mg /m^3
U	Ultraoligotrophic	< 500	< 10	< 3.5	Always
O	Oligotrophic	500 – 1250	10 – 25	3.5 - 10	9-12 months
M	Mesotrophic	1250 – 2500	25 – 50	10 - 35	4-9 months
E	Eutrophic	2500 – 5000	50 – 100	> 35	< 4 months
H	Hypertrophic	> 5000	> 100	> 100	Never

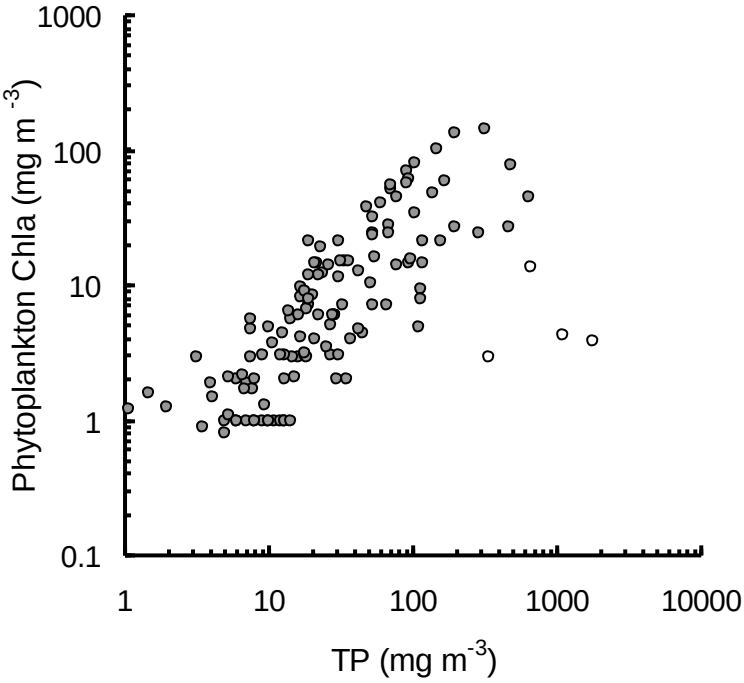


Figure 10.4 *Log-Log scatter plots of annual mean concentrations of phytoplankton chlorophyll-*a *against total phosphorus for 131 lakes (UK Lakes database): after Carvalho et al. (2005)*

Apart from direct effects of various factors on the simplistic P-Cha between-lake relationship, for a given lake, it cannot be assumed that Cha biomass will increase or decrease smoothly or proportionally with varying P concentration. This is particularly the case where P is not limiting (Reynolds, 1992) and/or where hysteretic shifts occur between stable ecosystem states because of feedback effects (Moss, 1991; Scheffer *et al.,*1992; Scheffer,1998; Scholten *et al*., 2005). These problems have crucial implications for making decisions about ecosystem reversibility in response to reduced P inputs (Dent *et al*., 2002) or the onset of eutrophic effects. Hosper (1998) showed that remediation of shallow lakes in the Netherlands was unsuccessful when concentrating solely on reducing external P inputs: they concluded that whole ecosystem approaches were required for success owing to the resilience of different stable states.

The variability of the relationship between P status and ecological status gives rise to challenges for WFD implementation. Søndergaard *et al*. (2005) have similarly highlighted the large variability of indicators within given lake classes, gradual rather than stepwise changes for all indicators of ecological status, and problems using the one out–all out WFD principle (European Communities 2003) for lake classification where different indicators suggest different WFD classifications. The Moss *et al*. (2003) and Moss (2007) systems, overcome these problems to some degree, by taking account of the natural variability of chosen indicators when assessing ecological status (these are discussed further in Section 11)

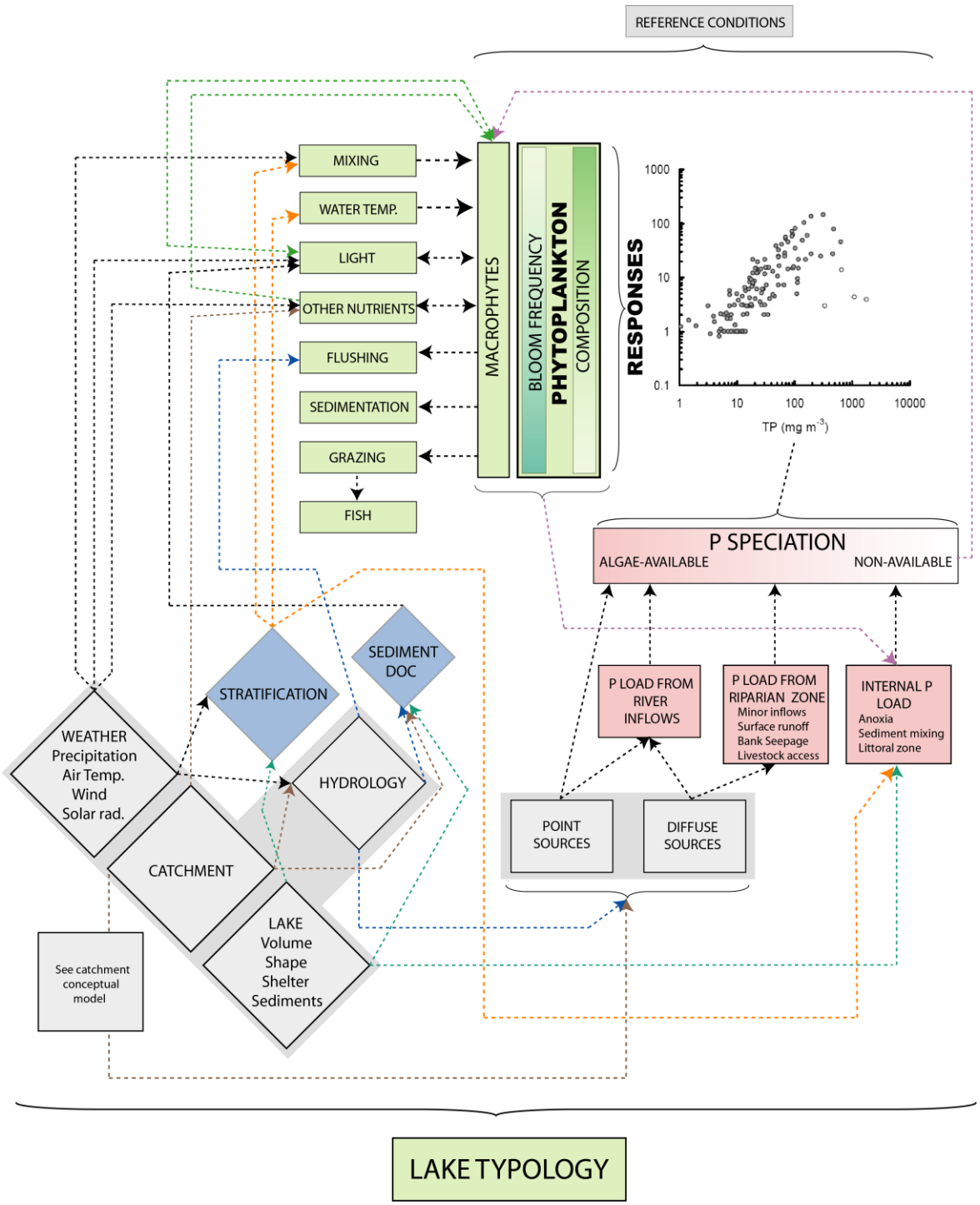


Figure 10.5 – Pre-interview phase conceptual model for lakes: grey boxes indicate quasi-fixed factors, forcing inputs or variables considered in other conceptual models

10.5 Expert weightings for chosen model components

For model components chosen by the experts, where necessary, there is a definition in the relevant sections that follow. The experts chose only to distinguish the lake typologies: shallow and deep as defined above. Composite (a combination of the 3 fuzzy distributions elicited from the experts: see Section 2) importance fuzzy weightings for the chosen components are presented in the left-hand pane of Figure 10.6 for shallow lakes and Figure 10.7 for deep lakes.

Comparison of the weightings between typologies highlights that, in general, there was a high degree of consensus between experts on both the importance and overall uncertainty of components and there were some significant differences between the weightings elicited for component between typologies:

- Internal P load, flushing, macrophytes, fish predation and zooplankton grazing are more important for shallow lakes;
- Lake mixing is more important for deep lakes.

The effect of these components upon the relationship between P and ecological indicators (and hence ecological status) is discussed in more detail below for each of the chosen components. Once their likely effect is described, particular attention is paid to the implications of these effects for designating lakes into WFD ecological status categories on the basis of P status using the two data-scenarios used here (see above). In the right-hand pane of Figures 10.6 and 10.7 the overall uncertainty associated (as composite of the 3 fuzzy distributions elicited from the experts) with each of the chosen components is presented. These distributions attach a measure of overall uncertainty to the component which can be broken down into the source of uncertainty using other fuzzy distributions elicited. Consistent with other subject areas studied under this project, the information elicited indicates that in most cases we are not limited by scientific understanding per se but by our ability to predict responses in complex, heterogeneous and stochastic systems which sparse data.

10.5.1 External P load

The discussion above on the link between lake P status and ecological impacts starting from the underlying assumption that P status is positively and causally related to mean annual phytoplankton concentration (e.g. akin to the Vollenwieder regression relationship; OECD, 1982; Vollenweider and Kerekes, 1980; Vollenweider and Kerekes, 1980). The flux of P from the catchment, from both point and diffuse sources, and the pattern of this flux over time is the primary control on in-lake P status. In some lakes, and during certain periods, internal sources of P can also be very important.

In general, the external P load tends to be more important for deep lakes as resuspension of sediments in deep lakes is likely to be of lesser importance compared to shallow lakes. Hence shallow lakes are likely to be less sensitive to external loads; this is supported by the higher expert weighting for external P load for deep lakes in Figure 10.7. For shallow lakes the external P load is, however, more important in winter but becomes less important as the summer progresses (when the internal P load may become more important): Expert 2l.

10.5.2 Internal P load

For the purposes this study, internal P load is defined as: the transfer of P from bed sediments and the hypolimnion to the photic zone. Although internal P fluxes are often more important for shallow lakes, our knowledge is greater for deep lakes (Expert 2l). Phosphorus fluxes are possible from the lake bed sediment, the littoral zone (reed beds here may be particularly important as they can become anaerobic) and from groundwater and are extremely difficult to quantify. Internal loads of P have been shown to delay the recovery of lakes where external P loads have been reduced (e.g. Jeppesen *et al*., 2005; Moss *et al*., 1997).

Much of the early literature on internal P load focuses on the release of P from sediment under anoxic conditions by the reduction of a FeOOH-phosphate complex (Golterman, 2001; Nurnberg and Peters, 1984) which requires both the P status of the sediment and the redox potential of the overlying and interstitial sediment to be known for flux predictions to be made. Anoxic release of P is what Kalff (2002) describes as the classical model of internal P cycling: a model which has more recently been shown to be too simplistic (particularly for shallow lakes). Other studies have shown pH controls of P release (Drake and Heaney, 1987) and P release under oxic conditions (Jensen and Anderson, 1992; Kilinc and Moss, 2002), particularly where the iron to P ratio is low (Bostrom *et al*., 1982).

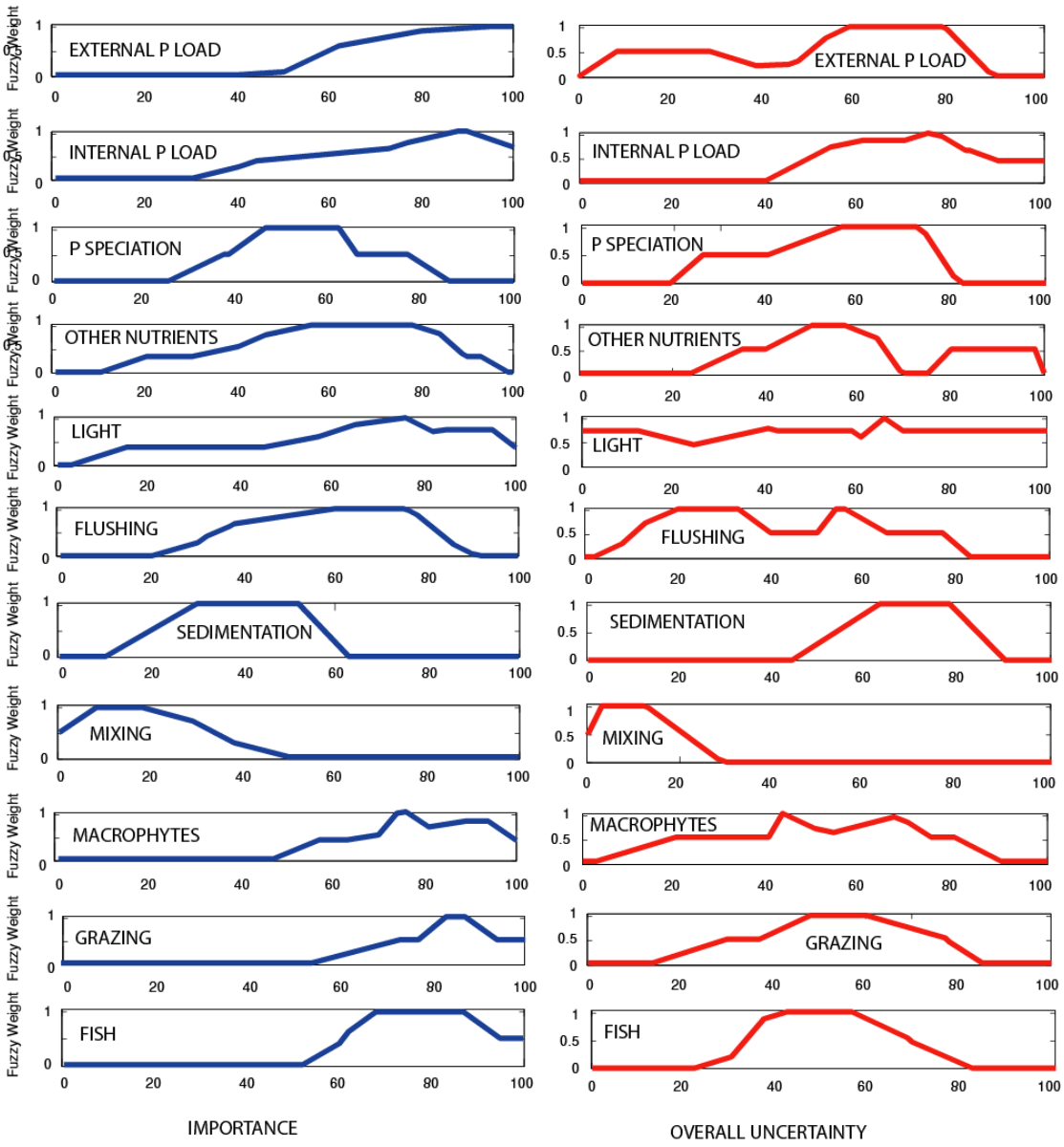


Figure 10.6. Shallow Lakes: Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts

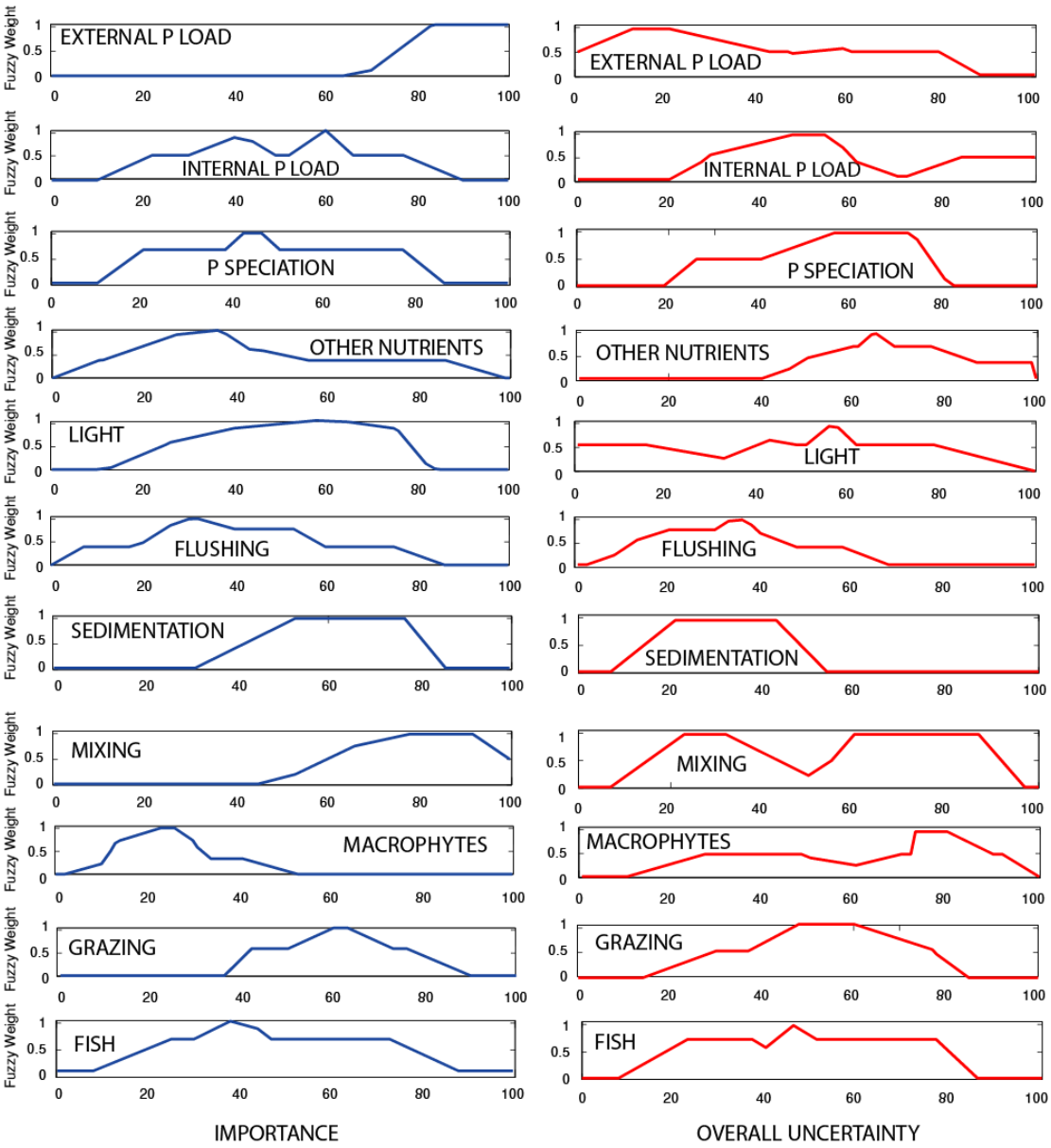


Figure 10.7. Deep Lakes: Importance and Uncertainty Ratings. In all cases the scales increase from either low importance (left-hand pane) or low overall uncertainty (right-hand pane) to high importance and overall uncertainty respectively. See Section 2 and Figure 2.3 for the detail regarding semantic anchors which guided the experts

In addition to these essentially chemical processes, there are physical and biological P release mechanisms such as physical sediment resuspension (Hamilton and Mitchell, 1997), bioturbation (Schaus and Vanni, 2000; Søndergaard *et al*., 2003) and microbially mediated release (Prairie *et al*., 2001). The processes operating in both shallow and deep lakes are temporally and spatial heterogeneous and simplistic estimation of internal P load is difficult; nevertheless, Kalff (2002) describes some general rules linked to water residence time and nutrient and organic matter loads that may be helpful in determining the likelihood of oxic or anoxic hypolimnia and likely magnitude of the sediment sink.

In terms of predicting the magnitude of internal P load for all UK lakes, the uncertainty regarding the factors affecting P is compounded, for deep lakes, by the requirement to estimate the mixing of hypo- and epi-limnetic waters as a transfer mechanism to the photic zone; this requires the thermal structure of the lake (or at least an approximation to annual average behaviour) to be estimated. In addition, It is difficult to resolve the proportion of P that comes from external and

internal sources (with which to constrain any modelling attempts) and to do so requires a large investment in data collection (e.g. see Kao *et al.*, 1998).

10.5.3 Phosphorus speciation

Phosphorus speciation refers to the importance of other P species (other than TP) with respect to bioavailability. For example, the risk misclassification based upon TP rather than a bioavailable measure. Although bioavailable P is more important than TP in driving ecological response, TP is a reasonable indicator of nutrient status for lakes. Much of the unavailable fraction of the measured TP can be made available when it passes through daphnids (Expert 2l), through fish (Persson, 1997) or by enzymes secreted by phytoplankton. The use of TP is justified to some extent by the TP-TCh-*a* relationship observed (see above) although part of this relationship is circular as TP includes biomass-derived P: in some cases this limitation can be minimised by using a measure of winter TP concentration (Expert 2l). The use of SRP is limited, as it can be too labile to be useful.

10.5.4 Other Nutrients

All three experts consulted agreed that nitrogen was the most significant nutrient, other than P, controlling ecological response by limitation or co-limitation (e.g. see Hameed *et al*., 1999; James *et al*., 2003; Moss *et al*., 1994). Nitrogen can be important in very unimpacted lakes (e.g. some UK upland lakes: Maberly *et al*., 2002), in summer for shallow lakes and even in deep lakes as the year progresses. It can also be important in lowland eutrophic lakes where P is not limiting (e.g. see Kilinc and Moss, 2002). With respect to macrophytes: there is a good inverse correlation between N and macrophyte diversity (Expert 2l: James *et al.*, 2005).

10.5.5 Light

Light is an important limiting factor; for the purposes of this study we are not considering variation of light diurnally, seasonally or with latitude, as these are relatively predictable, but the reduction of light resulting from:

- suspended and re-suspended matter e.g. sediment and humic substances (water colour);
- shading of one plant species by another;
- self-shading by phytoplankton.

Natural turbidity from suspended matter and water colour are initially catchment-derived and can be anthropogenically altered by land use practices and structures which change the natural flow regime of rivers supplying sediment and of the lake itself. These characteristics vary at a high temporal resolution in response to hydrological events but are predictable to some degree based upon catchment characteristics. In Scandinavia, colour, driven by humic substances, has been shown to be important in controlling eutrophication (Arvola, 1984; Jasser and Arvola, 2003). In the UK, Loch Ness has been shown to be lower in phytoplankton response per unit of P owing to its very peaty catchment (Jones *et al*., 1996). Competition for light plays a significant role in determining the plant community composition observed in a given lake and is one of the factors which promote the existence of stable ecological states (e.g. the extremes of phytoplankton or submerged macrophyte domination in shallow lakes) that, as discussed above, confuse the P-Cha relationship. For macrophytes, light penetration is particularly important (Spence, 1982) as light is required on the lake bottom for growth to occur: hence macrophytes are confined to the fringes of deep lakes and to very shallow lakes. In contrast, algae can move or are mixed towards the surface and do not need to be in the photic zone continually; this can be an advantage to certain taxa, so can affect algal community composition.

10.5.6 Flushing

Flushing is defined here as the horizontal flushing of phytoplankton and nutrients from a lake (a loss mechanism): primarily controlled by residence time. Flushing is more important for short residence time (shallow) lakes than long residence time (deep) lakes (c.f. Figures 10.6 and 10.7) and is explicitly taken into account in the Vollenweider model by incorporation of water volume exchanged per year and mean lake depth. Annual average and seasonal variations in residence times are relatively predictable, although there can be significant inter- and intra-year variability: nevertheless, mean and seasonal water residence times are commonly cited in the literature (e.g. George *et al*., 2007). It is also important to take into account imperfect mixing when calculating residence time (owing to thermal stratification, lake geometry, with respect to inflows and outflows, and other 'dead zones'): this is not always considered by researchers when presenting and interpreting their studies (Kalff, 2002). In general, shallower lakes tend to be more rapidly flushed and are sensitive to variations in flushing rate (this sensitivity is not simple and includes the interacting effects of nutrient supply and lake thermal structure: Jones and Elliott, 2007).

10.5.7 Sedimentation

Sedimentation is defined as: sedimentation of phosphorus-containing particles (phytoplankton cells and zooplankton faeces) through the thermocline to the hypolimnion. Sedimentation is more important for long residence time lakes than for short residence time lakes, so interacts with flushing rate.

10.5.8 Macrophytes

Macrophytes are important to the phytoplankton response and phytoplankton are important to the macrophyte response. Macrophytes are a confusing factor in the Vollenweider regression as they compete with phytoplankton for light and to a lesser extent P. As macrophytes obtain P from the sediment, competition for P is less important, but they do compete for N as both obtain N from the water. The Vollenweider model is more appropriate for deep lakes where macrophytes occur only at the periphery owing to the lack of light penetration and therefore have a reduced influence on P uptake. The extent to which deep lakes can be populated by macrophytes is hence partially controlled by lake geometry (bathymetry). Likely macrophyte coverage can be estimated from bathymetric maps (Carvalho *et al*., 2005) or, where there is no information on lake bathymetry, estimated from digital Terrain Models (DTM's) that can be extrapolated beneath the water. Once an estimate of bathymetry has been established, it is necessary to predict the possible depth range for macrophyte growth which requires that upper and lower limits are defined. The upper growth limit can be curtailed by exposure and lack of suitable sediment while the lower limit (in all but the deepest lakes where pressure and temperature may be important) is restricted by light availability (and possibly light quality in some cases). The actual macrophyte status will however be subject to many other factors: e.g. those determining whether a lake is in the clearwater or turbid state and hence will not change gradually along a eutrophication pressure gradient (Penning *et al.*, 2008). Carvalho *et al.* (2005) outline a simple method which would set an acceptable threshold phytoplankton chlorophyll *a* concentration based upon the value needed to allow enough light to penetrate to support macrophyte colonisation.

Where available, a 5 yearly macrophyte survey (although subject to seasonal and inter-year variability) would decrease the uncertainty in assessing macrophyte status significantly, compared to any estimate made without a site visit. However, in England and Wales, there are too few data on either the macrophyte communities or the phytoplankton biomass for lakes to derive direct biological impact measures for the risk assessment (Environment Agency, 2007b): thus inclusion of actual macrophyte status will be restricted to a higher tier approach.

