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Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies

Science Report SC020029/SR

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EXECUTIVE SUMMARY

One of the first tasks outlined in the Water Framework Directive (WFD) is for Member States to carry out a review of the environmental impact of human activity on the status of waters. As part of this review, it is a requirement to collect information on the type and magnitude of significant pressures to which surface waters are exposed, and based on the characteristics, or susceptibility, of water bodies to those pressures, carry out an assessment of the risk that water bodies will fail to meet the Directive's environmental objectives, notably the specific WFD objective of achieving "good status".

This report summarises the development of a number of methodologies required for carrying out this initial risk assessment for lakes, specifically in relation to nutrient pressures. It provides a number of approaches of increasing sophistication for assessing the magnitude of diffuse and point-source nutrient pressures and develops a nutrient classification for different lake types that can be applied to assess the impact of these pressures. Approaches to ecological classification for phytoplankton composition and abundance in lakes and slow-moving rivers are also outlined, as phytoplankton is considered the biological quality element most sensitive to nutrient pressures.

Development of the nutrient (total phosphorus (TP)) classification initially involved determining reference conditions. Five approaches for identifying site-specific reference conditions were examined and lake ecotype-specific reference conditions were then developed for risk assessment purposes. This analysis highlighted distinct differences between different lake types, with increasing reference TP concentrations with lake classes of decreasing lake depth and increasing alkalinity, i.e. lowest TP reference concentrations for deep, low alkalinity lakes and highest for very shallow high alkalinity lakes. TP concentrations were then derived for boundary values of the five WFD status classes (high, good, moderate, poor, bad) for each lake ecotype, with the good/moderate boundary set at a threshold representing a doubling of reference concentrations. Any sites observed or predicted to have TP concentrations higher than this boundary would be considered as at risk of failing to achieve good status.

The relationship between observed in-lake concentrations of TP and phytoplankton chlorophyll$_a$ were then explored for each lake ecotype, in order to develop an appropriate classification for phytoplankton abundance. This analysis revealed that, although deeper lakes appeared to show a reduced response to nutrient conditions than very shallow lakes, there was no significant difference in the response between different lake ecotypes, either in relation to depth or alkalinity type. A TP-chlorophyll$_a$ relationship specific to all GB lakes was, therefore, calculated for potentially deriving chlorophyll$_a$ reference conditions. An alternative approach to developing chlorophyll targets for lake types, independent of TP, was also outlined based on the light requirements for different macrophyte groups, or, oxygen requirements of fish. These latter classifications, however, require further development and validation before application to the risk assessment process.

A novel approach to ecological classification of phytoplankton community structure was developed using phytoplankton functional groups. Morphological or physiological characteristics of phytoplankton taxa have been used to populate the phytoplankton functional groups. This project developed probabilities for the occurrence of these functional groups in different lake and river types with increasing trophic status, at different times of the year, or with respect to rivers, different flow regime. A WFD-compliant assessment of ecological status is possible by comparing the similarity of observed phytoplankton assemblages with that of a pre-determined reference assemblage. Application of the approach was examined using the long-term phytoplankton dataset available for Windermere. An analysis of spring samples taken over the past five decades indicated that the phytoplankton classification tool gave a good representation of changing ecological impact associated with both increases and decreases in nutrient loading to the lake. Wider validation of the classification structure and its application to a number of lake ecotypes, however, must be carried out before it can be adopted nationally for WFD purposes.

The assessment of nutrient pressures considered three types of point-source pressures: sewage treatment works (STWs), septic tanks and cage fish farms. The inputs from the former are generally considered relatively well understood, but a review of methods and data availability highlighted that this is not the case. Not only are the number of people served by STWs poorly known (only design capacity is readily available), but the phosphorus export coefficients for humans following sewage treatment are not well defined. The limited data available on actual (rather than consented) flow and TP concentrations of STW discharges showed an apparent variability in STW efficiency. Due to the large sensitivity of any model of nutrient pressures to the load estimates from STWs, it is strongly recommended that the UK agencies commission a study on the export coefficient of phosphorus from humans before and after sewage treatment to establish an accepted value for future application.

The TP load from septic tanks is also difficult to evaluate separately from that emanating from STWs because the number, location and level of maintenance of private septic tanks is unknown. The very limited studies carried out using intensive methods on specific catchments, suggest the contribution from septic tanks can be significant. For this reason it is essential that the UK agencies determine the number, and preferably the location, of septic tanks across UK. The level of maintenance of septic tanks, and its relationship to P loss to nearby surface and groundwater, is also a critical area for further study in rural catchments. In the absence of recognised values for STWs or septic tanks, an average TP export coefficient value for humans is recommended for initial risk assessment purposes applicable to either secondary sewage treatment or treatment through a septic tank system.

The TP load from fish farms can be assessed for locations where the type of fish cultured and the annual tonnage produced are known. At present, these data are incomplete for Scotland and unavailable for England and Wales. Detailed studies at two sites suggest, where present, fish farms contribute a significant proportion of the TP budget to a lake. A database needs to be compiled containing location, size (e.g. consented biomass) and fish species data for all fish farms in GB to allow TP loads to lakes from these systems to be properly evaluated.

The assessment of diffuse nutrient pressures for a water body considered three approaches of increasing sophistication: a basic “risk screening” approach (tier 1) applicable to all GB lakes using land cover and animal stocking data to estimate TP loads to a lake. A slightly more sophisticated approach, the Pressure Delivery Risk Screening matrix (PDRS) (tier 2), which links estimates of the nutrient pressure associated with agricultural activity with characteristics of the catchment that indicate the likelihood of nutrients reaching a water body. Finally a third modelling approach, the Phosphorus Indicators Tool (PIT) was considered, which has three model layers: (1) phosphorus loss-potential from agricultural activities, (2) phosphorus transfer pathways in the catchment, and (3), phosphorus delivery. Both an uncalibrated PIT tool applicable at a national scale (tier 3a) and a site-specific calibrated PIT tool (tier 3b) were evaluated.

Comparison of measured and tier 1 modelled data for 50 test lakes suggested the tier 1 approach is a relatively robust method for general risk assessment purposes at a national scale. Animal stocking data for Scotland is now available in the GB lakes inventory, making it a consistent approach across the whole of GB. Validation and sensitivity analysis of export coefficients is, however, recommended as levels of uncertainty with coefficients are currently unquantified. The tier 2 approach is recommended as a national tool for an assessment of pressures only. It is particularly useful identifying the spatial distribution of nutrient pressures within catchments and is applicable to groundwater as well as surface waters. The tier 3a and 3b approaches provide much more detailed site-specific methods for understanding the main sources and pathways of nutrients within a catchment. They are likely to be particularly applicable in the design of the programme of measures at a catchment scale. To date coefficients within the PIT model have been calibrated for the Windermere, Slapton Ley, Esthwaite Water, Barton Broad and Blelham Tarn catchments. Ongoing work under the DEFRA funded PEDAL (Phosphorus Export and Delivery from Agricultural Land) project (PEO113) seeks to quantify the delivery and transfer of phosphorus to water bodies which in essence is the final element of the PIT approach. This will provide not only a spatial distribution of phosphorus predictions, but also an assessment of the absolute quantity of phosphorus distributed in a catchment that reaches the catchment outlet, which will allow predictions to be compared with in-lake water quality data, as is possible with tier 1.

An initial risk assessment using the recommended guidance for a tier 1 approach was carried out on all lakes in Great Britain >1 ha in size (>14,000 lakes). Overall 51% of sites are predicted to not meet the TP targets identified for high or good status and must, therefore, be considered at risk. Of the six lake ecotypes examined, very shallow, medium alkalinity lakes (equivalent to “mesotrophic lakes” under the Habitats Directive) appear to be at greatest risk (92 % of GB sites). There were also major regional differences in numbers of sites at risk. Scotland has by far the fewest sites at risk (18%), England by far the most (88%), with Wales having an intermediate percentage (56%). A large number of sites were also identified as “unknown” status, predominatly peaty lochs in Scotland. There is a high probability that most of these sites are not at risk from nutrient pressures, being largely present in the more undisturbed parts of northern Scotland. Further work is required not only to establish reference nutrient conditions for peaty, marl and brackish lake ecotypes, so their nutrient classification can be developed, but also the ecological impact of nutrient pressures in these lake types requires much further study.

An initial validation of the Tier 1 approach was carried out on 50 well-studied test lakes to examine whether the GB-wide results can be taken as a relatively true picture. Expert opinion on the project team (including Agency representatives), was generally in agreement with the predicted status classes for the test lakes. It appeared to identify sites generally considered of high (Loch Ness) or good (Loch Lomond) status, sites considered to be around the good/moderate boundary (Loch Leven, Loweswater, Malham Tarn), sites at risk for which there exists some concern (e.g. Esthwaite Water and Loch Earn) and sites clearly of poor or bad status (Rostherne Mere and Marsworth Reservoir). A few sites were predicted as not at risk, when in fact observed data suggests they are. This further supports the recommendation that all “important” sites are automatically selected for further investigation or operational monitoring. For similar reasons, representative sites predicted to be near the good/moderate class boundary should also be selected for further investigation.

In summary it does, therefore, appear that a large number of sites in GB are at risk from nutrient pressures. The risk assessment process outlined above is, however, just the first step in a tiered process in identifying sites at risk. Those identified require further investigation, both monitoring and modelling using the more sophisticated approaches outlined in this report. It may be that these more detailed studies do not significantly alter the results and that these initial results reflect a true picture of the impacts of nutrient pressures over the more populated regions and intensively farmed landscape of GB, particularly England compared with much of Scotland and North Wales.

The 2004 risk assessment is, however, just the first stage in delivering improved management of water resources through the WFD. Later stages in the WFD will deal with how we manage these risks at both a national and local scale. More sophisticated approaches to both the assessment of pressures and assessment of impact will ultimately be necessary.

1 INTRODUCTION AND PROJECT OBJECTIVES

1.1 Background

The Water Framework Directive (Council of the European Communities, 2000) is the most significant piece of European water legislation for over twenty years. Its implementation sets a challenging timetable for delivering the Directive's requirements, particularly in relation to technical Annexes II and V.

To begin the process of satisfying the Water Framework Directive's (WFD) requirements in relation to nutrients, in 2001/02 the Environment Agency contracted ENSIS Ltd (University College London (UCL)) and the Centre for Ecology and Hydrology (CEH) to undertake a literature review on the impact of nutrients on biological assemblages in surface freshwaters (Nutrient Conditions for Different Levels of Ecological Status and Biological Quality in Surface Waters (Phase I) - R&D Technical Report P2-260/4) (Carvalho *et al*., 2002). This review concluded that biological assemblages are shaped by both natural environmental factors (physical, chemical and biological) and a number of 'pressures' from human activity, including nutrient pollution. Areas where additional work was required before nutrient conditions and biological quality indicators could be linked with an acceptable degree of reliability were identified in the recommendations of the report, and have been prioritised by the project board to form this next stage of the project (P2-260/9).

In addition to P2-260/4, the Agency and the Scottish and Northern Ireland Forum For Environmental Research (SNIFFER) funded an R&D project to 'Develop a GIS-based inventory of standing waters for England, Wales and Scotland, together with a risk-based prioritisation protocol' (P2-260/2) which can be used to establish relative priorities for action (Bennion *et al*., 2002). This project was undertaken by the Environmental Change Research Centre/ENSIS Ltd at UCL (ECRC-ENSIS) and CEH, both of whom had access to extensive data sets and experience of developing similar systems elsewhere. The inventory has been produced, a prioritisation protocol developed and a basic risk assessment procedure outlined, but potentially the approach could be improved by:

- Developing a process-based approach for calculating the diffuse agricultural nutrient loading rather than using simple export coefficient models;
- Refining estimates of point-source nutrient pressures;
- Developing phytoplankton abundance response using metabolic models and ecotype-specific regression models, rather than OECD regression models;
- Developing a phytoplankton classification tool for predicting the response of phytoplankton to nutrient pressures;

- Developing detailed guidance on risk assessment based on refined approaches

Of particular priority, is the work required to combine outputs from the two projects outlined above and develop them further to address the WFD requirements for undertaking nutrient risk assessments, and for developing a phytoplankton ecological classification. The existing and refined approaches to risk assessment also require validation on a number of test lakes spanning a wide range of GB lake types.

This report describes the results of this work, carried out under Environment Agency R&D Project P2-260/9 "*Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies*".

1.1.1 Ecological Risk Assessment

The WFD requires each Member State to undertake an initial assessment of impacts (brought about by human activities) on the water environment by the end of 2004 (Article 5, and Annex II, Section 1 and 2). This process will identify those water bodies at risk of failing to meet their environmental objectives, set by the Directive. The risk assessments need to be structured so that the level of detail needed for the assessment is proportionate to the difficulty in judging whether or not a water body is at risk.