10.5.9 Zooplankton grazing

Zooplankton grazing can remove a significant proportion of the phytoplankton biomass (Lampert *et al*., 1986) and can play a significant role in maintaining a stable, clearwater state. Where other factors affect their population (e.g. the removal of pesticide sources or introduction/decline of fish stocks: Scholten, 2005) a shift to the turbid state can occur. There is some proof from Loch Leven (Scotland) that when grazing populations were reduced by pesticides the chlorophyll *a* concentration increased dramatically (Expert 3l; May pers. Comm.). Fish predation of zooplankton can be significant in reducing grazing populations and hence in increasing phytoplankton abundance (Vanni and Layne, 1997). In general, zooplankton grazing is less important for deep lakes than for shallow lakes where significant macrophyte beds act as good refuges (Moss *et al*., 1994): the hypolimnion of deep lakes is generally not as effective a refuge for most species (although not for all, such as *Chaorborus* larvae: Expert 1l). Grazing can greatly affect phytoplankton composition as zooplankton tend to prey on phytoplankton of certain physical sizes, allowing large or colony-forming taxa to gain an advantage. Overall, Expert 3l expressed the opinion that, although phytoplankton composition is important in ecological response, abundance is more important.

10.5.10 Mixing

Wind-induced mixing is a vertical phenomenon which can control algal community composition. It is important, primarily for deep lakes, in controlling the species composition: e.g. there are normally fewer cyanobacteria in a well-mixed lake. In general, shallow lakes will be less affected as they do not stratify and so are well-mixed.

10.6 Simplified conceptual models of P risk

As the experts chose to define the lake typologies (shallow and deep) by lake depth, it is not explicitly identified as a model component and we have developed conceptual models of shallow and deep lake function. The simplified conceptual models presented in Figures 10.8 and 10.9 are hence generic, and to some degree describe processes which are important at the extremes of the spectrum of lake depths (although, as mentioned above, we are not considering lakes that behave as rivers in this section). Lakes that fall between these extremes may then take on certain proportions of extreme case function (e.g. in fuzzy set terms, a given lake could be a partial member of the shallow and deep lakes fuzzy sets depending upon its characteristics). This set-theoretic concept may be useful in determining the likely behaviour of intermediate lakes in a predictive manner.

Although reduced in complexity from Figure 10.5 the simplistic representations of shallow and deep lake function may still be too complex for WFD purposes, and in particular, where estimations are to be made in data-poor situations (e.g. using the Tier 1 data scenario above of nationally available data). Where higher tier data are available (see definition above), the observations of current ecological status would partly override the need to make predictions on the basis of other model components. Perhaps the best way to use the conceptual models developed here is to test whether or not their components can explain the variation observed in the P-Ch*a* relationship (Figure 10.4). If this exercise was performed it may be possible to refine further the conceptual models: this may be possible *a priori* to a degree as there will be inadequate data for some chosen model components (e.g. zooplankton grazing). Explanation of the variability observed may be possible using statistical regression techniques or perhaps, owing to the high levels of uncertainty associated with some of the variables to be tested (e.g. an effective residence time), it may be more appropriate to use fuzzy techniques such as: fuzzy cluster analysis or fuzzy regression.

This type of approach, considering other factors as model components or as typological descriptors, is supported by Carvalho *et al*. (2005) who suggest nitrogen, light and residence times as factors which may be important in determining lake response. Carvalho *et al*. (2005) go on to suggest that acceptable levels of phytoplankton biomass could be expressed with

regard to their knock on effect to macrophytes which would implicitly include an element of plant community composition.

10.7 Acceptable lake P status for lakes

At a Tier 1 level (i.e. where no lake-specific information is available) the approach of the EA for assessing lakes assumes a direct link between lake P status and ecological status, as an Ecological Quality Ratio (EQR) is calculated using the ratio between reference and current P status (Environment Agency, 2007b; see also Figure 10.10). Lakes are subsequently assigned to a WFD class based on this EQR (Table 10.4). The reference condition for P is estimated using catchment geology and lake depth: by using the linkage between naturally available P and lake base status (the Morpho-edphic Index: Vighi and Chiaudani, 1985): and that deeper lakes tend to retain a larger proportion of the P load compared to shallow lakes. Importantly, this type of approach implies that an elevated P status is sufficient for designation of failure to meet given WFD standards in the absence of lake-specific ecological information.

In terms of uncertainty, this assumption may lead to a given lake being designated at risk when ecological status may suggest otherwise, and vice-versa. This, however, is unavoidable in the absence of more specific relationships between lake characteristics and P status and is taken into account to some degree with the assignment of confidence rankings to each designation (Environment Agency, 2007b).

The WFD is precautionary in as much as the worst of all indicators is used as the waterbody designation: P status is, however, meant to be only a supporting element (Expert 1l; Carvalho *et al*., 2005). Expert 2l has major concerns with the way that the EA are determining ecological status with solely phytoplankton (or a P concentration proxy). For example, Expert 2l highlighted that this means that, for some typologies, a 'lifeless swimming pool' would achieve good ecological status. What really should be done is to look at the functionality of the system in a more holistic manner– it should have:

1. parsimony of nutrients;
2. characteristic structure (e.g. macrophyte dominance for certain types);
3. connectivity (e.g. no dams stopping the migration of salmonids.

Table 10.4 – Ranges of EQR values assigned to each WFD classification (simplified from Environment Agency, 2007b)

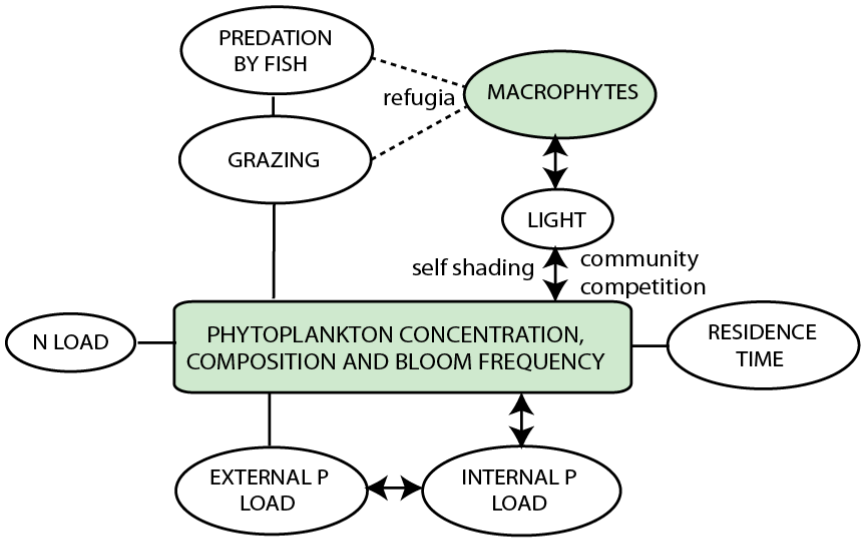
EQR	WFD STATUS
> 0.8	High
> 0.5 ≤ 0.8	Good
> 0.33 ≤ 0.5	Moderate
≤ 0.33	Poor or worse

The estimation of reference state is also a source of uncertainty. Expert 2l suggested that, as there are no pristine lakes in the UK, the reference conditions will not be pristine (they will be a slightly impacted state) and would like to see greater honesty about this fact in regulatory work. Carvalho *et al*. (2005) state that diatom-inferred TP is the most direct indicator available of past TP concentrations in a lake, but tends to over-estimate at low concentrations and under-estimate at high concentrations.

Expert 3l highlighted the fact that algal bloom frequency (and intensity) is required by the WFD; this is not covered by annual average P or phytoplankton concentration and would require very high-resolution data or modelling. In a similar manner, Instead of an annual average measure, Reynolds (1978) used a P-maximum Cha concentration which may be useful as the general public tend to remember extreme events (Jones *et al*., 1979).

SHALLOW LAKES

TIER 2



TIER 1

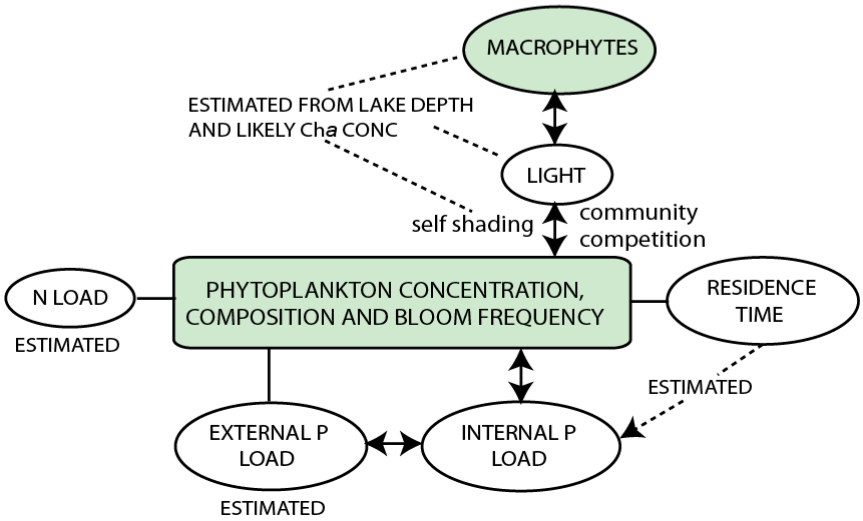


Figure 10.8 – Simplified conceptual model for shallow lakes including the primary components that determine the ecological response to changes in P status. Tiers 1 and 2 refer to the two different data scenarios Nationally available data and derived (modelled) datasets and higher resolution lake-specific observations (see definition above)

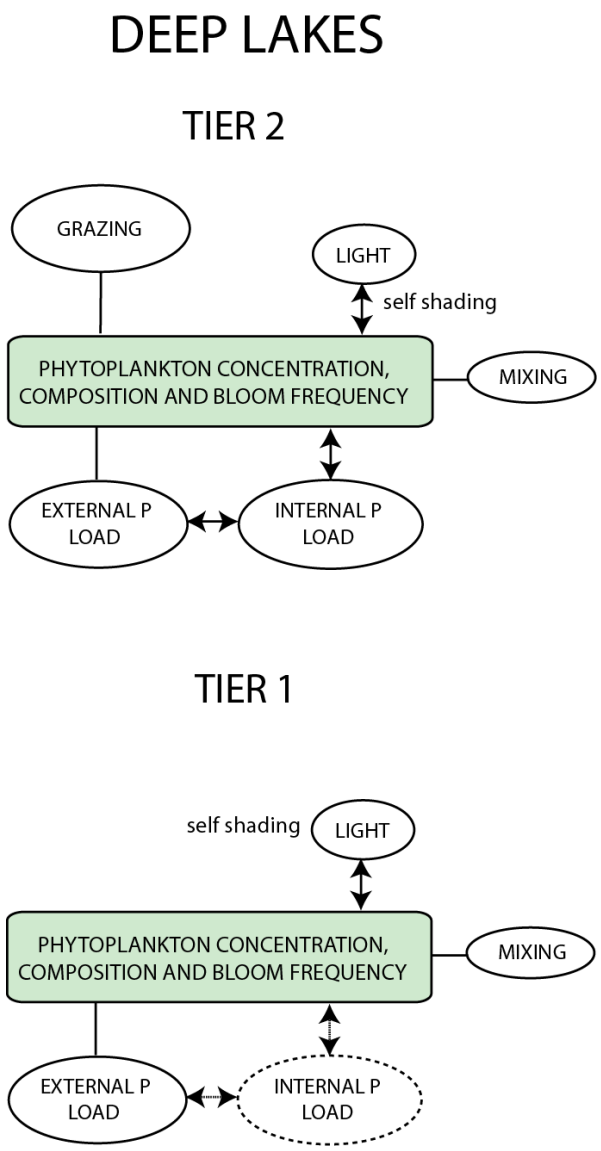


Figure 10.9 – Simplified conceptual model for deep lakes including the primary components that determine the ecological response to changes in P status. Tiers 1 and 2 refer to the two different data scenarios Nationally available data and derived (modelled) datasets and Higher resolution lake-specific observations (see definition above)

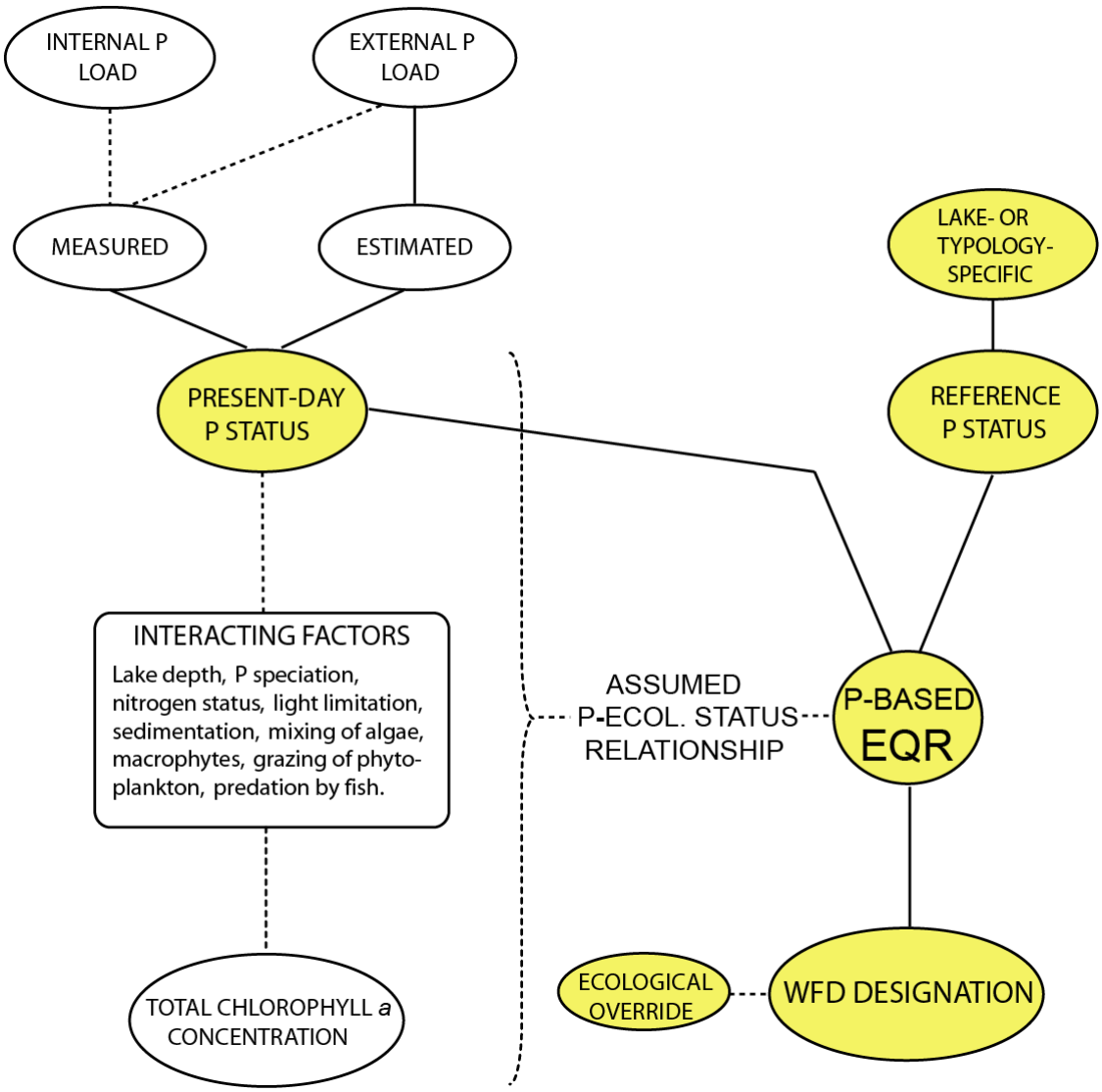


Figure 10.10 – A simplified schematic representation of the EA's assessment of P risk for lakes (highlighted in yellow) showing that the risk from P is related to a reference P status: thus assuming a direct relationship between P and ecological status

10.8 Research Priorities

Research priorities that are implicit from the expert elicited fuzzy distributions (Figures 10.6 and 10.7 above) are those components which were rated as highly important and highly uncertain. This is expressed simplistically in Figure 10.10 by a weighting calculated from the product of the ratings (the centroid of each fuzzy distribution) for each chosen component. The figure shows that there are higher research priorities for shallow lakes for most components with the main priority being the role of internal P load. For risk assessment purposes, proxy measures of the internal P load, indicative of the evidence of its significance for lake eutrophication risk, would be needed. For deep lakes, mixing of hypo- and epi-limnetic (and its control on redistribution of biota and nutrients) waters is identified as the main research priority.

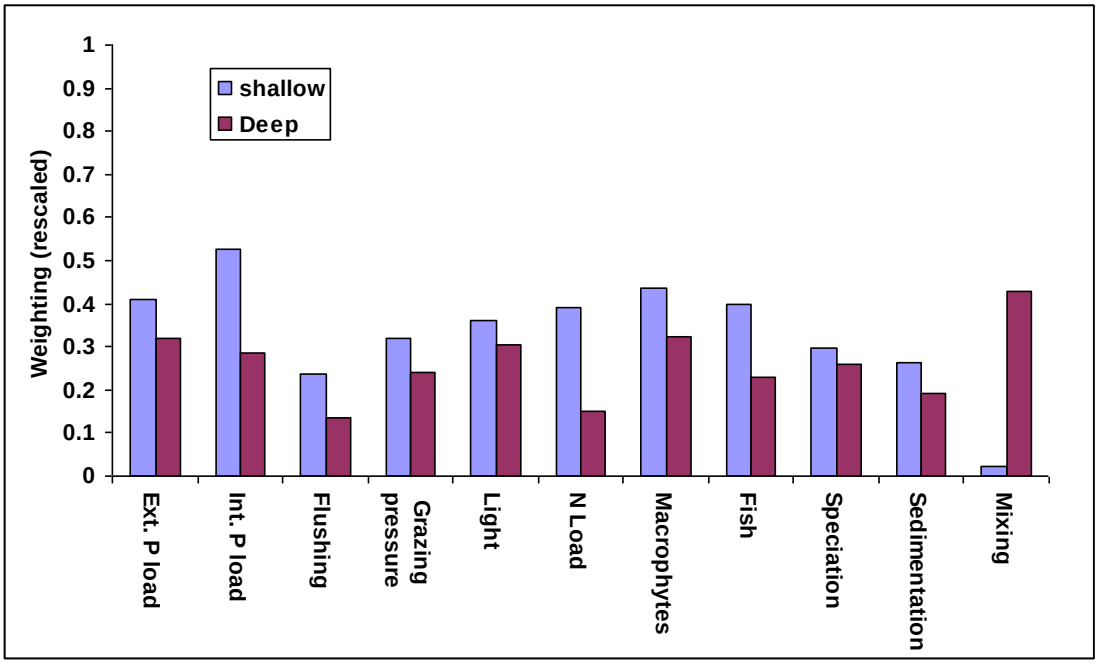


Figure 10.10 – Ratings for the lake model components with regard to research priority estimated using the product of the centriod of the importance and overall uncertainty fuzzy distributions

10.9 Conclusions

Estimating the effect of elevated P levels on lake ecology is not simple owing to the complex, dynamic and stochastic nature of lake systems. Any simplistic relationship between P and ecological status is confounded by interactions between various factors such as growth limitation by other nutrients, light and food web interactions which allow lakes to shift to alternate stable ecological states for a given P status. Use of simplistic P-ecological status dose-response relationships hence leads to significant uncertainties in WFD designations. These uncertainties could be reduced by better monitoring of a structured sample of lakes and very significantly for those lakes included within the sample for which lake-specific information is available. Given these problems, and similar to those identified in Section 8, it may be more realistic to use a holistic approach to determining lake ecological status and perhaps using a regime-based representations that allow for natural system variability. Risk analyses methodologies are discussed further in Section 11. The simplified conceptual models for shallow and deep lakes developed during the elicitation (Figures 10.8 and 10.9) will help focus future research and data collection to reduce risk analyses uncertainties.

11 Risk Analysis and Modelling for the Water Framework Directive

11.1 Summary

Estimating the likely risk to the ecological status of surface waters from elevated P levels is not simple. The link between the two is complex and involves many other factors, confounding the definition and use of simplistic dose-response relationships in most situations. Thus, it is somewhat artificial to consider the risk to ecology from elevated P levels in isolation, although this does not mean that low levels of P are not desirable. Under the WFD ecological status is expressed relative to a reference state which is difficult to estimate for individual waterbodies/waterbody types (particularly as no reference sites exist in the UK) and will not be associated with a known reference P status owing to system variability. Although problematic, the definition of ecological status and reference conditions is crucial and should be done in a holistic manner using a number of relevant variables combined in such a way to give robust designations. Only when this has been satisfactorily done can the estimation of current ecological status be carried out using a consistent approach. A number of the experts consulted expressed the opinion that the natural variability of ecological systems and the uncertainties associated with our risk analyses estimates makes the use of simplistic threshold-based water quality standards problematic and suggested that more flexible and holistic approaches are required. Poole *et al*. (2004) suggest the use of regime-based water quality standards which describe the system of interest in terms of valid distributions of variables in space and time. This type of approach could be used for EA WFD risk analyses using distributions of variables describing pressures on the system and factors which may make the system more sensitive or more resilient.

The experts' suggestions for improvement of WFD risk analyses are summarised by the following points:

- Good ecological status must be clearly defined.
    - Regulating bodies need to be transparent and honest about the definition and meaning of reference conditions;

- The link between ecological status and elevated P levels is complex and this should be allowed for.
    - Any risk analysis considering the risk to ecological status should be holistic and consider the primary factors that determine ecological status, rather than P pressure alone;
    - Ecological quality should be assessed using a robust suite of variables rather than simplistic dose-response relationships;
    - Risk analyses should take into account the natural variability of ecological systems with respect to present-day and reference conditions: consideration of how pressures and system sensitivity vary in time and space should be incorporated;
    - Environmental standards which take account of natural spatial and temporal heterogeneity of ecological function within lakes may be useful;
- Where information is known to be highly uncertain and errors are difficult to quantify, indices rather than an absolute measures may be more appropriate.
- The type of risk analysis and method for assessing uncertainty should be tailored to the specific question, the particular catchment/waterbody and the level of available information.

- Where possible uncertainties in our estimates should be made explicit for robust decision and policy making.
- Effective communication of uncertainties, and the sources of the uncertainties, is important.

11.2 Introduction

The WFD requires an analysis of ecological status of all waterbodies using biological indicators which may be supported by physical and chemical metrics; supporting metrics may be used to help explain failures and in the planning of mitigation programmes (Evers *et al*., 2007). Under the WFD, ecological quality is measured with respect to 'pristine' reference conditions from which waterbodies are required to deviate only 'slightly' to achieve good ecological status. Thus, for WFD purposes, estimates of 'current' and reference conditions are required for each water body (or water body type) as is a methodology for their designation into ecological status classes (i.e. bad, poor, moderate, good and high). The reference conditions, and variables used to describe them, define high ecological status and increasing degrees of degradation describe lower designations.

Ecological status is not simple to define and biological, hydromorphological and physicochemical conditions must be established. However, under the WFD ecological status must be quantified by calculation of an Ecological Quality Ratio (EQR: a ratio expressing current conditions compared to reference conditions). The use of EQRs is thought by some to be too simplistic (Expert2l; Expert 3m) as determination of definable ecological states for water body ecotypes can be confounded by natural variability including phenomena such as alternative stable states (e.g. see Moss, 2007). The move away from chemical measures to ecological indicators for determination of ecological quality is positive and significant but care must be taken that there is no reversion to measuring water quality parameters rather than the health of aquatic environments (Expert 2l; Expert 3m; Hatton-Ellis, 2008). However, where data are sparse the use of chemical measures as proxies is likely occur to some degree and is common in the type of dose-response relationships currently being employed.

Where water body-specific information is available, 'current' ecological status can be estimated directly. Even so, this requires applying some form of ecological status assessment model to interpret the data from each lake. Where no specific information is available current ecological status must be estimated by extrapolation, using 'modelling' approaches. Estimation of likely ecological status from given site characteristics and estimates of pressures implies that causal relationships and interactions between processes are known (see Sections 9 and 10 for a discussion). Herein lies the origin of much uncertainty as simplistic dose-response type relationships can be unclear in complex, dynamic ecosystems which are subject to significant natural variability and stochasticity (Expert 2l; Exert 3m; Rapport *et al*., 1998; Rapport and Whitford, 1999; Bohn and Kershner, 2002; Hatton-Ellis, 2008; Leuven and Poudevigne, 2002; Moss, 2007; Palmer *et al*., 2005; Poole *et al*., 2004). Simple dose-response relationships for P may not exist and are therefore may be an unachievable regulatory goal (particularly where the derivation of reference conditions is also affected). However, Leuven and Poudevigne (2002) argue that reductionist dose-response approaches (often favoured by regulatory authorities in regulated countries: Gilman *et al*., 2004) are of prime importance for risk assessments and complementary to more holistic approaches that consider whole ecosystems. Leuven and Poudevigne (2002) do, however, agree with the experts' views (Experts 3m and 2l) that more holistic approaches are of greater relevance and are generally preferable but have the disadvantage that data requirements may be high and subsequently may require an excessive amount of time to carry out such that they are only suitable for higher tier approaches (Expert 2m).

Given system complexities, it is clear that considering the effect of elevated P concentrations on ecological status in isolation to other controlling factors is somewhat artificial. However, since there is some consensus that waterbodies should be parsimonious with respect to phosphorus (Experts 3m and 2l; Moss *et al*., 2003; Moss, 2007) it seems logical that some targets, or target distributions for water body P status should be set as a precautionary measure, particularly where site-specific measures of ecological status' are not available.

A general form of risk analysis based upon WFD requirements is set out schematically in Figure 11.1 (see also Carvalho *et al*., 2005; Bennion *et al*., 2005; EA 2007 and Section 10). The schematic is general enough to show how information may be combined to estimate risk; the information used for any part of the analysis may be quantitative or qualitative (or a combination of both) and in both cases will be subject to some degree of uncertainty which should be taken into account in risk analysis methodologies. Where large uncertainties exist *absolute* quantitative estimates are difficult to make but are called for under the WFD (as, ideally, an EQR should be absolute); the WFD also requires that levels of confidence and precision are required to be specified within River Basin Management Plans (European Commission, 2000: ANNEX V, 1.3). The experts consulted for the modelling subject area of this study thought that, given the likely level of uncertainty associated with WFD risk analyses (especially national scale analyses), estimates made should be recognised as guides or indices for informing decisions, rather than precise quantitative estimates. For example, the available national risk analysis tools are best used to provide 'population statistics', such as the proportion of lakes at risk in a given area, rather than make assessments for individual water bodies (Expert 2m).