The risk assessment process consists of three components:

- Characterisation of water bodies - differentiation into types and identification of reference conditions;
- Identification of type and magnitude of the pressure (nutrients in this project);
- Assessment of impact, taking account of the sensitivity/susceptibility of the receiving water body to the pressure (nutrients in this project). This will depend on a water body's natural characteristics and the other pressures acting upon it.

Further information on the approach to undertaking risk assessments is available in Section 7, The Water Framework Directive - Guiding principles in the technical requirements (Environment Agency, 2002). Initial guidance on undertaking the risk assessment was developed by IMPRESS (2002), the WFD Common Implementation Strategy Working Group examining the assessment of pressures and their impacts, and more recently, by UK TAG (2003).

A tiered-approach to risk assessment was outlined in Bennion *et al*. (2002), as illustrated in Figure 1.1. The output of the risk assessment process will be a list of water bodies considered to be at risk of failing to achieve one of the relevant EC environmental quality objectives (WFD good status, Habitats Directive favourable conditions, etc.) and which may, consequently, need appropriate protection and improvement measures. Surveillance monitoring will be carried out at some sites to validate the outcome of the risk assessments and operational monitoring will be undertaken at sites to establish the status of those sites identified as at risk..

In this current project a “Tier 1” approach to risk assessment is used as a general phrase to identify an approach applicable to all GB lakes using nationally-available data. A detailed site-specific “Tier 3” approach is considered appropriate for particularly important sites and will also be critical in future stages of WFD implementation, such as informing the River Basin Management Plan Programme of Measures.

At the start of this project, there were a number of constraints on undertaking the WFD risk assessment, in particular the following tasks need to be completed:

- differentiation of water bodies into types;
- identification of reference conditions;
- definition of criteria for defining good status.

The first of these was completed during the course of the project (Phillips, 2003a) and is briefly reviewed here. The remaining two have been developed in tandem with the project and are considered in relation to nutrients, phytoplankton abundance (chlorophyll$_{a}$) and phytoplankton composition in the respective chapters (Chapters 2-4).

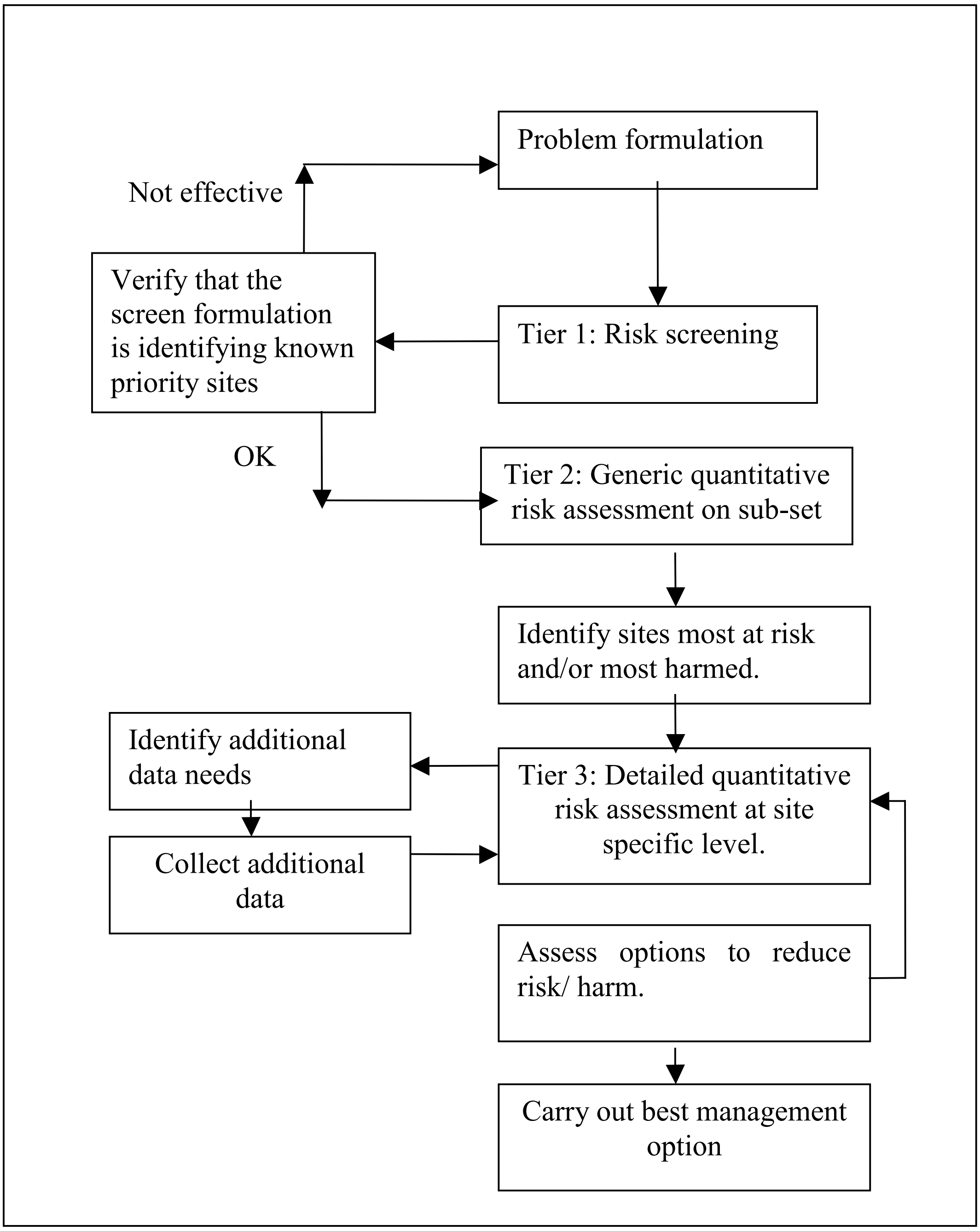


Figure 1.1 Risk-based prioritisation protocol approach for lakes (Bennion *et al.*, 2002)

1.1.2 Lake typology

At the outset of this project, a GB lake typology had not been finalised. Using expert judgement, a provisional typology based on depth and alkalinity was developed. These were considered to be 'functional' types, which, in general, should exhibit differing sensitivities to phosphorus (Table 1.1). The eight lake types outlined in this typology were used in the development of the phytoplankton classification tool (Chapter 4).

During the project, a core lake typology for GB was outlined by the Lakes Task Team (Phillips, 2003a). This core typology was based on geology rather than alkalinity, although alkalinity ranges for each geological class were provided and alkalinity data were used to over-ride the geology class when available (Table 1.2). The Lakes Task Team (LTT) typology was broadly similar to the projects interim typology, but had six geology classes, instead of the four alkalinity classes outlined in table 1.1 (marl and brackish added) and also used slightly different alkalinity boundaries. The 12 lake types in the core LTT typology were adopted in this project for the nutrient and chlorophyll$_a$ classifications and Tier 1 risk assessment procedure (Chapters 2, 3 and 7).

Table 1.1 Provisional lake typology for Great Britain

	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 1.2 Lakes Task Team lake typology for Great Britain (Phillips, 2003a)

	Peaty	Siliceous Low Alkalinity	Siliceous Medium Alkalinity	Calcareous High Alkalinity	Calcareous Marl	Brackish
Geology	>75% peat	>90% Si	50-90% Si	>50% $CaCO_3$	>65% limestone	
Alkalinity (µequiv. l^{-1})		<200	200-1000	>1000		
Conductivity (µS cm^{-1})						>1000
Colour (MgPt l^{-1})	>30					
Very Shallow (<=3 m)	Peat_S	Si_LA_S	Si_MA_S	Calc_HA_S	Marl_S	Brackish_S
Deeper (>3 m)	Peat_D	Si_LA_D	Si_MA_D	Calc_HA_D	Marl_D	Brackish_D

1.2 Project Objectives

Overall objective: To develop the existing risk-based prioritisation protocol for lakes to provide a robust risk assessment approach for lakes and slow-flowing rivers in relation to nutrients to satisfy the requirements of the WFD.

Specific objective 1 - to refine existing risk-based prioritisation protocol for lakes by improving the nutrient pressure, sensitivity analysis and nutrient and phytoplankton response elements.

Specific objective 2 - to develop a phytoplankton classification tool for lakes and slow flowing rivers

1.3 Organisation of the Report

This report is divided according to the risk assessment task structure illustrated in Figure 1.2. Chapters 2, 3 and 4 initially examine ecological classifications for the three quality elements considered most sensitive to nutrient pressures (nutrient concentrations, phytoplankton abundance and composition). Chapters 5 and 6 outline approaches to the assessment of point-source and diffuse nutrient pressures. Chapter 7 applies the recommended Tier 1 risk assessment guidance to all GB lakes and examines the output in detail on 50 well-studied test lakes. Each chapter concludes with summary sections outlining interim recommendations on current and future approaches to the tiered risk assessment and any data availability, harmonisation or quality control issues. The recommended guidance to assessing risks from nutrient pressures is summarised in Appendix 1.

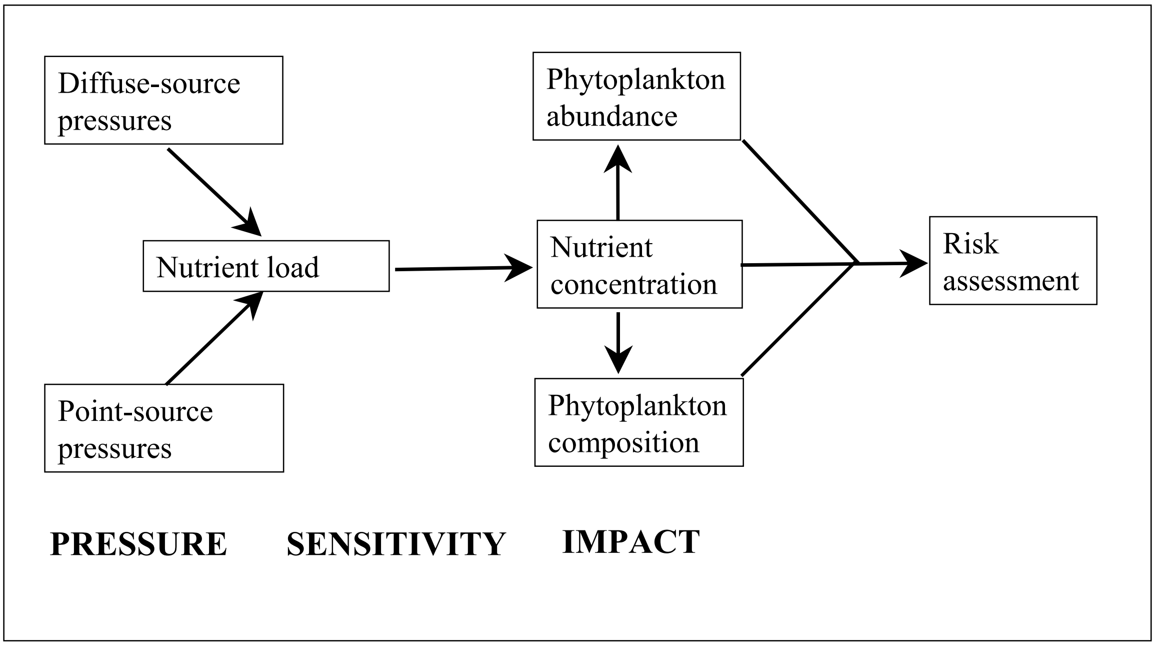


Figure 1.2 Risk assessment task structure

2 NUTRIENT CONDITIONS

Laurence Carvalho, Geoff Phillips and Helen Bennion

2.1 Introduction

The risk assessment approach follows WFD guidelines, by assessing likely risk of failing to achieve one of the relevant EC objectives (WFD good status, Site Condition Monitoring favourable conditions (Habitats Directive)). This chapter focuses on developing a preliminary classification of nutrient conditions for GB lake ecotypes, which can be used to assess the risk of not achieving Good Ecological Status under the WFD. Specifically, it focuses on developing risk boundaries for total phosphorus (TP) derived from the determination of reference conditions and TP concentrations which are likely to support the good/moderate ecological status class boundary. The nutrient classification is applicable to both operational monitoring and the assessment of status and the earlier risk assessment process.

To develop these criteria a number of tasks need to be considered:

- definition of criteria for defining good status;
- differentiation of water bodies into type;
- identification of site- or type-specific reference conditions;
- assessment of current conditions.

2.2 Good Ecological Status

The WFD requires ecological status to be measured in terms of an Ecological Quality Ratio (EQR). This represents the relationship between current observed conditions and reference conditions and should be a numerical value between 0 (bad status) and 1 (high/reference status). For a parameter (quality element) that increases in value with deterioration, such as TP and chlorophyll$_a$, EQR = reference/observed, whereas for a parameter that decreases in value with deterioration (e.g. pH, species richness), EQR = observed/reference.

There are three potential options for defining risk assessment boundaries for nutrient conditions based on the EQR scale (Table 2.1): (1) splitting the EQR scale into 5 equally-spaced classes for all lake types ("equal approach"); (2) using the same equally spaced EQR-classes for all lake types, but having a threshold default TP concentration for high and/or good status ("threshold approach"), or; (3) splitting the EQR scale into five unequally-spaced classes ("unequal approach"). Another consideration could be to use different EQR scales for different lake types. The "unequal approach" illustrated in Table 2.1 represents a doubling of TP concentrations.

A comparison of these three approaches and their implication in terms of nutrient concentrations for different risk assessment classes is illustrated for three lake types in Tables 2.2-2.4. These comparisons are for illustrative purposes only.

Table 2.2 illustrates the sensitivity of the EQR scale for an oligotrophic lake with a reference condition of 5 μgl^{-1}. If using the "equal" approach, an increase in only 3 μgl^{-1} TP would be sufficient to shift this lake from high to moderate status. This could well be within the precision of the analysis under standard EA and SEPA sampling and laboratory quality control procedures. Concentration shifts of this order of magnitude could also be seen at a single site simply due to inter-annual variability. This approach is, therefore, not recommended for nutrient-poor lakes.

The "unequal" classification illustrated, based on a doubling of phosphorus concentrations is less sensitive at the lower end of the TP concentration scale and appears to produce ecologically and practically more appropriate phosphorus bands for oligotrophic and mesotrophic lakes, but may be unacceptable for naturally eutrophic lakes (good status up to 100 μgl^{-1}).

The only alternative for dealing with the sensitivity of the EQR scale for oligotrophic lakes in the "equal approach" is to use a default threshold value, such as all sites <10 μgl^{-1} achieve high status, as is used in the current SEPA quality classification of Scottish standing waters (Fozzard *et al.*, 1999).

Initially, we recommend adopting the "unequal-doubling approach" as this appears to produce classifications that more closely resemble the well-established OECD TP classification scheme. The OECD scheme was developed from the consensus of opinion of a wide range of lake experts considering nutrient criteria for distinguishing lakes in different trophic states and must, therefore, be considered as a helpful comparison.

Further comment on these threshold values and boundaries is provided in Chapter 7 based on the quantification of EQRs for 50 well-studied test lakes. Further guidance may also come from the CIS Working group on ecological status (ECOSTAT) and the planned inter-calibration process.

Table 2.1 Options for ecological risk assessment class boundaries

Status class	Equal	Threshold	Unequal
High	0.8 - 1.0	0.8 - 1.0 or <7.5 μgl^{-1} TP	0.8 - 1.0
Good	0.6 - <0.8	0.6 - <0.8 or <10 μgl^{-1} TP	0.50 - <0.80
Moderate	0.4 - <0.6	0.4 - <0.6	0.25 - <0.50
Poor	0.2 - <0.4	0.2 - <0.4	0.125 - <0.25
Bad	0 - <0.2	0 - <0.2	0 - <0.125

Table 2.2 Comparison of total phosphorus concentration ranges for three classification approaches for an hypothetical oligotrophic lake (5 $\mu g l^{-1}$ TP reference condition)

Status class	Equal	Threshold	Unequal
High	<=6.25	<7.5	<=6.25
Good	>6.25 - 8.4	7.5 - <10	>6.25 - 10
Moderate	>8.4 - 12.5	10 - 12.5	>10 - 20
Poor	>12.5 - 25	>12.5 - 25	>20 - 40
Bad	>25	>25	>40

Table 2.3 Comparison of total phosphorus concentration ranges for three classification approaches for an hypothetical mesotrophic lake (25 $\mu g l^{-1}$ TP reference condition)

Status class	Equal	Threshold	Unequal
High	<=31.25	<=31.25	<=31.25
Good	>31.25 - 42	>31.25 - 42	>31.25 - 50
Moderate	>42 - 62	>42 - 62	>50 - 100
Poor	>62 - 125	>62 - 125	>100 - 200
Bad	>125	>125	>200

Table 2.4 Comparison of total phosphorus concentration ranges for three classification approaches for an hypothetical eutrophic lake (50 $\mu g l^{-1}$ TP reference condition)

Status class	Equal	Threshold	Unequal
High	<=62.5	<=62.5	<=62.5
Good	>62.5 - 83.3	>62.5 - 83.3	>62.5 - 100
Moderate	>83.3 - 125	>83.3 - 125	>100 - 200
Poor	>125 - 250	>125 - 250	>200 - 400
Bad	>250	>250	>400

2.3 Typology

Ecological status represents the relationship between current observed conditions and reference conditions. Reference nutrient conditions are likely to be largely dependent on catchment geology and land-use. The LTT lake typology outlined in Chapter 1 (Table 1.2; Phillips, 2003a) was, therefore, used to derive type-specific reference conditions for this purpose. This typology splits lakes into two depth classes, “very shallow” lakes with a mean depth <=3 m and “deeper” lakes with a mean depth >3 m.

In addition to the lake types outlined by the LTT, for some analyses “deeper” lakes (mean depth >3 m) were split into two depth categories: (1) “shallow” lakes with a mean depth of 3-15 m and (2) “deep” lakes with a mean depth >15 m. These latter depth categories are being used, in addition to the third “very shallow" category (mean depth <3 m), in lake typologies being developed by other EC member states for WFD purposes. It was, therefore, considered important to examine their separate responses from the combined "deeper" GB lake type.

2.4 Reference Nutrient Conditions

Where available, site-specific TP reference conditions should be used, but where these are not available type-specific criteria are needed. Five approaches were used to identify type specific boundary criteria, the first used a statistical distribution of contemporary observational data, the remainder were based on site-specific models:

1) Spatial-state scheme: lower 25th percentile of contemporary observations
2) Diatom-inferred TP, modelled from bottom sediment core samples (e.g. Bennion *et al*., 2001)
3) PLUS export coefficient model (Scotland only) (Ferrier *et al*., 1996)
4) Morphoedaphic index (MEI) (Vighi and Chiaudani, 1985)
5) Land Use Regions (LUR) export coefficient model (England and Wales only) (Johnes *et al*., 1996)

Quality control checks were carried out for identifying and removing "questionable" values including when:
1) Reference TP concentrations higher than recently measured TP concentrations
2) Reference TP >100μg l^{-1} [extremely unlikely without human impact]
3) Current DI-TP more than four times current measured TP

Using the lower 25th percentile of contemporary observations assumes 25% of sites have had only minor influence from anthropogenic nutrient sources. In some cases (e.g. low alkalinity upland lakes) this may represent less lakes than are actually impacted by only 'minor' disturbance, whereas in other lake types (e.g. high alkalinity lowland lakes) this may represent too large a population of lakes that are notionally relatively undisturbed. For this reason, a spatial-state approach was not considered suitable for estimating reference conditions.

The diatom-inferred TP (DI-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. There can also be potential problems with its applicability in some lake types (particularly shallow, high alkalinity lakes) (Sayer 2001).

The PLUS and LUR approaches both indirectly estimate lake TP concentrations, using export coefficient models to firstly estimate TP loads to a water body, and then convert them to a lake concentration using general OECD equations. Both the model structures and coefficients used differ, in particular the PLUS model incorporates a 'transfer coefficient' based on catchment slopes (See Chapter 6 for more detail). Furthermore, the PLUS model hindcasts to 1850 while the LUR model uses a 1930s baseline. The LUR model appeared to have more problems with over-estimation, if compared with current TP concentrations (particularly for shallow, high alkalinity lakes). It has been suggested that the 1930s baseline used in the LUR model may be more representative of a slightly disturbed state, or “good status” (Phillips, 2003c), therefore, it was decided that this approach would not be used to estimate reference conditions.

The MEI uses alkalinity or conductivity data and information on mean depth to estimate TP concentrations resulting from natural loadings in lakes (Vighi and Chiaudani, 1985). This approach is simple and appears to be particularly applicable to identifying reference conditions for cool, temperate lakes.

As it is not possible to validate any method for estimating TP reference conditions, mean site-specific reference conditions were calculated from the three approaches considered most reliable, DI-TP, PLUS and MEI. If all available approaches produced site-specific reference conditions higher than recent measured TP, then the latter was taken as the site-specific reference concentration (e.g. Wast Water).

Type-specific reference conditions were then calculated for tier 1 risk assessment purposes only, based on mean of the mean site-specific values using all three approaches. These are summarised in Table 2.5. Type-specific reference conditions based on 75th percentile of three approaches were examined but were considered less precautionary, and not, therefore, suitable for risk assessment purposes. It is also important to note that so few site-specific values are available for peat, marl and brackish lake ecotypes that type-specific values cannot currently be derived.

Table 2.5 shows that deeper lakes (mean depth >3 m) have consistently lower type-specific TP reference conditions than very shallow lakes (mean depth <=3 m) for all three geological types, the only exception being results obtained using the PLUS model for high alkalinity lakes in Scotland. Table 2.5 also reveals that the spatial state approach was generally less stringent than the modelling approaches (low alkalinity lakes being the exception), highlighting the fact that the majority of lakes in GB are currently in an impacted state, and their current chemistry should not be used to identify reference conditions.

Assuming the type-specific reference concentrations identified represent an EQR of 1, they can then be used to establish status class boundaries based on the “unequal” approach highlighted in Table 2.1. The boundary between high/good status classes was subjectively placed at an EQR of 0.8, whereas the good/moderate, moderate/poor and poor/bad class boundaries were based on repeat doublings of TP concentrations from reference (EQR 0.5, 0.25, 0.125 respectively) (Table 2.6).

Table 2.5 Comparison of reference conditions for total phosphorus ($\mu g\ l^{-1}$) based on five approaches (annual mean values)

		Spatial	Diatom-inferred		PLUS		MEI		LUR model			
		25th Percentile Observed	Mean	75th Percentile	Mean	75th Percentile	Mean	75th Percentile	Mean	75th Percentile	Mean Mean (DI, PLUS, MEI)	Mean 75th Percentile (DI, PLUS, MEI)
Peat	Deeper	24					8	8			8	8
	Very Shallow	20					15	17	12	15	15	17
LA	Deeper	5	8	10	6	6	6	7	7	8	7	8
	Very Shallow	11	13	16			9	11	26	30	11	14
MA	Deeper	12	13	18	9	9	10	13	9	11	11	13
	Very Shallow	43	15	17	19	23	23	25	37	55	19	22
HA	Deeper	34	45	45	20	20	22	25	44	49	29	30
	Very Shallow	42	48	68	13	11	36	41	56	71	32	40
Marl	Deeper											
	Very Shallow								9	9		
Brackish	Deeper											
	Very Shallow						38	38			38	38

Table 2.6 Annual mean TP concentrations ($\mu g\ l^{-1}$) representing type-specific reference conditions and status class boundaries for GB lake types

				WFD type-specific boundary			
		Number of data points	Reference (EQR = 1)	High/Good (EQR = 0.8)	Good/Moderate (EQR = 0.5)	Moderate/Poor (EQR = 0.25)	Poor/Bad (EQR = 0.125)
Peat	Deeper	2	8	10	16	32	64
	Shallow	2	15	19	30	60	120
LA	Deeper	59	7	9	14	28	56
	Shallow	9	11	14	22	44	88
MA	Deeper	29	11	14	22	44	88
	Shallow	18	19	24	38	76	152
HA	Deeper	11	29	36	58	116	232
	Very Shallow	41	32	40	64	128	256
Marl	Deeper	0					
	Very Shallow	0					
Brackish	Deeper	0					
	Very Shallow	1	38	48	76	152	304

2.5 Current Nutrient Conditions

Lake nutrient data were primarily sourced from the GB lakes database, SEPA's loch monitoring programme and CEH datasets. These were reduced down to a selection of 131 lakes (Appendix 2), which had a minimum of four regularly-spaced observations for total phosphorus and chlorophyll$_a$ and passed all quality assurance checks.

Lake status can then be established by calculating an EQR, which represents the relationship between reference conditions and current observed conditions (reference/observed) producing a numerical value between 0 (bad status) and 1 (high/reference status).

If measured TP data are not widely sufficiently available, such as for tier 1 risk assessment purposes, current nutrient concentrations are based on modelled in-lake TP concentrations. Chapters 5 and 6 detail recommended approaches for estimating TP loads. TP load can then be converted to an in-lake concentration using the equations outlined in OECD (1982).