A further requirement for implementing the WFD, which is driven by the need to restore non-conforming waterbodies to good ecological status by 2015, is the attribution of the causes of reduced ecological status to help plan mitigation programmes (e.g. identification of the relative risk posed by point and diffuse sources of P or identification of the primary drivers of P fluxes to surface waters within a given area) which will also require estimates of the likely time for improvements to manifest themselves. These types of analyses may require more focussed methodologies, and field observations, to be able to make reasonable estimates that attribute risk.

In the sections that follow general methodologies to estimate P pressures and 'current' and reference ecological status of rivers and lakes are discussed. Where relevant, there is a distinction between methodology structures where site-specific information exists and where it does not. The different methodologies are sometimes referred to as Tiered approaches (e.g. see Carvalho *et al.,* 2005; Heathwaite *et al*., 2005a). A Tier 1 approach (sometimes described as risk screening) is likely to use 'national scale information' and give estimates of high risk areas at large scale whereas higher tiered approaches are likely to be able to provide estimates for more detailed studies at specific locations, where site-specific information and more detailed modelling may be used, or will be able to carry out detailed 'what-if' scenarios, examining changes of risk and behaviour within a system (Expert 2m).

A full discussion of risk analysis/model predictive capability is beyond the scope of this report, what follows is a brief summary of approaches that may be appropriate for WFD risk analyses where data are normally sparse and uncertainties potentially significant.

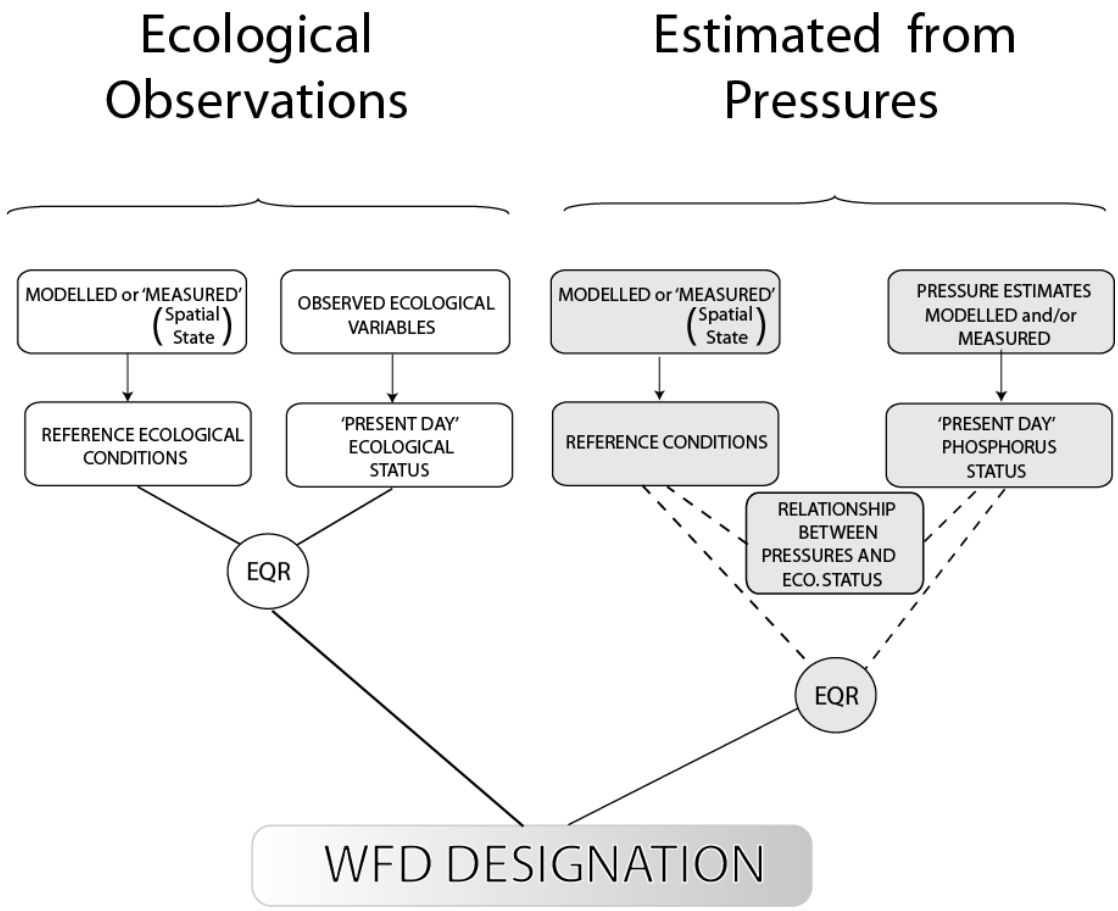


Figure 11.1 - Generic schematic representation of the requirements for WFD risk analysis for surface waters. The left-hand side of the schematic shows a 'direct' assessment of current ecological status where observations are available via a comparison with estimated reference conditions. The right-hand side shows an indirect assessment of ecological status where no observations of ecological variables are available and WFD designation must be made using modelled and/or measured estimates of pressures on the waterbody

11.3 The risk to ecology from elevated phosphorus levels

In Sections 9 and 10 the relationships between elevated P levels and ecological status for rivers and lakes is discussed and it is highlighted that relationships are complex and involve many other biological, geochemical and physical factors. As a result of these complications, any P status targets set using P-biological indicator relationships will implicitly include the effects of other factors. Thus there is a requirement to estimate risk from uncertain estimates of P pressure and uncertain relationships that define water body sensitivity to elevated P levels. Under such conditions, the experts consulted within the ecological areas of this study recommend that:

- Good ecological status must be clearly defined.
- Risk analyses should be holistic and include the combination of a suite of variables, rather than single or simplistic dose-response relationships.
- Risk analyses should take into account the natural variability and stochasticity of ecological systems.

11.4 Risk analysis spatial and temporal resolution

The relevance of any risk assessment is dependent on the spatial and temporal attributes of the pressure and the sensitivity of the responding ecosystem components (Rapport *et al*., 1998; Leuven and Poudevigne, 2002). Phosphorus pressure and water body sensitivity both vary in time and space (Expert 1m; Edwards and Withers, 2007) such that risk analyses resolution ultimately has an effect on WFD designation (Bohn and Kershner, 2002). Thus it is possible that an analysis may need to consider multiple timescales (e.g. the estimation of both seasonal and annual average conditions depending upon the receptor) and include likely recovery response times where water bodies require restoration to meet good status.

To a certain degree, the spatial scales of WFD risk analyses are determined by the type and size of water body and its catchment location. For EA purposes WFD catchments have been identified and are generally in the range 10-100 km^2 with associated river reaches of approximately hundreds of metres to tens of kilometres. Thus, at these scales, it is almost certainly a requisite that estimates of risk arising from multiple sectors (i.e. urban, rural, point, diffuse) will be needed, increasing the uncertainty and the political significance attributed to the model results (especially the apportionment of risk to a given sector). If a risk analysis is to inform decisions regarding reduction of P pressure, WFD catchment scales may not be appropriate and estimates may need to be made at higher resolution to inform policy on best agricultural practice. For example, Withers and Lord (2002) introduce a concept where boundaries of acceptable and unacceptable field management might be set for a given intensity of land use within a catchment, taking account of the intrinsic vulnerability of the landscape. The need for higher resolution estimates was supported by a number of experts who commented that decisions are often made at farm or field scale (see also the discussion of Magette *et al*., 2007).

11.5 Estimating Reference conditions

Good ecological status is defined with respect to reference conditions which need to be consistent with the measures of 'current' conditions for an EQR to be calculated. As there are few, or no, pristine reference sites in heavily populated countries (Carvalho *et al*., 2005; Hatton-Ellis, 2008; Expert 2l) reference conditions must be estimated. A lack of reference sites also means that it is difficult to assess the natural variability associated with reference conditions (Hatton-Ellis, 2008), such as natural shifts between stable ecological states and the effects of natural extremes . Given these difficulties, rather than attempt to recreate unachievable or even unknown historical conditions, some argue that we should seek to achieve the least degraded state possible given the physical and social pressures within a catchment (Palmer *et al*., 2005) or at least to be transparent about the fact that our reference conditions may not be truly pristine (Expert 2l). This type of thinking is embodied in the WFD to some degree under the definitions of infeasibility or disproportionate expense of environmental objectives (European Commission, 2000). Reference conditions are hence general guides and likely to be uncertain, particularly for individual waterbodies. Dodds (2006) and Dodds and Cole (2007) reported a pragmatic approach to setting 'relatively pristine' conditions by creating frequency distributions of determinands (P, N and Ch*a* for US streams) to represent 'likely reference conditions' in a similar way to the spatial-state approach described by Carvalho *et al.* (2005). In the UK, where no pristine reference sites exist, only indirect estimates of chemical and biological reference conditions have been made as follows:

Lake Ch*a* reference concentrations have been estimated using (see Section 10):

- Lake type-specific TP-ch*a* regression models;
- Export coefficient models and OECD TP-Ch*a* regression equations.

Lake TP reference concentrations have been estimated using:

- The Morphoedaphic index method;
- Diatom-inferred methods;
- Export coefficient models.

Riverine SRP reference conditions have been estimated using (see Section 9):

- The balance between nutrient-tolerant and nutrient-intolerant benthic diatom taxa and associated SRP concentrations (Duncan *et al*., 2006);
- Macrophyte community response to trophic status using the LEAFPACS Method of Willby *et al.* (2006): see also Penning *et al*. (2008).

The reference conditions defined using these methodologies are subject to significant uncertainties which are taken into account to some degree by the breadth of the acceptable ranges for each of the WFD designation classes (Sections 9 and 10). These class boundaries are, however, discrete which increases the chances of misclassification for marginal waterbodies, particularly where uncertainties originate from biases.

11.6 Estimating 'current' conditions

Current conditions are required for comparison with reference conditions in estimation of an EQR. Current conditions may relate to pressures on a water body or observed ecological measures, or both, and is ultimately determined by the poorer of the two as a precaution (European Commission, 2000).

P pressure is one of many pressures on surface waterbodies and can be highly spatially and temporally variable, hence the way in which P pressure is expressed (or defined) can have an effect upon the determination of current status. Various methods for estimating P pressure are discussed below.

Where site-specific ecological information is available a direct (but still uncertain) measure of present day status can be made. The assessment is direct in the sense that it is a measure of ecological status, independent of any measures of pressures on the water body. In the strictest sense, a water body observed to be in a good ecological state may still be designated otherwise owing to other pressures (including high P exposure) (i.e. whilst there is a risk of future degradation – due to high pressures – even a site presently at good status can be designated as high risk). In this situation the level of confidence associated with estimates is crucial and will be controlled by the type, quality and quantity of information available and on the operational definition of good ecological status.

Where no site-specific ecological information is available, ecological status must be inferred from estimates of all significant pressures. The risk from P pressure is, in some cases, likely to be independent of ecological status per se and should be estimated using an understanding of the system in question. From the responses gained from the experts, what is required in our estimates is explicit recognition of the complex, dynamic and stochastic nature of ecological systems in their response to given pressures. This variability needs to be incorporated into any risk analysis structure for robust estimates to be made together with a measure of confidence. Thus, for waterbodies without site-specific information, a holistic estimate of likely ecological status should be made using multiple variables (not solely P) that may be estimated from nationally available information where possible. There are a number of holistic methodologies suggested by the experts, and published in the scientific literature, which are outlined below.

11.7 Holistic determination of present-day ecological status

11.7.1 Lakes

Moss (2007) presents a system for the classification of shallow lakes whereby a suite of variables is used in the determination of ecological status. The system uses the overall concept of three primary drivers: nutrient parsimony, characteristic structure and connectance of hydrological system. These are disaggregated to the variables: pH, TP, TN, macrophyte abundance and composition, Ch*a* concentration or Secchi transparency, frequency of cyanobacterial blooms, absence of invasive species, intactness of flow regime and connectivity of the hydrological system to the ocean. Many of these variables are difficult or impossible to estimate without visiting the site in question perhaps making it most useful for higher tier approaches. This system does highlight the variables required for a holistic analysis and is supported by other authors calling for holistic metrics to be employed (e.g. Hatton-Ellis, 2008). The variables in the Moss (2007) system are broadly consistent with the simplified conceptual model for shallow lakes developed in Section 10 and extend it to include connectivity and intactness of the hydrological system. However, Moss (2007) warned that it is not easy to assign numbers to the criteria used in his scheme and a consistent definition of good ecological status is first required.

Reynolds and Maberly (2002) present a simple method for determining the constraint on production of Ch*a*. The method estimates whether waterbodies under pressure from elevated P concentrations will be sensitive to changes in P concentrations by using stoichiometry to calculate the supportive capacity of P, N, silicon and light to identify which is the limiting factor. The methodology also requires estimates of lake retention time, light extinction coefficients for algae and suspended particles/colour. Similarly, Reynolds (2003) presents a scheme whereby an index of lake sensitivity to phosphorus can be generated by combination of criteria of current lake P status, lake retention time, bicarbonate alkalinity and proportion of lake depth less than 5m. An additional criterion of lake importance (with respect to conservation value or other social or political criteria) was also included in the sensitivity index. The index indicates the likely sensitivity of a lake to change if external P fluxes were regulated. For the variables used:

- P status is described by the average TP concentration and a measure of months per year where MRP is below a value where limitation of phytoplankton growth is likely to occur. Hence sensitivity is more likely for oligotrophic lakes.
- Lake retention time is used as a measure of lake flushing rate and its regulation of biomass and removal of unused P remaining in the water column. Long retention time lakes with net sedimentary storage are likely to be the most sensitive to load changes and short retention time lakes the least sensitive.
- Alkalinity and the proportion of lake depth less than 5 m are included as measures of likely pressure from internal P recycling: desorption from lake sediments and resuspension of bed sediment. Thus low alkalinity lakes (where lower internal loads are likely) and deep lakes (where there is a lower likelihood of sediment resuspension) are likely to be more sensitive to changes in external P load.