2.6 Recommended guidance for risk assessment

The nutrient (TP) classification outlined is required for assessing the ecological impact in relation to nutrient pressures. It is recommended that for risk assessment purposes, the ecological response is only considered in terms of supporting nutrient conditions within the lake. The response in terms of phytoplankton composition and abundance requiring further development of classification schemes (See Chapters 3 and 4), although even when established, tier 1 risk assessment is still likely to focus on modelled nutrient concentrations. More sophisticated (and data hungry) phytoplankton models (such as PROTECH) being more applicable to higher-tier site-specific risk assessments and in defining a suitable programme of measures for a site

The risk assessment for nutrient concentrations is structured into a number of steps

Step 1. Determine water body type based on LTT typology, as in Table 1.2 (Phillips, 2003a)

Step 2. Ideally identify site-specific reference conditions, using approaches outlined in Section 2.4. If not available, use type-specific reference conditions outlined in Table 2.6.

Step 3. Establish current nutrient conditions, preferably from measured TP concentrations (minimum four, regularly-spaced sampling occasions per year).

Step 4. If measured data are unavailable, estimate current conditions from nutrient load determined by GIS-derived catchment land-use and point-source pressures as outlined in Sections A1.2 and A1.3. TP load can then be converted to an in-lake concentration using the equations outlined in OECD (1982).

Step 5. Calculate an Ecological Quality Ratio (EQR) by dividing reference TP concentration with current (measured or modelled) TP concentration.

Step 6. "Not at risk" status is achieved with an EQR of 0.5 or above. If below 0.5, list water-body as "at risk" of failing to achieve good status.

2.7 Recommendations for Future Development

Further site-specific studies of reference conditions based on historical data, MEI, DI-TP, or export-coefficient models should be carried out for lake types with no, or few, existing studies, in particular for peaty, marl and brackish lakes. The MEI approach may need validation for the latter three lake types.

The boundary values for high/good and good/moderate risk assessment classes are largely subjective. It is, therefore, important that they are validated more closely for a number of test lakes (see Chapter 7). Further validation on specific well-understood case-studies of nutrient enrichment and recovery (e.g. Windermere, Loch Leven, palaeo studies) would also be helpful. Further guidance could also come from the inter-calibration process or Common Implementation Stragey (CIS) ECOSTAT working group.

Any validation of a nutrient classification requires good quality monitored nutrient data, with frequent and evenly-spaced sampling programmes (preferably monthly) and recognised quality assurance checks (see section 2.8 below). One priority for research is an analysis of the uncertainty in classification associated with reducing frequency in sampling to only quarterly sampling. Misclassification could mean that measures to improve status of a water body could be inefficiently targeted, justifying the extra costs associated with increased monitoring effort (ECOSTAT, 2003)

2.8 Data Availability and Quality Assurance

Total phosphorus data from regular lake monitoring programmes are not widely available and only exists for less than 100 sites across GB (mainly Scottish). These data were supplemented by regional surveys carried out by a number of organisations. More extensive lake monitoring programmes that regularly record nutrient and chlorophyll concentrations need to be established. Analytical laboratories need to be able to achieve high analytical accuracy and precision, particularly if dealing with water samples from relatively nutrient poor waters, with TP concentrations <50 $\mu g\ l^{-1}$. Sites with reference conditions <10 $\mu g\ l^{-1}$ will require particularly stringent sampling and analytical procedures to ensure misclassification does not occur through analytical error. Water quality data should also be passed through a number of quality assurance checks:

- At least 4 regularly spaced samples should have been taken throughout the year (i.e. no seasonal bias in sampling).
- To check reliability of values a comparison should be made against previous sampling dates (particularly of a similar seasonal period from previous years)

- An examination of where a site sits on a scatter plot of TP/Chlorophyll$_a$ (See Chapter 3). If a site appears to be an outlier the raw data should be reviewed, or reasons sought.

3 PHYTOPLANKTON ABUNDANCE

Stephen Maberly, Laurence Carvalho and Geoff Phillips

3.1 Introduction

The previous chapter developed a preliminary nutrient classification for risk assessment purposes and examined the risk assessment process based on the in-lake chemical response to catchment nutrient pressures. This chapter examines classification and assessing risk based on the biological response to nutrients, specifically in terms of phytoplankton abundance. How effectively nutrients are transformed into phytoplankton biomass is dependent on a number of different 'sensitivity' factors, such as depth, retention, and colour. In general, the impact of nutrients is likely to be less in deep, rapidly flushed, peaty waters compared with shallow, poorly flushed, non-peat-stained waters. The risk assessment process is carried out in similar steps to the one based on nutrient concentrations but with a number of changes in detail.

3.2 Good Ecological Status

As with nutrient concentrations, the "at risk" boundary is set at an EQR of 0.5. If the EQR is below 0.5, the water-body will be considered "at risk" of failing good status.

3.3 Typology

As already stated, individual lakes have different biological responses to nutrients. This chapter examines how the response differs between the 12 lake types outlined in the LTT typology (Table 1.2). It also considers the "deeper" lake category split into two depth categories: shallow (3-15 m mean depth) and deep (>15 m mean depth) as is the case in many other EC Member States.

3.4 Current Chlorophyll$_a$ Concentrations

Data were collated from all available sources for lakes with measured depth, alkalinity, TP and chlorophyll. Data were primarily sourced from the GB lakes database, SEPAs loch monitoring programme and CEH datasets. These were reduced down to a selection of 131 lakes (Appendix 2), which had a minimum of four regularly-spaced observations of TP and chlorophyll$_a$ and passed all quality assurance checks. Further work is required to assess the increasing precision of annual mean chlorophyll estimates with increasing sampling effort.

If no measured chlorophyll data are available, currently the risk assessment should be based on nutrient concentrations alone. Once general or site-specific TP-chla regression models are developed, current chlorophyll$_a$ concentrations could be predicted from measured or modelled TP_{lake}.

3.5 Reference Chlorophyll$_a$ Concentrations

Site-specific reference chlorophyll conditions are unavailable. Type-specific reference chlorophyll conditions can be derived using two approaches:

1. Spatial-state scheme: lower 25th percentile of contemporary observations
2. Predict from type-specific reference TP concentrations, using type-specific TP-chlorophyll regression models

3.5.1 Overall relationship between TP and chlorophyll

All data were first analysed to obtain an overall relationship between annual mean phytoplankton chlorophyll TP concentrations. The results show the expected increase in phytoplankton chlorophyll$_a$ in response to increasing concentration of TP. Four outlier sites (Albury Mill, Alderfen Broad, Hanmer Mere and Llyn Penrhyn) were removed from the regression analysis as they appeared to be particularly unresponsive to phosphorus in terms of chlorophyll$_a$ concentrations. Further investigation of the reasons behind their poor response needs to be carried out to ensure their removal is justified. Another cluster of Scottish sites all had a measured chlorophyll concentration of 1 µg l^{-1} irrespective of TP concentrations but were not obvious outliers, so could not be justifiably excluded from the regressions (Figure 3.1).

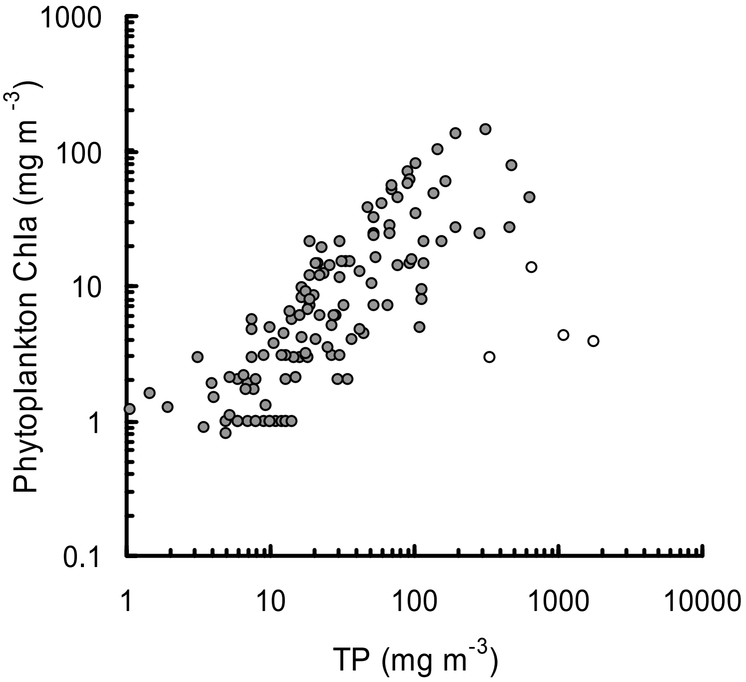


Figure 3.1 Log-Log scatter plots of annual mean concentrations of phytoplankton chlorophyll*a* against total phosphorus for 131 lakes

Note: Outliers removed from the regression analysis are indicated by open circles

The overall regression, after removal of the four outliers, is shown in the first row of Table 3.1. The power-equation (log-log plot) explained 67% of the variation and the regression was highly significant.

Table 3.1 Results of regression of log phytoplankton chlorophyll *a* against log total phosphorus concentration for different lake geology and depth typologies

IC-Geotypes	Depth	Slope	Intercept	Adj R2 (%)	P
ALL	All	0.897 (0.056)	-0.437 (0.084)	67.2	<0.001
Si_LA	All	0.632 (0.126)	-0.246 (0.128)	33.7	<0.001
	Very Shallow	0.840 (0.382)	-0.397 (0.486)	38.9	0.079
	Shallow	0.964 (0.241)	-0.628 (0.251)	35.7	<0.001
	Deep	0.293 (0.174)	+0.039 (0.142)	12.3	0.119
	Shallow + Deep	0.547 (0.142)	-0.184 (0.139)	25.1	<0.001
Si_Ma	All	0.917 (0.137)	-0.378 (0.206)	61.1	<0.001
	Very Shallow	0.956 (0.326)	-0.495 (0.590)	38.7	0.014
	Shallow	1.442 (0.491)	-0.960 (0.656)	48.8	0.022
	Deep	1.245 (1.011)	-0.764 (1.019)	7.9	0.273
	Shallow + Deep	1.415 (0.290)	-0.929 (0.349)	60.3	<0.001
HA_Calc	All	0.858 (0.147)	-0.373 (0.280)	45.3	<0.001
	Very Shallow	0.953 (0.199)	-0.557 (0.385)	42.1	<0.001
	Shallow	0.694 (0.192)	-0.074 (0.349)	57.3	0.007
All	Very Shallow	0.949 (0.119)	-0.516 (0.214)	53.8	<0.001
	Shallow	0.969 (0.109)	-0.554 (0.143)	61.7	<0.001
	Deep	0.417 (0.177)	-0.124 (0.157)	18.4	0.030
	Shallow + Deep	0.805 (0.129)	-0.362 (0.134)	40.0	<0.001

Note: Mean depths of “very shallow” (<3 m), “shallow” (3-15 m) and “deep” (>15 m) lakes. The “Shallow + Deep” category is equivalent to the LTT “Deeper”category in Table 1.2.

Slope and intercept statistics are presented in Table 3.1 for deriving predictive equations (standard error in parentheses), the adjusted R^2 values give the percent variance accounted for, and the significance of the regression is shown in the final column.

3.5.2 Allocation of lakes to lake-types

The 131 lakes dataset collated on current lake conditions (Appendix 2) were sorted into lake types according to the latest guidance from the Lakes Task Team (Phillips, 2003a). Table 3.2 outlines the number of lakes with data for the different LTT types.

Table 3.2 Numbers of lakes in the different LTT geology classes and two depth categories

	Shallow	Deeper	**Total**
Peat	2	2	**4**
LA (Siliceous)	8	44	**52**
MA (Siliceous)	13	16	**29**
HA (Calcareous)	33	11	**44**
HA (Marl)	1	0	**1**
Brackish	1	0	**1**
Total	**58**	**73**	**131**

It is clear from Table 3.2 that sufficient data for developing type-specific TP-chlorophyll regression models are only currently available for shallow and deeper lakes for the three main alkalinity classes (LA, MA, HA). Insufficient data currently exists for brackish, marl and peaty lakes.

Sufficient data also exits for splitting the “deeper” lakes into “shallow” (mean depth 3-15 m) and “deep” (mean depth > 15 m) categories for the low and medium alkalinity classes, although data are limited for deep medium alkalinity lakes (Table 3.3). No data are currently available for deep high alkalinity lakes.

Table 3.3 Numbers of lakes in the different LTT geology classes and three depth categories

	Very Shallow	Shallow	Deep	**Total**
Peat	2	2		**4**
LA (Siliceous)	8	28	16	**52**
MA (Siliceous)	13	9	7	**29**
HA (Calcareous)	33	11		**44**
HA (Marl)	1			**1**
Brackish	1			**1**
Total	**59**	**50**	**23**	**131**

3.5.3 Categorising lakes by geology

Lakes are categories by geology class to test whether or not a better relationship is obtained for the separate categories (Figure 3.2). The regression equations for these three geology classes are shown in Table 3.1 (All depth categories) and the slopes of phytoplankton chlorophyll$_a$ per concentration of TP is shown graphically in Figure 3.3.

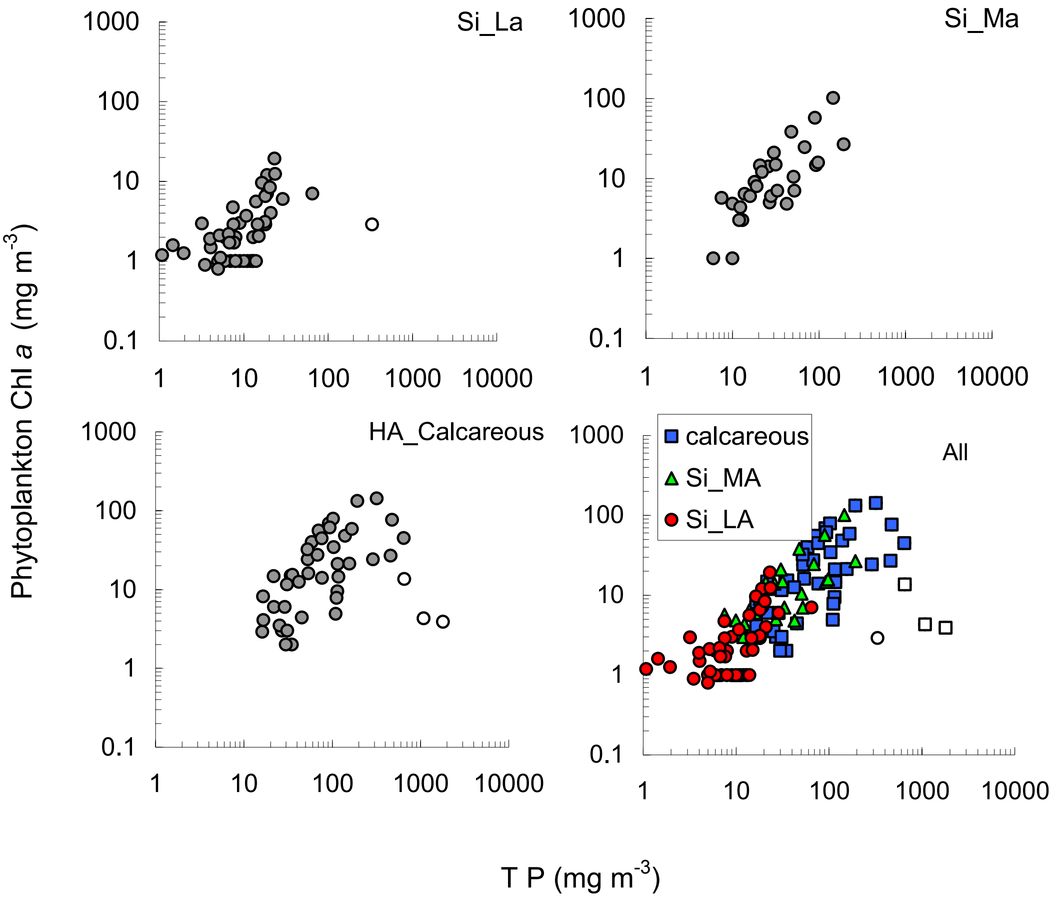


Figure 3.2 Log-log scatter plots of site-average concentrations of phytoplankton chlorophyll$_a$ against total phosphorus for all the data and the three geo-type categories with more than ten sites

Note: outliers removed from the regression analysis are indicated with open symbols

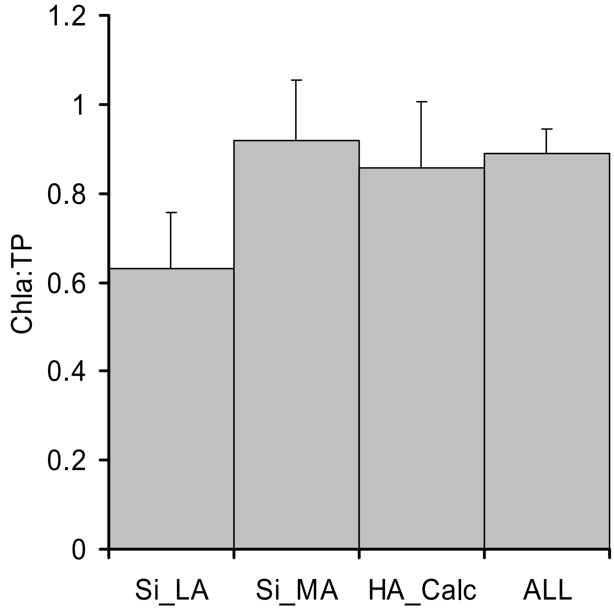


Figure 3.3 Slope of phytoplankton Chla per TP concentration (unitless) for different geology classes, derived from regression analysis in Table 3.1

Note: error bars show one standard error

The lowest alkalinity class appears to have a slightly lower slope- i.e. a lesser amount of chlorophyll$_a$ produced per unit concentration of TP (Figure 3.3). A variance ratio test (F-test) was undertaken to test whether any of these apparent differences were statistically significant. The test is based on whether or not a significantly greater amount of the variation is explained in a regression for each individual category compared to including all the data in one category. The results suggest that separating lakes into different geology classes cannot be justified statistically (Table 3.4). If the lakes are first split into different depth categories then there is a significant difference between the geology categories for the 'shallow + deep' lakes, but not for any of the other depth categories.

Table 3.4 Results of variance ratio test to compare geology categories vs all geology categories combined for the depth categories and for all depth categories combined

Depth category	F	P
Very Shallow	0.090	0.985
Shallow	2.300	0.073
Deep	1.114	0.349
Shallow + Deep	5.713	0.006
All geologies	1.911	0.113

3.5.4 Categorising lakes by depth

A similar exercise was also carried out for the different depth categories. Of the four outliers, three were from the very shallow (<3 m) category and the fourth was from the shallow (3-15 m) category (Figure 3.4). The deep (>15 m) lakes appeared to have a lower slope (Table 3.1, Figure 3.5) i.e. a lesser amount of chlorophyll$_a$ produced per unit concentration of TP, but this was in part because there were no deep lakes with high concentrations of TP or chlorophyll$_a$. The variance ratio tests showed that separating lakes into different depth categories did not produce a significantly better regression (Table 3.5, final row). Furthermore, none of the different geology classes benefited from splitting the data into different depth categories (Table 3.5).

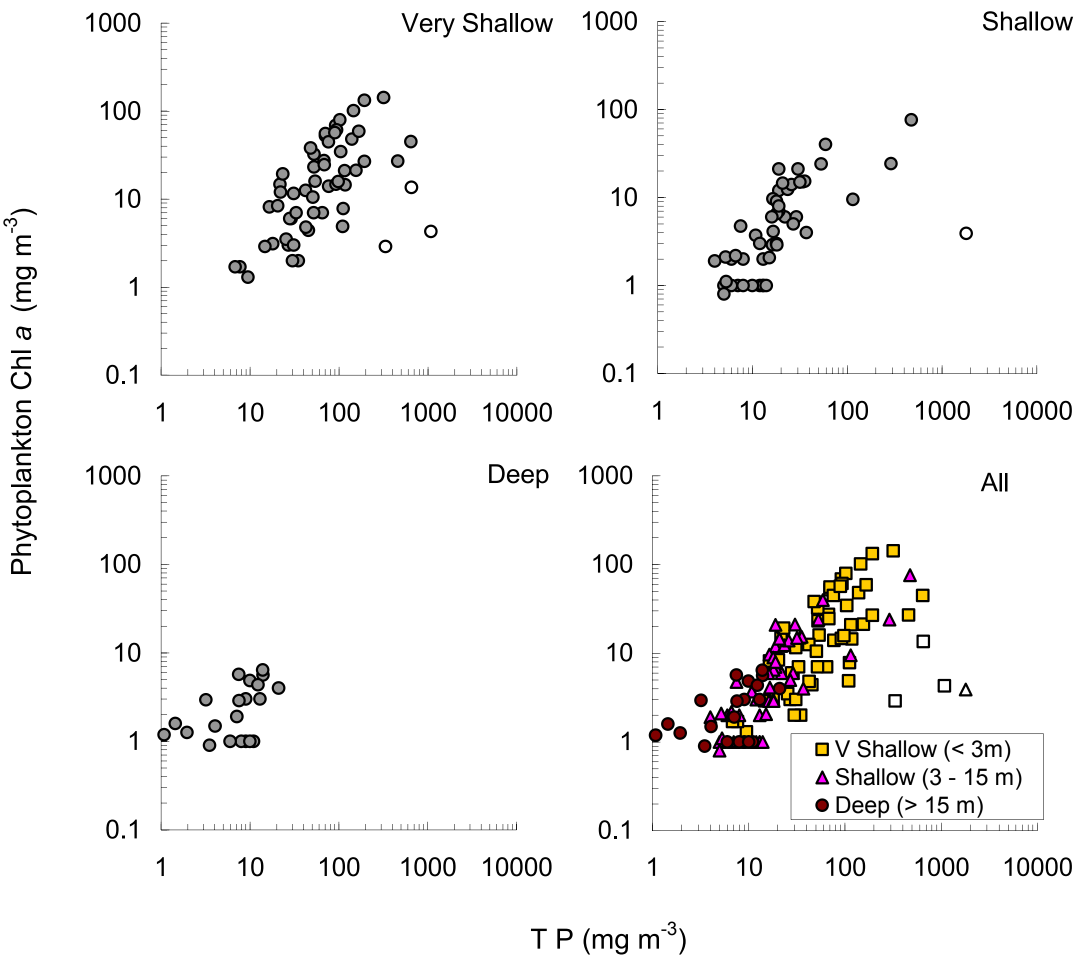


Figure 3.4 Log-log scatter plots of site-average concentrations of phytoplankton chlorophyll$_a$ against total phosphorus for all the data and the three depth categories with more than ten sites

Note: outliers removed from the regression analysis are indicated with open symbols

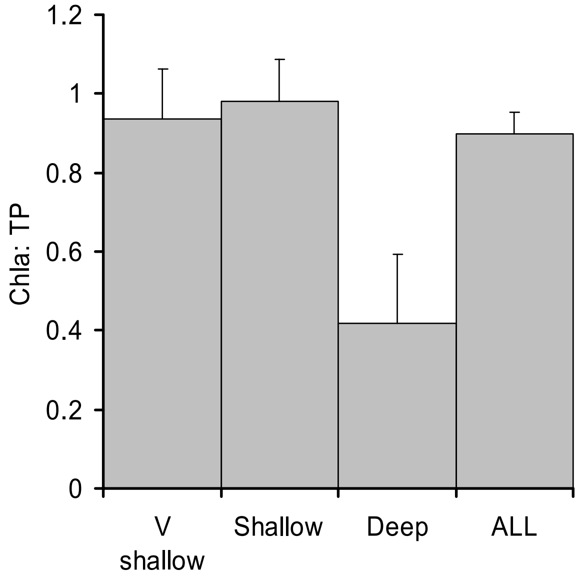


Figure 3.5 Slope of phytoplankton Chla per TP concentration (unitless) for different depth classes, derived from regression analysis in Table 3.1

Note: Error bars show one standard error

Table 3.5 Results of variance ratio test to compare different depth categories vs all depths combined for the geology categories and for all geological categories combined

Geology category	F	P
Si_LA	1.652	0.177
Si_MA	0.734	0.734
Calcareous	0.354	0.704
All geologies	1.574	0.186

3.5.5 Overall relationship between phytoplankton chlorophyll$_a$ and TP

The analyses above suggest that there is little justification for splitting the lakes into different categories and using different relationships between annual mean phytoplankton chlorophyll a and concentration of TP. Out of the array of variance ratio tests conducted, the only one that was significant was for splitting different geologies for 'shallow and deep' lakes (Table 3.4). However, differences among geology categories was not significant for all depth categories.

To examine whether or not the regression for all the data (excluding the four outliers) was reasonable, the intercept and slope in Table 3.1 were compared graphically with other published values. The results (Figure 3.6) show that the equation developed for the UK lakes is very similar to many developed for other lakes worldwide and very similar to the average slope and intercept for all the regressions.