For both the Reynolds and Maberly (2002) and Reynolds (2003) methodologies to be able to determine status relative to reference conditions: reference conditions must thus be derived using the same indices to allow calculation of an EQR. These types of metabolic approach have been recommended as areas for future development by Carvalho *et al*. (2005) and Dodds and Cole (2007).

Carvalho *et al*. (2005) also describe a framework for defining a phytoplankton classification tool that is aimed at predicting the phytoplankton functional types associated with types of water body in different seasons and under differing nutrient regimes; this methodology would require a significant amount of data and is more suitable for higher tier approaches. Other higher tier approaches may include lake-specific modelling approaches where phytoplankton are modelled dynamically (e.g. using the PROTECH model: Reynolds *et al.,* 2001) such that other important phenomena such as bloom frequency (which is requirement of the WFD: Solheim *et al.,* 2008) can be investigated as well as remediation scenarios.

All of the approaches discussed in this section fit the criteria set out in Section 10 to be more holistic than simplistic dose-response relationships and share similarities with the conceptual models identified. The remaining criteria to be fulfilled would include regime-based measures of

the variables that would account for natural variability and to explicitly include uncertainties (e.g. see the approach of Arhonditsis *et al*., 2007).

11.7.2 Rivers

For rivers there are fewer studies aimed at identifying methodologies to assess ecological status under the WFD. The current EA WFD technical risk assessment for rivers primarily considers P pressures from land and considers ecological sensitivity by using P thresholds set for different river typologies (Environment Agency, 2008). The thresholds used were those defined by the UKTAG (Guthrie *et al*., 2006; Section 9) based upon the composition of benthic diatom species. Similarly, P thresholds based upon macrophyte communities and their response to eutrophication are being investigated using the LEAFPACS Methodology (Willby *et al*., 2006; Penning *et al*., 2008; Section 9). These methods seek to identify dose-response type relationships based upon the assumption that predictable species shifts occur with increasing P enrichment (Solheim *et al*., 2008). As for lakes, some expert opinion indicates that a methodology is required that is holistic, takes into account natural variability and provides chemical and biological targets for regulatory bodies as distributions rather than thresholds (Palmer *et al*., 2005; Bohn and Kershner, 2002; Poole *et al*., 2004). Bohn and Kershner (2002) promote a holistic *Watershed Analysis Procedure* as having many advantages as it provides a systematic characterisation of the physical and biological catchment processes and describes their spatial distribution, history, and linkages at the appropriate 'ecosystem scale'.

Holistic schemes are presented as being more robust and as allowing a higher level of confidence in designations (Poole *et al*., 2004) as it is unlikely, given normal data constraints, that a high level of statistical certainty can be achieved for individual measures of biological quality (Hatton-Ellis, 2008). Perhaps one of the most promising approaches at a Tier 1 level is the habitat matrix conceptual model for stream periphyton reported by Biggs *et al*. (1998) which uses the variables: flood disturbance, nutrient resource supply and invertebrate grazing (see Section 9). This type of approach could be adapted to simulate responses of the aquatic plant community considered and extended to incorporate the regime-based approach of Poole *et al.* (2004). If this was to be attempted, it would be necessary to collect data on the matrix variables for river typologies as a research priority. It would also be necessary to define reference conditions consistent with the estimates made from the habitat matrix model: e.g. by producing distributions of periphyton biomass given reference flows, nutrient status and grazing pressure.

Higher tiered approaches may utilise more complex modelling approaches such as the instream module of INCA-P (Wade *et al*., 2004) if sufficient data is available for model evaluation; e.g. it has been demonstrated that, using the INCA model, equifinality in model structures and parameters resulting partially from a lack of data makes it difficult to separate the role of individual processes (Wade *et al*., in press; Dean *et al*., in press).

11.8 Estimating phosphorus pressure

Where water body-specific P pressure observations are made, they may replace the need for modelled estimates, at least for determination of 'present-day' conditions. This may only be the case where high resolution data is available as low resolution data may lead to biased P pressure estimates. Where bias is likely, some form of correction may be required such that P flux or concentration measures for defined periods are obtained. It is important to note that the form of P measures (e.g. TP or a measure of bioavailable P) may be critical for the phosphorus risk analysis (Expert 2m)

Where no water body-specific observations are available modelled estimates will be required and must be made on the basis of catchment-specific information such as the variables discussed in Section 3. The confidence associated with such estimates primarily depends upon the quality and quantity of data available for model inputs, model evaluation and the methodology used to quantify associated uncertainties (together with the experience and

expertise of the modelling team: Expert 2m). In the case of estimating P pressure at national scales, the evaluation data available (i.e. GQA data) includes P from all sources upstream of the point of measurement, weakening the constraining power of the data considerably and correspondingly increasing uncertainty (see below for a more detailed discussion of GQA data).

11.8.1 Methods for estimating phosphorus pressure

Conceptual models of point and diffuse P pressures on receiving waterbodies are presented in Sections 3, 4 and 5. These conceptual models include components chosen by the experts as critical in driving P fluxes at the spatial and temporal scales of interest here. In some cases, however, the scientific evidence to support the inclusion of the components was shown to be derived from small-scale studies that do not necessarily help us understand their effect at larger scales where the magnitude of effect of different controlling phenomena may differ (Addiscott, 1998a; Beven *et al*., 2005).

Figure 11.2 shows a histogram of the scales of studies which experts from the catchment processes and agricultural practices subject areas used as evidence to support their opinions where it can be seen that there are fewer large scale studies, presumably owing to the need for detailed studies for gaining understanding of individual processes. A review of various P transfer studies at various scales by Dougherty *et al*. (2004) concluded that the findings of some studies would have had more value if scale had been considered in the design or interpretation of the experiments. Similar issues are central to the discussion of sediment and P delivery to streams by Beven *et al*. (2005) who describe the difficulties in extrapolating from small scale studies. These difficulties arise from the fact that not all of the sediment or P mobilised at small scales will reach the water body, as redeposition and readsorption may take place during transport (Walling, 1983; Beven *et al*., 2005). Other processes also become more important as scale increases such as the effects of macropores and artificial drainage (Expert 3c) or storage and release of P from stream bed sediments where the scale increases to include waterways (de Vente *et al*., 2007: Section 6). The importance of P delivery is supported by the choice of conceptual model components identified as critical for risk analyses (i.e. hydrological connectivity, soil type and topography - see Figures 3.4 and 3.5, Section 3.

Additional problems arise in scaling up from theories derived at small scales when averaging across space and time. This is a particular problem when models use representations of complex non-linear processes as there is an inconsistency between the average of a heterogeneous nonlinear process and a similar nonlinear representation of the average heterogeneous behaviour (Addiscott and Mirza, 1998b; Neal, 1996). Thus for small scale theories to be used, a scale dependent approach will be required (Addiscott, 1998; Addiscott and Mirza, 1998a; Addiscott and Tuck, 2001). Scale-dependent approaches vary. They can be corrective scale-dependent rules such as those derived by statistical analysis (e.g. Prairie and Kalff, 1986 showed that TP flux was related to (catchment area)$^{0.77}$), the use of effective parameter values (see the discussion of Beven, 1995) or by approaches which set out to identify scale-dependent rules (Addiscott, 1998; Soulsby *et al*., 2006; Tetzlaff *et al*., 2007). For example, the PEDAL approach[ℵ] uses delivery coefficients determined by measurement at the scale of interest. These scaling problems are not restricted to hydrological and geochemical sciences, the same issues occur for biological systems (Standing *et al*., 2007).

[ℵ] **P**hosphorus **E**xport and **D**elivery from **A**gricultural **L**and - Defra project PE0113

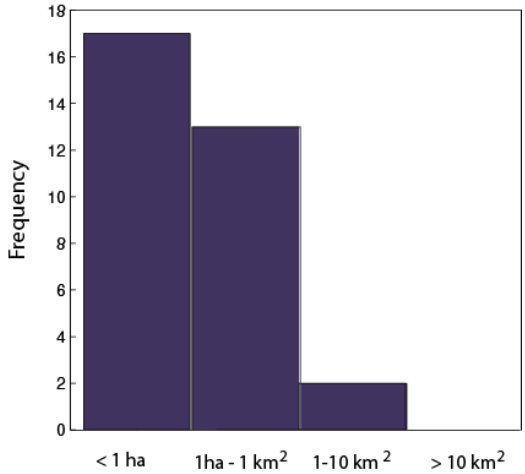


Figure 11.2 – Histograms showing experimental scales from selected papers identified by the experts from the Catchment Processes and Agronomic Practices sub areas as those that contributed to their opinions

Approaches which use understanding gained at smaller scales are often process-based models, frequently with an element of spatial distribution and may be aimed at simulating the effects of agricultural management strategies. Expert 2m was of the opinion that some elements of these types of model (e.g. system inputs, and mobilisation processes such as soil erosion) may be calibrated at smaller scales (e.g. farm, headwater or possibly slightly larger scales) and their results summed to larger scales but that the aggregated effects of P transport/delivery mechanisms cannot.. Expert 1m had a similar opinion with regard to delivery of P to water bodies where even at headwater scales it is very difficult to resolve the effect of individual processes (given the generally low levels of data available), resulting in significant uncertainties; Expert 1 proposed alternative techniques such as PEDAL where processes controlling the transport of mobilised P are implicit in 'lumped' delivery coefficients estimated from scale-dependant measurements.

The uncertainty associated with explicit scale-dependent modelling of P transport mechanisms can be significant as there are generally inadequate data for robust calibration to allow resolution of the magnitude of effect of individual processes, even at research sites. This means that it may not be possible to determine whether or not models are producing acceptable results for the right reasons as many permutations of model parameters and structures can give a similar result (i.e. equifinality). For these reasons some authors are putting more emphasis on using new technologies, that improve data collection, as a means to help understand catchment processes at larger scales, (e.g. Soulsby *et al*., 2008; Soulsby *et al*., 2006).

The challenges of scale and appropriate process representation for quantifying P fluxes from catchments has spawned the construction of models along the spectrum extremes of 'top-down' data-driven approaches (e.g. the lumped statistical model of Kronvang *et al*., 2003), to 'bottom-up' process-based models (e.g. Arnold *et al*., 1998) together with intermediate approaches including: export coefficient models (e.g. Johnes *et al*., 1996); more conceptual catchment screening models (e.g. PSYCHIC - Davison *et al*., 2008 - which is part of the EA's Decision Support Tool); risk mapping models (e.g. Heathwaite *et al*., 2003); non-parametric models (Schärer *et al*., 2006); and P risk indexes (e.g. Sharpley *et al*., 2003; Hewett *et al*., 2004; Magette *et al*., 2007). The more complex approaches such as the Soil and Water Assessment Tool (SWAT: Arnold *et al*., 1998) and INCA-P (Wade *et al*., 2004) require more data inputs and can suffer from the evaluation problems discussed above which can lead to higher levels of predictive uncertainty (Beven, 2007; Dean *et al*., in press: Wade *et al*., in press). It has also been argued that these complex approaches may not necessarily represent P flux processes any better than more simplistic methods such as P indexes (Veith *et al*., 2005). There is a large amount of literature debating the most appropriate modelling approaches which will not be reviewed here (e.g. see Beven, 2002; Beven, 2007; Refsgaard *et al*., 2007; Krueger *et al*., 2007). There are also more specific reviews of the requirement and roles of models for WFD purposes (Irvine *et al*., 2005; Rekolainen *et al*., 2003) and on harmonisation of nutrient

modelling approaches for the WFD (e.g. the EUROHARP[3] project: Silgram and Schoumans, 2004). One of the outputs from the EUROHARP project was a comparison of modelled phosphorus for European catchments, using tools representative of different tiers. It was generally concluded that no one tool was better than another – each could capture the general trend in risk between catchments – but there was a great deal of variability. The variability was most apparent when analysing the field scale results and demonstrated the perils of calibration at catchment scale without sufficient field and farm scale data to check that over-compensation for errors in processes had not occurred (i.e. getting the right results for the wrong reasons: Expert 2m).

11.8.2 Estimating phosphorus pressure for WFD risk analyses

The discussion above highlights some of the challenges associated with trying to estimate P pressure and identifies two types of methodology which look at the problem from different viewpoints: i.e. top-down empirical approaches and bottom-up process-based approaches. These two extreme approaches mirror the often divided expert opinion on the most appropriate way of making estimates within complex environmental systems. The choice of modelling approach is, of course, not simple and must be made with regard to the questions that need to be answered and the available information: e.g. some simplistic empirical approaches will not be able to predict the effect of changes within a catchment required to inform mitigation programmes. For WFD risk analyses the following methodologies are recommended:

- Tier 1 – a simplistic approach with minimal variables that represent best the observed variations in GQA data. This may be carried out statistically or by use of a decision-tree approach (Expert 2m) which will give only an estimate of likelihood of being at risk without a detailed appraisal of the cause(s) of risk. This may be done for either P fluxes or concentrations and for different periods (e.g. annual or seasonal averages). Care must be taken to represent the GQA data as accurately as possible, with particular attention to biases that result from low temporal sampling resolutions. This approach implicitly includes the effects of instream P processing as these effects cannot be separated from fluxes to waterbodies using GQA data alone. It is important that some estimate of point and diffuse contribution for a given location (e.g. GQA sampling location) is provided where estimates of pressure from different sources is required. Expert 2m also expressed the opinion there needs to be much more specific WFD monitoring to support modelling approaches.