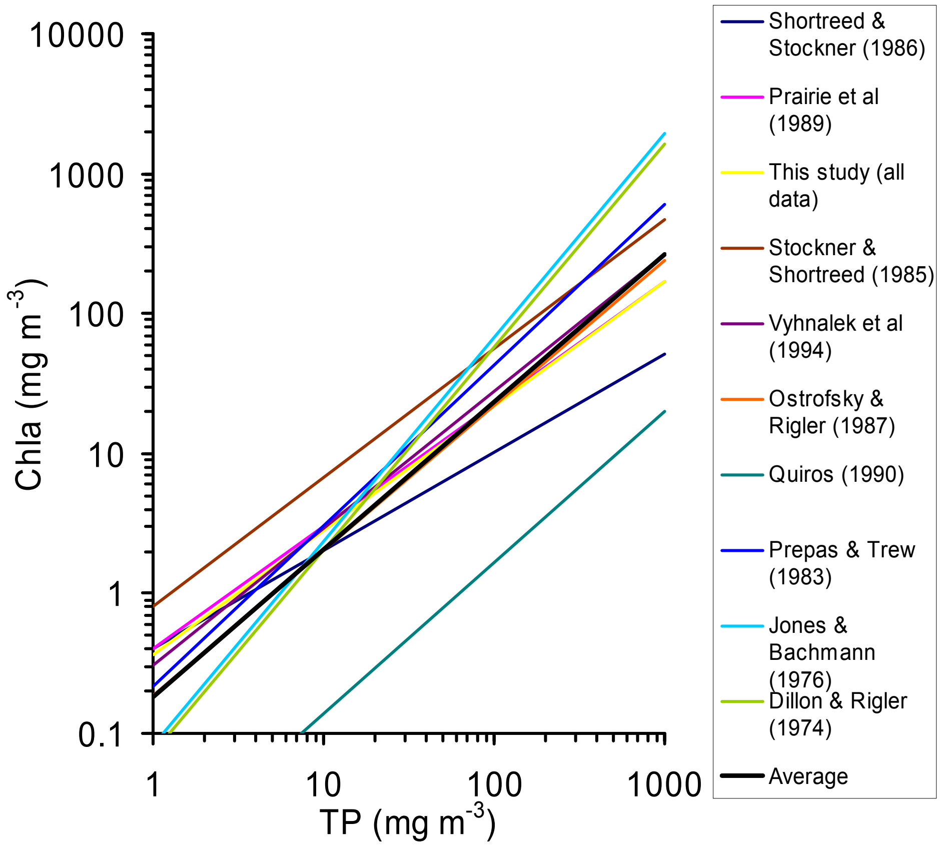


Figure 3.6 Comparison of regression of Chlorophyll$_a$ vs TP in this study with other published values

Simple linear regression equations can be derived for all the lake types listed in Table 5.3 using the slope and intercept values. These equations should, however, only be used to predict chlorophyll$_a$ from TP for the lake types with significant relationships with high adjusted-R^2 values. We recommend that if reference chlorophyll concentrations are required, these should currently be based on the regression equation derived for all GB lakes (i.e. type-specific reference conditions do not currently exist).

We would also recommend that the relationships are, however, explored more through further data collection for all lake types, particularly at the large number of Scottish sites which all appear to have the same annual mean chlorophyll concentration. Consideration should also be given to the use of metabolic models for screening P-limited lakes, as other factors, such as light or nitrogen, may be limiting phytoplankton abundance. This was not possible in this current project due to the lack of physical (transparency) and chemical (winter loadings of P, N and Si) data required in this analysis. The use of multiple regression equations, incorporating factors such as depth and retention time, could also help to differentiate type-specific relationships and could be explored further if an enlarged dataset was available.

As chlorophyll targets are currently derived from TP targets we recommend that the risk assessment process is based on TP concentrations only. If chlorophyll targets can be derived independently from TP (Section 3.6 below) and current measured chlorophyll concentrations exist for a site, then it would be possible to carry out a status/risk assessment based on measured data.

3.6 Defining chlorophyll class boundaries using light attenuation

One possible way of defining boundaries for phytoplankton biomass, independent of TP concentrations, is to express their effect in terms of light attenuation, which will have a knock-on effect on other biota, in particular macrophytes. For example, the depth limit of aquatic macrophytes is usually controlled by light (Spence, 1982; here 'light' refers to photosynthetically available radiation (PAR), 400–700 nm). Laboratory studies in Denmark have shown, at 7 ^{o}C and a 16h light: 8h dark photoperiod, that the average growth compensation point for macrophytes is 6.1 μmol m^{-2} s^{-1}. This equates to about 128 mol m^{-2} y^{-1}, which is about 1.8% of typical surface light in Denmark of 6930 mol m^{-2} y^{-1} (Sand-Jensen and Madsen, 1991). Surveys of macrophyte depth limit as a function of sub-surface light suggest a range of values from 3.3 to 37 % of surface PAR (Chambers and Kalff, 1985). Part of this variation relates to the type of macrophyte (charophyte, bryophyte or angiosperm) and equations have been developed to relate maximum depth to Secchi depth for these three groups (Chambers and Kalff, 1985). Part of this variation also derives from variation with latitude (Duarte and Kalff, 1987) and an inconstant relationship between Secchi depth and light attenuation (since widely available Secchi disc depth has been used to derive light attenuation values) (Middleboe and Markager, 1997). In a recent extensive survey, Middelboe and Markager (1997) quote average percent surface light at depth limits of 2.2% for bryophytes, 5% for charophytes, 16.3% for isoetid macrophytes and 12.9% for elodeid macrophytes, the latter benefiting from an ability to employ shoot extension as a way of 'foraging' for more light higher in the water column.

For the purposes of this example, 5% surface light is used as an indication of the macrophyte depth limit. The depth at which 5% surface light occurs can, with two assumptions, be related to the chlorophyll concentration. The two assumptions concern first, the effect of background attenuation and secondly the chlorophyll-specific attenuation coefficient. In addition to phytoplankton, water itself, humic substances (gilvin) and non-living particles (tripton) will attenuate light (Kirk, 1994). Attenuation by water is of course constant and that of gilvin is likely to be fairly constant for a given lake. Attenuation by tripton, such as suspended solids, is likely to be variable, especially in shallow lakes which are susceptible to wind-induced sediment disturbance.

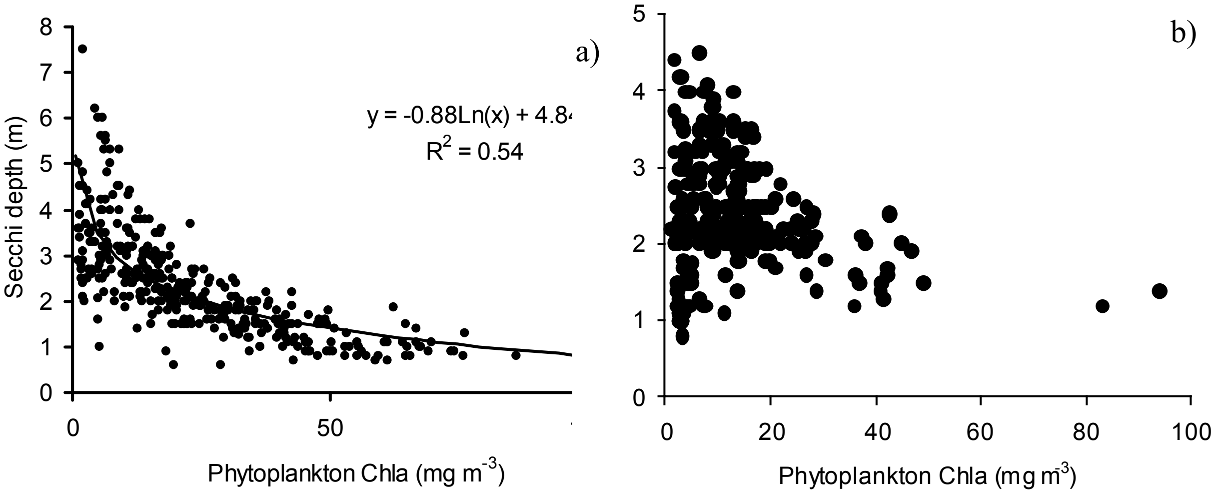


Figure 3.7 Long-term relationships between Secchi depth and phytoplankton chlorophyll$_a$ in a) Esthwaite Water and b) Bassenthwaite Lake

For example phytoplankton chlorophyll$_a$ in Esthwaite Water, with little suspended material, is related to Secchi depth (approximately, Kd = 1.44/Secchi depth), whereas in Bassenthwaite Lake where suspended solids are very variable there is a very weak relationship between Secchi depth and phytoplankton chlorophyll (Figure 3.7). The second assumption is the relationship between the concentration of phytoplankton chlorophyll$_a$ and attenuation. This is not a constant because of variability in the amounts of pigments other than chlorophyll$_a$ and because different 'packing' of chlorophyll in cells of different sizes affects attenuation (sieve effect, see Kirk 1994). Nevertheless, published values for the chlorophyll-specific attenuation coefficient are relatively constrained between about 0.01 and 0.03 m^2 mg^{-1} Chl$_a$, with an average of about 0.02 m^2 mg^{-1} Chl$_a$ (Kirk, 1994).

Light attenuation through a uniform water column follows Beer's law (strictly this refers to a single wavelength):

$$I_Z = I_0 \exp^{-(K_d z)} \tag{1}$$

Where I_z is the light at depth, I_0 is the light at the surface (or more strictly the sub-surface), K_d is the downward attenuation coefficient (m^{-1}) and z is depth (m). This can be re-arranged to calculate the depth at which a particular proportion of sub-surface light is found:

$$z = \frac{\ln I_0 - \ln I_z}{K_d} \tag{2}$$

The downward attenuation coefficient can be split into the sum of a background attenuation (K_B) and that resulting from phytoplankton chlorophyll$_a$ (K_{chl}):

$$K_d = K_B + K_{chl} \tag{3}$$

K_{chl} in turn can be calculated from the product of chlorophyll$_a$ concentration (Chl_a, mg m^{-3}) and the chlorophyll-specific attenuation coefficient (k_c, m^2 mg^{-1}).

The effect of chlorophyll$_a$ concentration on the calculated 5 % depth, is illustrated in Figure 3.8 for different values for K_B and the chlorophyll-specific attenuation coefficient, using equations 2 and 3:

$$z_x = \frac{\ln 100 - \ln(100 - x)}{K_B + (k_c \times Chla)} \tag{4}$$

Where x is the percent surface depth (ie 5% in this example), and z_x is the depth at which that percent of surface light occurs.

For the middle value of K_B and the chlorophyll-specific attenuation coefficient, a chlorophyll$_a$ concentration of 1 mg m^{-3} would allow a 5% depth of about 9.3 m but this would decline to 6.0, 2.3 and 1.3 m for chlorophyll$_a$ concentrations of 10, 50 and 100 mg m^{-3}, respectively (Figure 3.8).

This approach could be used, in conjunction with hyposgraphic data in a given lake, to estimate the proportion of the lake area that could be colonised by macrophytes, although this would not include the shallow water that may be colonisable because of wave action. Hypsographic data is, however, only available for relatively few lakes. A simpler approach would just be a comparison of the maximum colonisable depth with the mean and maximum lake depth since these data are more widely available. A third, even simpler, approach would be to set general acceptable depth limits for macrophyte colonisation, such as 5 m, and this could be directly related to an approximate phytoplankton chlorophyll$_a$ concentration. Table 3.6 shows an example of the maximum annual average chlorophyll$_a$ concentration that would allow colonisation to 1, 3 or 5 m for different groups of macrophyte. There is a wide range: if 1 m was acceptable this would be consistent with up to 76 and 175 mg m^{-3} chlorophyll$_a$ depending on macrophyte group. In contrast this reduces to between 1 and 23 mg m^{-3} if 5 m is set as the limit.

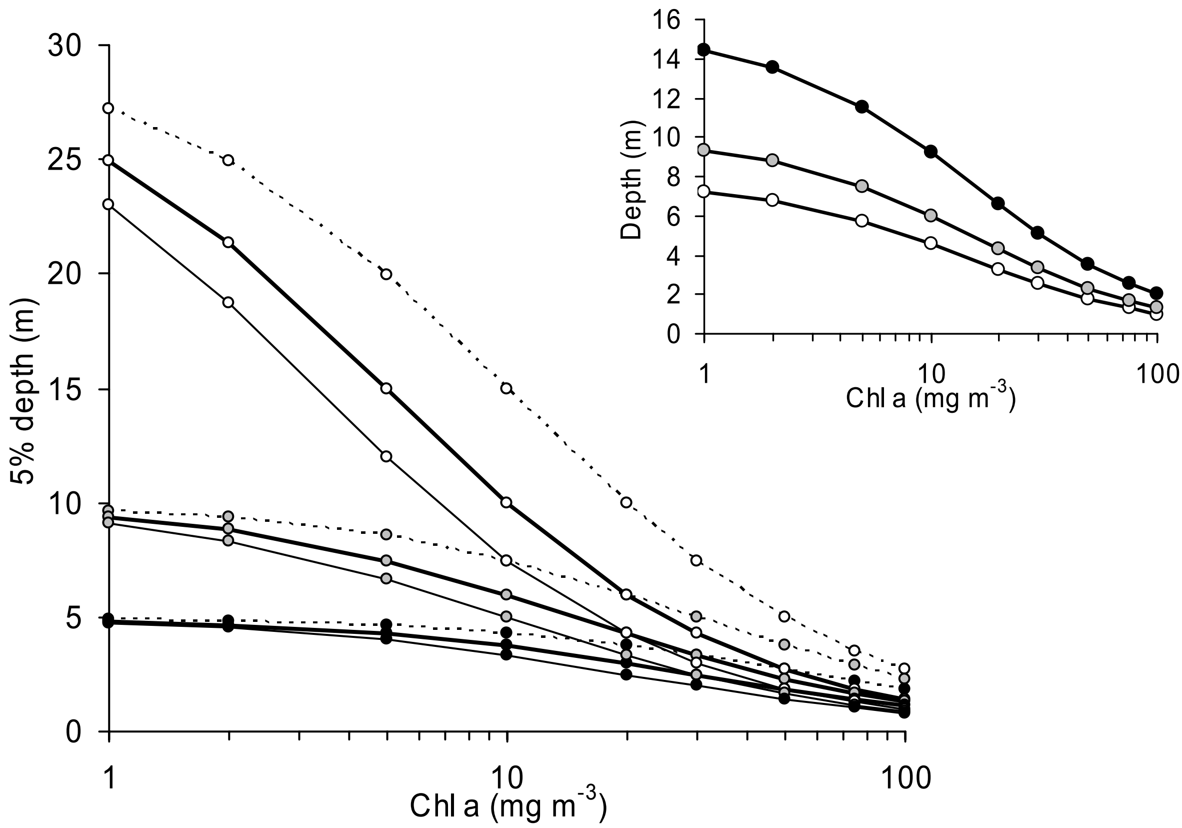


Figure 3.8 The effect of phytoplankton chlorophyll*a* concentration on the depth at which light is attenuated to 5% of the subsurface value

Note: Calculated for three different background attenuation coefficients (m^{-1}), 0.1 (open circles), 0.3 (grey circles) and 0.6 (closed circles) and three chlorophyll-specific attenuation coefficients (m^2 mg^{-1}), 0.01 (dashed lines), 0.02 (heavy lines) and 0.03 (thin lines). The inset shows 1% (closed circles), 5% (grey circles) and 10% depths (open circles) for the middle background coefficient and chlorophyll-specific attenuation coefficient.

Table 3.6 Maximum annual average concentration of phytoplankton chlorophyll$_a$ concentration (mg m^{-3}) that would permit colonisation of different macrophyte groups to three stated depths

		Acceptable depth limit (m)		
Macrophyte group	% depth limit	1	3	5
Bryophytes	2.2	175	49	23
Charophytes	5.0	135	35	15
Elodeids	12.9	88	19	5
Isoetids	16.3	76	15	3

Note: % depth limit data from Middleboe and Markager (1997)

This approach has some promise as it would allow phytoplankton chlorophyll$_a$ concentrations to be set in relation to their ecological effect, in this case the effect on light attenuation with obvious ‘knock-on’ implications for other lake biota such as macrophytes and benthic algae. It also allows some ‘fine-tuning’ depending on a particular conservation end-point, e.g. preservation of charophyte-dominated lakes. Probably the biggest obstacle in this approach would be to decide how to deal with the different background (non algal) attenuation. This will be particularly important in turbid and coloured lakes, but different appropriate background attenuation values could be set for these lake types.

3.7 Data Availability and Quality Assurance

There is still a great need to assess the level of confidence in measured annual mean chlorophyll data with decreasing sampling frequency. This could be carried out through the analysis of intensively sampled chlorophyll records from several lake types (Loch Lomond, Loch Leven, Windermere, OakMere, Rostherne Mere).

Sufficient chlorophyll concentration data were only available for 131 lakes, compared with data on TP from more than 400 sites. More frequent and evenly-spaced monitoring programmes are clearly required to further develop and validate a chlorophyll classification.

Measured data needs to be passed through a number of quality assurance checks:

- At least 4 regularly spaced samples are required throughout the year (i.e. no seasonal bias in sampling), although a clearer idea of frequency of sampling required needs to be investigated further (see above).
- To check accuracy of values carry out a manual check against previous sampling dates (particularly of a similar seasonal period from previous years).

3.8 Recommendations for Future Development

Future development could attempt to classify lakes in other ways apart from depth and geology categories. For example, a ‘metabolic model’ approach could be used to screen lakes that are not primarily limited by phosphorus, but by nitrogen or light, and treat these in a different way. Other possible site-specific sensitivity factors, such as retention time, could also be included.

The effect of light attenuation on macrophyte colonisation is well known and this can be predicted to a reasonable extent from phytoplankton chlorophyll concentrations. The use of macrophyte light requirements to derive chlorophyll targets needs to be developed, tested and validated further. This approach is appropriate for the WFD, where chemical targets are supposed to be derived which “support” biological targets.

Finally, although not developed here, it is theoretically possible to relate phytoplankton biomass to subsequent oxygen depletion in the hypolimnion through a stoichiometric relationship between chlorophyll, carbon and oxygen.

Preliminary analyses (not presented) suggest this could work, but probably only on lakes where hypsographic data are available for estimating the relative volume of the epilimnion and hypolimnion.

4 PHYTOPLANKTON COMPOSITION

Colin Reynolds, Stephen Maberly, Laurence Carvalho and Alex Elliott

4.1 Introduction

4.1.1 Phytoplankton and the Water Framework Directive

The ecological classification of waters, outlined in Annex V of the WFD, is based upon a gradient of human disturbance. It is logical that the pressures and impacts are represented in the quality elements of the monitoring programme. Phytoplankton was chosen as the most sensitive indicator of changing nutrient conditions because of the potential rapidity of population responses and the existing established use of phytoplankton chlorophyll$_a$ as a measure of lake status with respect to nutrient conditions (Carvalho *et al.*, 2002). This is tempered by the fact that, although there are well known general relationships between the biomass of phytoplankton and the supportive capacity of the external nutrient inputs, they are not usually site-sensitive and they are not effective under all circumstances. Furthermore, there is limited knowledge available to predict or interpret change in the species composition of phytoplankton. If use is to be made of the dynamic sensitivity of phytoplankton to environmental change, urgent progress is required in the area of floristic association with the ecological status of water bodies. A series of previous publications has demonstrated the extent to which it is possible to allocate phytoplankton to functional classes (Reynolds 1980, 1984, 2000; Reynolds *et al.*, 2002). The outstanding need is to apply current knowledge to devising a practical application – a “phytoplankton classification tool”.

One objective is to reveal the development of phytoplankton communities in terms of the functional traits that are selected, rather than of individual species. This requires considering species in ecological or ‘functional’ groups rather than taxanomic units. The second objective is to predict, on a probability basis, the functional types that may be represented in the phytoplankton in given types of water body in different seasons and under different nutrient regimes, so that change and susceptibility to change can be measured.

4.1.2 Environmental Sensitivity and Community Assembly in the Phytoplankton

Both comparative and predictive issues relating to phytoplankton often consider one of two major questions. One concerns how much biomass might be reasonably explained or expected, given the nature and location of the waterbody. The second explores the likely species composition, mainly as it affects the perception of quality and safety of the water as a resource, or as an amenity. Progress in these topics formed the substance of two summaries, published recently in the primary literature (Reynolds and Maberly, 2002;

Reynolds *et al*., 2002). Metabolic relationships described in the former of these two papers are invoked to assist the development of a “phytoplankton classification tool”, but, its qualitative basis draws heavily on the latter.

The main conclusion of the paper by Reynolds *et al*. (2002) is that the presence of individual species in a habitat, if not an entirely random occurrence, is difficult to predict, save on the basis of experience and probability. This may be true for other ecosystems, however, there is often a higher level of classification which will predict, with greater certainty, the functional roles and structural adaptations of the main species: some divide producers from consumers, some distinguish those that are invasive from those tolerant of site maturation, whether they share tolerances to acidity or waterlogging or other environmental constraints, and so on. Communities are better, more reliable indicators of habitat conditions than are the presence or absence of individual component species.

Vegetation ecologists even have the advantage of a nomenclature, drawn from the work of the great phytosociologists (Tansley, 1935; Tüxen, 1955; Braun-Blanquet, 1964; see also Shimwell, 1971), for diagnosing and naming the very distinct associations of plant species that constitute vegetation. In essence, these associations are the basic functional units. Each is named after one or two species that are characteristically represented in that particular community-type, using a distinctive binomial construction based on the name of one of them: *Lemnetum minoris* is the association of free-floating mats of duckweed, typically including *Lemna minor* (Haslam *et al*., 1975). The system works because the overlapping requirements of individual species can often be satisfied simultaneously in particular locations, so long as the adaptations of each allow them to tolerate the conditions.

What Reynolds *et al*. (2002) have tried to do is to extend the approach to the structure of phytoplankton communities. Over twenty years a list of 32 associations have been refined, and identified by an alphanumeric codon (**A**, **Sn**, **W2**, etc). The associations have not been grouped by a consistent, or robust, method. There is, however, a consistency in the species clusters from the point of view of morphology, physiological sensitivities and tolerances and frequency of co-occurrence. The scheme has been found to be widely applicable in the UK, in continental Europe, Australia and South America (Padisák and Reynolds, 1998; Beyruth, 2000; Borics *et al*., 2000; Fabbro and Duivenvoorden, 2000; Huszar *et al*., 2000; Melo and Huszar, 2000; Kruk *et al*., 2002).

The coda proposed by Reynolds *et al.* (2002), together with examples of the type and their main adaptive traits, are listed in Table 4.1. Although essentially provisional, it is not expected that this list will alter greatly in the future with further validation, except through subdivision. The coda are listed again in Table 4.2 with symbols indicating tolerance (+) or avoidance (-) of each of a selection of key environmental constraints, so far as is known.

4.2 Developing the Classification Tool

In order to adapt this classification to the requirements of the environment agencies, three main tasks have to be progressed. The first of these is to provide an expanded list of phytoplankton species, each ascribed to its relevant functional association. The second task involves assigning the coda or functional associations according to their likely distribution among a range of water-body types. The issue of prediction of functional types and the sensitivity to change is the third task. The latter is also the most difficult as, even using functional associations, an element of uncertainty persists, representation still fluctuates with season and production is sensitive to event-driven dynamics. The best that can be offered is a probabilistic view of likely representation.

Table 4.1 Trait-separated functional groups of phytoplankton (modified from Reynolds et al., 2002)

Codon	Habitat	Typical representatives	Tolerances	Sensitivities
A	Clear, often well-mixed, base poor, lakes	*Urosolenia*, *Cyclotella comensis*	Nutrient deficiency	pH rise
B	Vertically mixed, mesotrophic small-medium lakes	*Aulacoseira subarctica* *Aulacoseira islandica*	Light deficiency	pH rise, Si depletion stratification
C	Mixed, eutrophic small-medium lakes	*Asterionella formosa* *Aulacoseira ambigua* *Stephanodiscus rotula*	Light, C deficiencies	Si exhaustion stratification
D	Shallow, enriched turbid waters, including rivers	*Synedra acus* *Nitzschia spp* *Stephanodiscus hantzschii*	Flushing	nutrient depletion
N	mesotrophic epilimnia	*Tabellaria* *Cosmarium* *Staurodesmus*	Nutrient deficiency	stratification pH rise
P	eutrophic epilimnia	*Fragilaria crotonensis* *Aulacoseira granulata* *Closterium aciculare* *Staurastrum pingue*	Mild light and C deficiency	stratification Si depletion
T	deep, well-mixed epilimnia	*Geminella* *Mougeotia* *Tribonema*	Light deficiency	Nutrient deficiency
S1	turbid mixed layers	*Planktothrix agardhii* *Limnothrix redekei* *Pseudanabaena*	highly light deficient	flushing conditions
S2	shallow, turbid mixed layers	*Spirulina* *Arthrospira* *Raphidipsis*	light deficient conditions	flushing
S_N	warm mixed layers	*Cylindrospermopsis* *Anabaena minutissima*	light-, N-deficient conditions	flushing
Z	clear, mixed layers	*Synechococcus* prokaryote picoplankton	low nutrient grazing	light deficiency
X3	shallow, clear, mixed layers	*Koliella, Chrysococcus* *Pseudopedinella* eukaryote picoplankton	low base status	mixing, grazing
X2	shallow, clear mixed layers in meso-eutrophic lakes	*Plagioselmis* *Chrysochromulina*	stratification	mixing, filter feeding
X1	shallow mixed layers in enriched conditions	*Chlorella, Ankyra* *Monoraphidium*	stratification	nutrient deficiency filter feeding
Y	usually, small, enriched lakes	*Cryptomonas*	low light	phagotrophs
E	usually small, oligotrophic, base poor lakes or heterotrophic ponds	*Dinobryon* *Mallomonas* *(Synura)*	low nutrients (resort to mixotrophy)	CO_2 deficiency
F	Clear epilimnia	colonial Chlorophytes e.g. *Botryococcus* *Pseudosphaerocystis*	low nutrients	?CO_2 deficiency high turbidity