- Higher tiered approaches – All three of the experts consulted in the modelling subject area expressed the view that more detailed approaches for higher tiered modelling should also include increased data collection to support modelling efforts. Expert 2m thought that higher resolution observations in areas of high estimated risk or high uncertainty would be the best approach with modelling estimates acting as additional evidence to support the explanation of failures.

11.8.3 Attribution of P pressure and estimating best mitigation options

As the WFD ultimately requires restoration of good ecological status, higher tier risk analyses should ideally provide an estimate of which factors contribute to P pressure for a given area such that informed choices can be made on appropriate programmes of measures. Estimation of exactly which factors are drivers of P fluxes from individual catchments is difficult and uncertain and suffers from the modelling problems discussed above. That is not to say, however, that estimates cannot be made with the best information available on primary drivers. For example, one of the most frequently used tools for ranking areas (often at field scale) for risk of P loss is some form of P Index which combines information on chosen variables into a relative risk index. The index type methods are hence a simple methodology for identifying

[3] http://euroharp.org/index.htm

spatial variation in risk and 'hotspots' in space. It is becoming widely accepted that mitigation should target these 'hotspots' (or Critical Source Areas: CSAs). Identification of CSAs is hence desirable and a priority for effective catchment management (Expert 1m; Heathwaite *et al*., 2005b. CSAs can however, vary over time and with spatial scale (e.g. CSAs may be defined as a high P catchment, field or vulnerable area within a field Hughes *et al*., 2004). See also the discussion of CSAs in Section 3.

P indexes have been developed largely on the basis of small scale experimentation and on expert opinion (particularly in determination of way that information on each variable included is combined: i.e. the form of the P index equation). Some evaluation of P indices has occurred showing that higher P-Index rankings correspond to greater P fluxes in general, but more validation work is desirable (Osmond *et al*., 2006) to provide evidence that the expert weightings are reasonable. One of the main problems with the form of P indices is likely to be in the way that scale-dependencies have (or have not) been incorporated when upscaling from the underlying experimental results (e.g. the way in which connectivity of spatial units to watercourses is represented and the way in which the issues associated with P delivery to watercourses is taken into account (Beven *et al*., 2005).

In addition to the identification of spatial risk for informing best mitigation options, knowledge of likely timescales for improvements will be required, particularly as good ecological status needs to be achieved by 2015. These kinds of estimates may modify with mitigation measures will be most efficacious: e.g. measures to reduce soil P status may take decades to have the desired effect on reduction of P fluxes whereas installation of measures which intercept P lost via surface pathways have the potential so provide immediate benefits.

11.8.4 Source apportionment

Partitioning of P fluxes into contributions from point and diffuse sources (i.e. source apportionment) is required for WFD risk analyses, is a significant future challenge for river basin managers (Kronvang *et al*., 2007) and includes much uncertainty (de Wit and Bendoricchio, 2001). Even with significant uncertainties, the many interested parties (e.g. regulating bodies, utility companies and representatives of the farming community) will require estimates for decision making. Source apportionment is also required to help researchers identify the diffuse signal such that the effects of catchment processes can be better evaluated (Expert 3ag). This is the case as the high population density England and Wales means that there are few areas (apart from headwaters at very small scales) where a true diffuse source signal is observable making evaluation of diffuse P trends in rivers difficult to ascertain (Expert 2m; Mainstone and Parr, 2002; Bowes *et al*., 2005; Neal and Jarvie, 2005; Jarvie *et al*., 2006). True diffuse signals may be best identified at research farm sites and small headwater catchments where distributed point source inputs from septic tanks and farmyards are absent or more likely to be known (Expert 1m; Expert 2m).

For EA WFD risk assessment scale, and where only GQA information is available, the data almost always includes both point and diffuse source components and the additional effects of instream processing. This makes the apportionment problem hard to resolve and there is an urgent need to identify the best framework to do this (Expert 1m). A number of the different methods that have attempted source attribution for England and Wales, which have not been compared to date on a WFD catchment basis, are listed below:

- For lake catchments by combining inventories of diffuse and point sources of pollution and model output (using PSYCHIC) with measured P concentrations (Defra Project WT0750CSF).
- For rivers using GQA data together with independent point and diffuse (PSYCHIC model output) P pressure data within a regression analyses (EA 2008).
- For rivers using a GIS based SIMCAT model (available for all 11 River Basin Districts in England and Wales) and independent point (PSYCHIC model output) and diffuse load estimates (Crabtree *et al*., 2007).
- Defra projects WQ0106 (Anthony, 2006), WT0750CSF (Anthony *et al*., 2008), ES0205 and WT0719CSF (Anthony and Lyons 2006). These projects have developed a

methodology that predicts losses of P to rivers and in-stream concentrations of P, based on inputs from multiple and distinct source sectors, and critically represents the impact of changes in land management practices in response to Environment Schemes for policy evaluation. A source apportionment is provided for loads, concentrations, between sectors and between farm types, and different phosphorus sources and source areas. This has been done for all WFD catchments across the country. One feature of these approaches has been to use the GQA data for large numbers of catchments simultaneously to carry out the minimum necessary model calibration in order to avoid unjustified over-calibration to any one catchment and to include error estimates with model predictions.

11.8.5 Constraining P pressure model estimates

At a Tier 1 level it is likely that GQA data will be the principal data for constraining P pressure estimates. These data are collected at approximately monthly intervals and measure unfiltered reactive phosphorus as a proxy for bioavailable P. There are approximately 7000 GQA monitoring sites across England and Wales and they are primarily located towards the lower reaches of rivers and often upstream and downstream of WwTW. Jarvie *et al*. (2003) showed that the EA's GQA network was weighted towards measurement of point rather than diffuse sources with their analysis of 10 years of data from the Wye catchment. The GQA data gives a relatively good level of spatial coverage but a limited resolution temporally. The monthly sampling regime employed means that low-flow measurements are over-represented and high flow measurements are under-represented and probably absent altogether for the highest flows. Subsequently (depending upon the method used) these problems can lead to biased estimates when calculating average concentrations and fluxes for a given period (see also the discussion in Johnes, 2007 and Kronvang *et al*., 1997). This is particularly true where concentrations are used independently of the associated flow value to calculate P flux. The pairing of GQA P concentration measurements with a flow estimate should give significantly better estimates of P flux where records are long enough and relatively stable relationships between P concentration and flow can be established (e.g. see Section 9; Bowes *et al.*, 2008; Bowes *et al. accepted* ; Jarvie *et al*., 2003; Neal *et al*., 2002). Where there are few data, or where the pairing of flow and concentration measurements is too costly, corrective rules may be derived using higher resolution datasets for sites of differing flow regime and P source characteristics (i.e. point or diffuse source dominated). For example Johnes (2007) found that catchments with a high baseflow index and/or low population density were found to return a lower RMSE[ϒ] on load estimates compared to those with a low baseflow index and high population density. The aforementioned problems relate to calculation of P fluxes rather than instream concentrations at critical times; the GQA data is perhaps more suited to the estimation of likely growing season P concentrations (particularly the higher concentrations associated with low flow conditions for point source dominated reaches) during ecologically sensitive periods (Edwards and Withers, 2007).

Defining simple rules to resolve the contribution from different sources is more difficult as both the point and diffuse P estimates will be associated with significant uncertainty as discussed above. Thus, in general, GQA data is thought insufficient to characterise diffuse-source contributions as diffuse-source inputs are typically dependent upon storm events for delivery to the stream channel (Expert 1m; Jarvie *et al*., 2003). However, inclusion of additional constraining data such as tracers that give quantitative estimates of point sources may prove invaluable (Expert 2m).

11.9 Taking account of instream processes

In Section 6 two primary reasons why instream modification of terrestrially-derived P fluxes should be taken into account were identified:

[ϒ] Root mean square error

- they potentially modify instream P concentrations and fluxes, particularly during ecologically sensitive periods.
- to determine how representive EA GQA P concentration measurements are for the associated WFD catchment, especially given their proximity to upstream point sources.

To be able to predict the likely effect of instream processes, it was suggested that more detailed reach scale studies are required, perhaps over an extended period of time, such that useful relationships can be found with catchment and reach characteristics (Expert 1m; Sections 6 and 7). The studies would need to be detailed if the resolution of the individual effects of important processes is required such that uncertainties associated with terrestrial P flux estimates do not obscure instream effects (Moss *et al*., 1988).

11.10 Incorporating uncertainties into risk analyses

The discussion in the previous sections of this report have shown that there are not only significant uncertainties associated with estimating P pressure to surface waters but also in estimating how P pressures are likely to affect ecological status and in the definition of good ecological status. It is likely that future risk analyses will use quantitative, qualitative, absolute and relative information to make the various estimates necessary. The information used and subsequent estimates will be uncertain to varying degrees and it is desirable (and called for in ANNEX V, section 1.3 of the WFD) for uncertainties to be taken into account and propagated to final results such that policy makers can make fully informed decisions (Irvine *et al*., 2005; Leuven and Poudevigne, 2002; Williams and Kapustka, 2000). There needs to be an appreciation of the strengths, weaknesses and uncertainties of individual models, data and methods used (Irvine *et al*., 2005). There are many methods to quantify uncertainties, depending upon the estimation methodologies used and once quantified uncertainties may be incorporated using probabilistic measures, where data allows, or fuzzy measures (Fisher, 2003; Beven, 2008) where there are fewer data or where more qualitative information is used. Where risk analyses estimate uncertainty WFD ecological status designations will be uncertain as will estimates of likely causes of failures. In this situation methods to deal with uncertainties that provide robust decisions must be used (Schultz, 2008), for example procedures that combine uncertain pressure estimates with uncertain system sensitivity (e.g. probabilistic risk analysis – Bedford and Cooke, 2001 - or fuzzy methods - Fisher, 2003) as well as adequate communication between scientists and policy makers regarding uncertainty estimates (Borowski and Hare, 2007; Hatton-Ellis, 2008; Liu *et al*., 2008; Faulkner *et al*., 2007) as at some point uncertain estimates are converted into ‘crisp’ decisions.

11.11 Conclusions

Estimating the likely risk to the ecological status of surface waters from elevated P levels is not simple as the link between the two is complex and involves many other factors confounding the definition and use of dose-response relationships in most situations. Thus, it is somewhat artificial to consider the risk to ecology from elevated P levels in isolation, although this does not mean that low levels of P are not desirable. Under the WFD ecological status is expressed relative to a reference state which is difficult to estimate for individual waterbodies/water body types (particularly as no reference sites exist in the UK) and will not be associated with a known reference P status owing to system variability. Although problematic, the definition of ecological status and reference conditions is crucial and should be done in a holistic manner using a number of relevant variables combined in such a way to give robust designations. Only when this has been satisfactorily done can the estimation of current ecological status be carried out using a consistent approach. A number of the experts consulted expressed the opinion that the natural variability of ecological systems and the uncertainties associated with our risk analyses estimates makes the use of simplistic threshold-based water quality standards problematic and suggested that more flexible and holistic approaches are required. Poole *et al*. (2004) suggest the use of regime-based water quality standards which describe the system of interest in terms of valid distributions of variables in space and time. This type of approach could be used for EA WFD risk analyses using distributions of variables describing pressures on the system and factors which may make the system more sensitive or more resilient.

In the pursuit of estimating the required variables data limitations preclude the use of directly observed measures at many locations and some form of modelling will be required. There are a number of risk-screening methodologies in existence for Tier 1 assessments of risk to surface waters but there are few full assessments of the uncertainty associated with their estimates and subsequent implications for WFD designations (although the EA risk analyses have used estimates of the confidence in WFD designations based upon the data available and the statistical fit of Tier 1 type model estimates to observed data). These screening tools do however provide indices of pressures and sensitivities that are the first step in focussing measures for environmental protection and highlighting areas for more detailed investigation: e.g. critical times and locations where high pressures and system sensitivities coincide and areas where confidence in estimates is low. Given the high levels of uncertainty associated with Tier 1 analyses and the inherent natural variability of ecological systems a simplistic method of estimating and combining relevant variables is needed that will minimise the chances of misclassification. Methodologies to investigate these are likely to require a higher level of monitoring to support or refute Tier 1 estimates and to support higher-tier modelling approaches that may improve confidence in estimates and provide more specific information for planning and implementation of mitigation programmes.

12 Overall Conclusions

A structured procedure was developed to elicit information on the appropriate conceptual models and likely uncertainties for the EA's WFD P risk analyses. The elicitation procedure was successful, although some of the experts expressed the concern that quantitatively ranking the effects of individual conceptual model components, at spatial scales and temporal scales of interest, was difficult because of interactions between components. These problems make the identification of mutually-exclusive, cause-effect relationships difficult in such complex, dynamic systems where nonlinearities and feedback effects can have a large affect on system response. Thus, the elicitation questioning meant that experts were required to 'think across scales' and provide informed estimates of likely effects for which there is often little or no supporting scientific research at appropriate scales. Therefore the experts needed to account for the limitations of individual studies at smaller scales and extrapolate to the scales required for WFD risk analyses: the fuzzy weightings elicited are hence subjective.

The estimation of diffuse P pressure based upon the risks posed by intrinsic catchment characteristics and agronomic practice was explored and a simplified scale-dependent conceptual model developed. The chosen model components were dominated by physical factors that control hydrological pathways and connectivity to surface waters and by the management of livestock. There was generally a good degree of agreement between experts on which components were included in the model and on the relative importance and uncertainty weightings assigned to each component. In many cases, however, the supporting scientific evidence was identified as being based upon small scale studies such that the changes in relative importance with increasing scale is not well known leading to significant uncertainty. At larger scales, a lack of data and the spatial and temporal heterogeneity of the system make it difficult to apply the relatively high levels of process-understanding we have and provide only weak constraint on large scale modelling efforts. Thus, for certain elements of EA WFD risk analyses it may be beneficial to collect data at WFD catchment scales such that scale-dependent rules may be derived to inform future modelling approaches.

It is now recognised that septic tanks have the potential to contribute a significant proportion of P to some waterbodies, although the evidence base to support this is limited, uncertain and sometimes anecdotal. The P flux from septic tanks may result from poor tank construction, maintenance and location. With little quantitative evidence of the magnitude of P fluxes (particularly at larger scales) most attempts to quantify them will be highly uncertain and often use published export coefficients or retention coefficients. At all scales, without more quantitative studies that apportion the relative magnitudes of P flux between sources there is likely to be little improvement in our confidence in estimates of P flux from septic tanks. Better quantitative information may allow simplistic rule-based approaches to estimate the likely variability in flux given certain catchment properties. Perhaps one of the most positive directions for future research is in the use of tracers, although more progress will need to be made for quantitative source apportionment.

The accurate calculation of point source P fluxes from WwTW is crucial for source apportionment which serves to isolate the diffuse contribution to catchment P fluxes. The uncertainty associated with point source flux estimates is derived mainly from the data available; there is likely to be a gradual increase in confidence from methods that used generalised, back-calculated per capita export coefficients to those using flows and concentrations measured at adequate frequency. The latter may now be possible with the current availability of WwTW daily flow data coupled with WIMS P concentration data. Calculation of fluxes in this way is time-consuming and expensive, but would serve as good estimates of point source P fluxes for a number of years. A requirement to characterise the relationship between flow and P concentration for major point discharges, and for a representative sample of smaller discharges, is therefore a pre-requisite for constraining uncertainties in catchment flux estimates.

The instream processing of phosphorus, including within the HZ, is of particular importance for the EA's risk assessments as many of the models currently used to predict diffuse P fluxes make estimates of P delivered to the stream and do not explicitly take into account instream processes. Additionally, instream P measurements made as part of the EA's GQA monitoring

strategy need to be put into context within their WFD catchments or reaches with respect to the proximity to upstream sources. Phosphorus fluxes to rivers are likely to undergo numerous physical, chemical and biological transformations during downstream transport. These interacting processes act at timescales from effectively instantaneous chemical reactions to the gradual and stepwise transfer of P and sediment during hydrological events. The significance of these processes, with regard to the risk from P of degraded ecological status, is primarily associated with the net retention, or release, of bioavailable P concentrations during the growing season: although the resuspension of P-rich sediments during the autumn and winter may be important for lakes and transitional waters. There are hence two primary considerations for instream processing: the attenuation/release of growing season river P concentrations and net effect of these processes over annual or longer timescales. At WFD scales, the identification of attenuation/retention processes occurring, their relative magnitudes and temporal variation is highly uncertain as resolution of individual effects is difficult and as with terrestrial P losses our knowledge of scale-specific representations or methods of upscaling from detailed studies is limited. Studies are needed which allow the development of simplistic rules, perhaps for river reach typologies where a significant amount of reach/catchment scale data has been used in their derivation.

As there are no drinking water standards for P and there are no measures of direct ecological effects in groundwater (GW), surface waterbodies are the ultimate receptors for GW P (EA, 2007). Principal sources of P in groundwaters (over and above naturally occurring background levels) are derived from diffuse agricultural sources, point agricultural sources (e.g. leakage from slurry tanks) and quasi-point sources: septic tanks and sewer leakage. Both point and diffuse derived P also enter groundwaters during recharge from rivers. The flux of P to GW from terrestrial systems is primarily controlled by the degree of attenuation in soil and drift deposits. The subsequent flux of GW P into surface waters is dependent upon further attenuation within the aquifer and the degree of connectivity between the GW body and surface water bodies. At WFD catchment scales, the primary controls on P pressure to surface waters from GW can be described by the P pressure (estimated from terrestrial models) on groundwater and the degree of connectivity between the groundwater and surface water (e.g. using BFI). This type of approach gives a general P flux estimate to surface waters. However, in reality GW contributions can be highly variable with higher base flow contributions during ecologically sensitive summer low flow periods. It is hence desirable that seasonality is taken into account in risk assessments. The degree of uncertainty associated with predictions of the likely magnitude of effect of GW P on surface water bodies is determined primarily by limited observational data and system heterogeneity.

The link between elevated P concentrations and ecological status in rivers is less clear than that observed in lakes, primarily as a result of shorter reach residence times and the influence of disturbance by high flow velocities. The experts consulted generally agreed on the effects that eutrophication had on the abundance and composition of the aquatic plant community and on the factors that control P-ecological indicator relationships. The primary controlling conceptual model components identified were the effects of flow regime, grazing of biofilms and the bioavailability of P species. The experts were not fully clear on the significance of the role of alkalinity in changing the response of the ecological community to given levels of P and thought that the WFD SRP concentration standards adopted by UKTAG (which are set significantly higher for high alkalinity waters) may not be precautionary enough.

As for rivers, the link between elevated P levels and lake ecology is not simple owing to their complex, dynamic and stochastic nature. Any simplistic relationship between P and ecological status is confounded by interactions between factors such as growth limitation by other nutrients and light and food web interactions which allow lakes to shift to alternate stable ecological states for a given P status. Use of P-ecological status dose-response relationships hence leads to significant uncertainties in WFD designations. These uncertainties can be reduced where lake-specific information is available. Given these problems, the lake system experts suggested that it may be more realistic to use holistic approaches for determining lake ecological status, perhaps using regime-based representations that allow for natural system variability.

Given the complexities of the relationships between P levels and ecological status in surface water bodies, it is somewhat artificial to consider the risk to ecology from elevated P levels in isolation: although this does not mean that low levels of P are not desirable. Under the WFD

ecological status is expressed relative to a reference state which is difficult to estimate for individual waterbodies/water body types and may not be associated with a known reference P status owing to system variability. Although problematic, the definition of ecological status and reference conditions is crucial and should be done in a holistic manner using a number of relevant variables combined in such a way to give robust designations. Only when this has been done satisfactorily can the estimation of current ecological status be carried out using a consistent approach. A number of the experts consulted expressed the opinion that the natural variability of ecological systems and the uncertainties associated with our risk analyses estimates makes the use of simplistic threshold-based water quality standards problematic and suggested that more flexible and holistic approaches are required, such as regime-based water quality standards which describe systems in terms of valid distributions of variables in space and time (Poole *et al*., 2004). This type of approach could be used for EA WFD risk analyses using distributions of variables describing pressures on the system and factors which may make the system more sensitive or more resilient.

For WFD purposes where estimates are required nationally, data limitations may preclude the use of directly observed measures at many locations and some form of modelling will be required. Although uncertain, the modelling estimates available provide indices of pressures on surface waters and their sensitivities that are the first step in focussing measures for environmental protection and highlighting areas for more detailed investigation. Methodologies are required that can combine uncertain indices of relevant factors that can take into account natural variability such that chances of misclassification are minimised. These methodologies are, in some cases, likely to require a higher level of monitoring to support or refute Tier 1 risk estimates and to inform higher-tier modelling approaches that may improve confidence in estimates made and provide more specific information for planning and implementation of mitigation programmes.

12.1 Final Recommendations

12.1.1 Policy research priorities

Environment Agency requirements, with respect to the effect of elevated P levels on surface water ecology, are all-encompassing and are treated in detail in the sections above. The following points highlight the primary ways in which these requirements can be met. In the text below italicised text indicates EA requirements and un-italicised text indicates the corresponding recommendations.

EA policy requirements include:

- *Setting and refining standards with improved ecological relevance;*

General recommendations: Setting and refining standards to be ecologically relevant assumes a well-established causal relationship between elevated P levels and ecological status. This assumption has been shown to be unclear in the discussion above (particularly for shallow lakes and rivers) such that regulatory authorities need to decide upon how precautionary standards should be as part of a risk-based approach that should also include other uncertainty estimates (see below). In parallel, there should be a focus on other interacting controls on ecological status (e.g. nitrogen status and/or the effect of river flow regime) that may help explain some of the observed variation.

Specific recommendations:

I. Consideration of ecologically sensitive periods within risk analyses (e.g. the use of growing season bioavailable P concentrations, rather than annual averages);

II. Development of holistic methods of estimating ecological status that allow for system variability (particularly for rivers);

III. More focus on defining the sensitivity of the receiving waterbody (e.g. does sensitivity go beyond alkalinity classes);

IV. See also science recommendations below.

- *Understanding sources, processes, pathways and the links to standards and effects, for rivers and for lakes, to a 'sufficient degree[ϖ]' to decide on appropriate control measures at sub-catchment, catchment and larger (including national) scales.*
- *Determination of minimum monitoring levels and model data requirements;*
- *Identification of the most appropriate models for our purposes and the associated uncertainties.*

General recommendations: Sections 3 and 11 above highlight that, in general, we have a good level of scientific understanding of sources, processes and pathways but cannot predict well how they interact in time and space as manifest P pressure. Thus without detailed and expensive data collection there will always be a limit to the accuracy of determination of P pressure, such that even catchments having similar observable characteristics may have significantly different P pressure signatures. In this situation scale-dependent and application-dependent data collection and modelling will be required as will a quantitative assessment of uncertainty. Quantitative uncertainty estimation will help to inform future monitoring and modelling strategies by highlighting the value of different types and resolution of information in reducing uncertainties. In this way, models can become part of an ongoing dynamic process of refining the representation of catchments, rather than the static representations that they often become. This process should include the generation of scale-dependent hypotheses to be tested in real catchments, the understanding from which then can feed into model development.

Specific recommendations:

I. Robust analysis of the likely improvement of WwTWs P fluxes using recently available WwTW flow data;

II. A study that quantifies P losses at GQA gauging locations using paired flow and P concentration measurements (interpolated flow estimates will be required at some locations);

III. Better use of the knowledge that already exists within the EA. This could be an initial desk study aimed at assessing the likely value of information held at local EA offices;

IV. See also science recommendations below.

12.1.2 Science research priorities

Research priorities relating to fundamental scientific questions are relatively few compared to those which are required to improve our ability to predict the responses of complex systems at larger spatial and longer temporal scales. This problem is often exacerbated by low resolution, and in some cases low quality, data which is to be expected for large scale risk assessments.

Applied priorities:

I. More detailed modelling/data collection in a few catchments/reaches;

II. Scale-dependant determination of mitigation measure efficacy (including combined effects): requires long-term catchment manipulation studies;

III. Reach-scale studies to determine the net effects of in-stream processes (including HZ processes) during ecologically sensitive periods and for a number of stream typologies. Combined with studies to compare reach behaviour to, river geomorphological and habitat surveys and broader catchment characteristics;

[ϖ] 'Sufficient degree' implies a chosen level of risk – in this case chosen by the regulator

IV. Comprehensive model evaluation (and intercomparison) including uncertainty analyses for models used within risk analyses and for source apportionment;
V. Scale-dependant determination of transport/delivery of P;

Fundamental science priorities:

I. Greater understanding of the functioning of wetland and HZ processes. These are 'bottlenecks' in the system and determination of the magnitude of their effect is a priority;

II. Research into pathways for P into groundwaters: e.g. quantification of fluxes from anaerobic plumes from septic tanks and unlined farm slurry pits;

III. Understanding the interacting roles of flow-regime, biofilm grazing and nutrient status on periphyton communities;

IV. Quantifying of internal P loads in shallow lakes and mixing of hypo- and epi-limnetic waters for deep lakes.

13 List of Abbreviations

BFI:	Base Flow Index
BGS:	British Geological Survey
CEH:	Centre for Ecology and Hydrology
Cha:	Chlorophyll *a*
CSA:	Critical Source Area
DWF:	Dry Weather Flow
EA:	Environment Agency
ECSFDI:	England Catchment Sensitive Farming Delivery Initiative
EPA:	Environmental Protection Agency
EQR:	Ecological Quality Ratio
GQA:	General Quality Assessment
GW:	Groundwater
GWDC:	Groundwater Dominated Catchment
HER:	Hydrologically Effective Rainfall
HZ:	Hyporheic Zone
NUSAP:	Numerical, Unit, Spread, Assessment and Pedigree
MRP:	Molybdate Reactive Phosphorus
P:	Phosphorus
PE:	Population Equivalent
Q5:	Flow magnitude exceeded 5% of the time
Q50:	Flow magnitude exceeded 50% of the time
Q95:	Flow magnitude exceeded 95% of the time
RBC:	River Basin Characterisation
RIVPACS:	River Invertebrate Prediction and Classification System
SRP:	Soluble Reactive Phosphorus
SW:	Surface Water
SWDC:	Surface Water Dominated Catchment
TP:	Total Phosphorus
TRP:	Total Reactive Phosphorus
UKTAG:	United Kingdom Technical Advisory Group
WFD:	Water Framework Directive
WIMS:	Water Information Management System

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