		Coenochloris *Oocystis lacustris*		
G	Short, nutrient- rich water columns	*Eudorina* *Volvox*	high light	nutrient deficiency
J	shallow, enriched lakes ponds and rivers nutrient poverty	*Pediastrum, Coelastrum* *Scenedesmus* *Golenkinia*	high light,	settling into low light
K	short, nutrient-rich columns	*Aphanothece* *Aphanocapsa*	deep mixing	
H1	dinitrogen-fixing Nostocaleans	*Anabaena flos-aquae* *Aphanizomenon*	low nitrogen low carbon,	mixing, poor light, low phosphorus
H2	dinitrogen-fixing Nostocaleans of larger mesotrophic lakes	*Anabaena lemmermanni* *Gloeotrichia echinulata*	low nitrogen	mixing, poor light,
U	summer epilimnia	*Uroglena*	low nutrients	CO_2 deficiency
L_O	summer epilimnia in mesotrophic lakes	*Peridinium* *Woronichinia* *Merismopedia*	segregated nutrients	prolonged or deep mixing
L_M	summer epilimnia in eutrophic lakes	*Ceratium* *Microcystis*	very low C,	mixing, poor stratification light
M	dielly mixed layers of small eutrophic, low latitude	*Microcystis* *Sphaerocavum*	high insolation	flushing, low total light
R	metalimnia of mesotrophic stratified lakes	*P. rubescens* *P. mougeotii*	low light, strong segregation	instability
V	metalimnia of eutrophic stratified lakes	*Chromatium,* *Chlorobium*	very low light, strong segregation	instability
W1	small organic ponds	Euglenoids, *Synura* *Gonium*	high BOD	grazing
W2	shallow mesotrophic lakes	bottom-dwelling *Trachelomonas*	?	?
Q	small humic lakes	*Gonyostomum*	high colour	?

Table 4.2 Responses to habitat properties of functional groups of phytoplankton (from Reynolds *et al.*, 2002)

Variable(1): Codon	h_m < 3	I^* <1.5	θ <8	[P] $<10^{-7}$	[N] $<10^{-6}$	[Si] $<10^{-5}$	[CO_2] $<10^{-5}$	f <0.4
A	-	?	+	+	+	+	-	-
B	-	+	+	+	-	-	-	-
C	-	+	+	-	-	-	?	-
D	+	+	+	-	-	-	+	-
N	-	-	-	+	-	+/-	-	?
P	-	-	-	-	-	+/-	+	+
T	-	?	-	+/-	-	+	?	+
S1	+	+	+	-	-	+	+	+
S2	+	+	-	-	-	+	+	+
S_N	+	+	-	-	+	+	+	+
Z	+	-	+	+	+	+	?	-
X3	+	-	+	+	-	+	-	-
X2	+	-	+	?	-	+	?	-
X1	+	-	+	-	-	+	+	-
Y	+	+	+	-	-	+	?	-
E	+	+	+	+	-	+	-	-
F	+	-	+	+	-	+	-	-
G	+	-	+	-	-	+	+	+
J	+	?	+	-	-	+	?	-
K	+	?	-	-	-	+	+	?
H1, H2	+	-	-	-	+	+	+	+
U	+	-	?	+	-	+	-	+
L_O	+	-	-	+	-	+	-	+
L_M	+	-	-	-	-	+	+	+
M	+	-	-	-	-	-	+	+
R	+	+	-	-	-	+	?	+
V	+	+	-	-	-	+	-	-
W1	+	+	+	-	-	+	?	-
W2	+	+	+	-	-	+	?	?
Q	+	+?	+?	?	?	+	?	?

Table Notes

Entries in table are to denote tolerance (+) or no positive benefit (-) of the environmental condition set.; "+/-" is used to denote that some species in the association are tolerant; "?" denotes that tolerance suspected but not proven.

(1)- Variables signified are: depth of surface mixed layer (h_m, in m from surface); mean daily irradiance levels experienced (I^*, in mol photons m^{-2} d^{-1}); water temperature (θ, in ^{O}C); the concentration of soluble reactive phosphorus ([P], in mol l^{-1}); the concentration of dissolved inorganic nitrogen ([N], in mol l^{-1}); the concentration of soluble reactive silicon ([Si], in mol l^{-1}); the concentration of dissolved carbon dioxide ([CO_2], in mol l^{-1}); and the proportion of the water processed each day by rotiferan and crustacean zooplankton (f).

4.2.1 Site classification

The classification of water bodies proposed by the agencies adopts a hierarchical approach, distinguishing rivers from deep and shallow lakes and then according to base status (calcareous through to acidic and humic). As far as phytoplankton is concerned, rivers and shallow lakes offer similar habitats.

The phytoplankton classifications are considered separately for each lake type. Lake types follow a simplified version of the GB lake typology (Phillips, 2003a; Section 1.1.2), with codifications (1-4) distinguishing along a gradient of alkalinity (equivalent to dystrophic, LA, MA, HA) and separating shallow and deeper systems (S, D) (Table 4.3).

Table 4.3 Site classification categories first stage separation of lakes according to depth and base status

Depth	Lakes in which <50% is <5 m in depth	Lakes in which > 50% is <5 m in depth
Lakes of zero alkalinity, strongly acidic, humic and dystrophic.	1D	1S
Lakes of low alkalinity (0 – 200 mequivs/m^3) Slightly acid to quasi-neutral.	2D	2S
Lakes of moderate alkalinity (200 – 2000 Mequivs/m^3), quasi-neutral to high pH	3D	3S
Lakes of high alkalinity (> 2000 mequivs/m^3), Buffered at pH ~ 8.5 and precipitating marl	4D	4S

It is well understood that phytoplankton abundance may vary with season (especially with respect to day length and light dose) and with variations in the nutrient supply. The functional associations of phytoplankton are also differentially sensitive to these factors, therefore, they need to be recognised within the structure of the tool. A water-body may support *Asterionella* in the spring, *Dinobryon* and *Ceratium* in the summer, then *Tabellaria* or *Cosmarium* in the autumn, all in response to seasonal or short-term environmental fluctuations, but with no significant variation in the basic ecological quality of the water.

The scheme developed here incorporates a strong seasonal component, to accommodate anticipated changes in phytoplankton composition, independently of longer-term variations in nutrient resources. The predictions are based on "seasonal blocks", roughly corresponding to the spring bloom (say February/March, in shallow lakes, or March/April in deeper lakes), the early summer (immediately after the onset of thermal stratification, where relevant, when insolation and resources are relatively abundant), the mid-summer (June/July) and late summer (August/September) and finally the period of autumn mixing (September to November).

A similar blocking is proposed for the nutrient condition. The scheme for describing trophic state is based essentially on the OECD (1982) classification. The modifications attempt to avoid the assumption that metabolism is an exclusive or continuous function of phosphorus load. The logic of the separation is that the assembly of pelagic biomass can proceed to the stoichiometric limit of the capacity of the nutrient resource with the least biological availability (limiting nutrient) and at a rate that may eventually depend on the rate of its supply. In both cases, either phosphorus or nitrogen is usually supposed to exert the critical limitation. As capacity is raised by an increased supply of phosphorus, the dependence of growth rate may switch increasingly to the carbon supply; in deep or well-mixed lakes, the dilution of light also places a limit on the rate of carbon fixation.

Because algae are unequal in their affinity for nitrogen, phosphorus and carbon or in their ability to harvest light, the selection of functional groups in particular lakes is biased by their sensitivity to small variations in the factors closest to becoming limiting. The richer the lakes are in phosphorus, the more likely is the algal community to be subject to the assembly constraints determined by the carbon dynamics of the system and by the way the available light is distributed and absorbed.

The second level of site classification attempts to capture these critical dimensions. The basis for separating five metabolic categories are summarised in Table 4.4.

Thus, for any class of lake (1D, 2D, 2S, etc.,) we may visualise a matrix of possibilities invoking annual (seasonal) cycles and interannual variations in the dynamics and metabolism of carbon (including progressive eutrophication or its reversal).

Table 4.4 Trophic categories of lakes adopted in the classification with associated characteristics of maximum biomass, total phosphorus content and an indication of the period when available phosphorus may regulate the further assembly of phytoplankton biomass

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

4.2.2 The phytoplankton template

The essential, underpinning idea for the phytoplankton classification is that the seasonal and trophic changes in the phytoplankton can be predicted, at least to functional group, given the basic morphometric and alkalinity state of the water body in question. It must, however, be accepted that even this prediction is subject to habitat variability, weather and the source and size of the inocula at the base of the developing populations. The design of the tool accommodates all these sources of variation through a representation of "adjacency". The finished tool is a reference set of 25 templates, for each of the given lake states, in which a consistent arrangement of functional groups is repeated. The arrangement, detailed in Table 4.5 reflects group coherences to temporal and trophic influences. The groups more indicative of oligotrophic conditions are shown at the top, grading to those of progressively more enriched conditions towards the bottom. Groups tending to appear early in the annual sequence are shown towards the left, grading to those appearing later in the year.

Table 4.5 Template of phytoplankton functional groups

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

The arrangement of the templates for a particular lake type then follows quite naturally (Table 4.6). The rows correspond to the five trophic categories; the columns follow the time line from the spring bloom (in February/March in shallow lakes, or March/April in deeper lakes), the early- (May), mid- (June/July) and late- (August/September) summer periods, through to the late-season mixing (in reality, this can occur at any time in the second half of the year when the water is still warm but the days are shortening.

Table 4.6 The arrangement of phytoplankton templates under each category of lake type (1S, 2S, 2D, etc.)

Trophic categories	Spring bloom	Apr-May	Jun-Jul	Aug-Sep	Autumn mixing
U					
O					
M					
E					
H					

Functional group representation

The final step in assembling the tool is to ascribe to each tropho-temporal phytoplankton template (within each lake type) a measure of the probability of the representation by one or other functional groups. The use of probability is critical to the correct deployment and use of the phytoplankton tool. On the premise that any plankton alga can be relied to grow wherever and whenever it can, it would be unsafe to predict presence or absence as a criterion for judging the ecological state of the lake. If an unexpected representation was detected, the quality of the habitat could be misinterpreted. On the other hand, in the more unlikely event of a failure to detect an expected representation, a highly erroneous interpretation could be reached. It would also be risky to attach too much significance to a single phytoplankton sample; it is as much the changing pattern of representation through time that is preferable for interpretation. Nevertheless, the arrangement of functional groups on the template is intended to aid the determination of significant variation because nearest neighbours are likely to be co-habitants in the water body – a block of representative groups, rather than any single one of them, is likely to respond to a change in habitat quality.

Four categories of probability of representation are used and shown on the templates by the use of colour, as shown in Table 4.7.

Table 4.7 Scheme of colouring template cells to show probability that the functional group will be represented in the phytoplankton

	>75%
	50 – 75%
	25 – 50%
	< 25%

The ascriptions of functional groups to each lake type- temporal-trophic cell is made on the basis of the relative group-characteristic tolerances to low water temperatures, poor insolation, low nutrients, low carbon dioxide availability, short days and high light dilution. The effects of heavy grazing disfavouring any group, however, are ignored. Further work is required to validate the attribution of the functional groups to these templates.

Each template for an individual lake type is, therefore, populated within itself by 25 trophic-temporal templates and each template is coloured to show the probability, against fertility and through time, of any given algal functional group being represented in the water. Illustrated examples for all the lake types are provided in Appendix 3.

4.3 Phytoplankton in Rivers

Developing the phytoplankton tool for the consideration of river phytoplankton requires a slight variation of approach. This must take account of the recent advances in the understanding of the factors governing the dynamics and composition of potamoplankton (Reynolds and Descy, 1996; Gosselain, 1998; Reynolds, 2001), particularly the sensitivities of potamoplankton to river flow. The typical composition of the phytoplankton of rivers is drawn from a much shorter list of candidate species than is that of lakes, but there are few, if any, differences in the chemical requirements, or sensitivities, of river species. The principal drivers are related, directly or indirectly, to the unidirectional passage of water. The age of the water in the river is crucial to the chance of its colonisation by planktonic algae and the number of population doublings that can be accommodated before the water hits the sea. The true time of travel of suspended solids is greatly influenced by river storage (conservatively 20 – 40% of the volume of a river is not moving at all) and channel retentivity is quite well predicted by gradient and sinuosity. The slope of the channel is also a primary factor in accumulating sediment. Where this occurs, most of the finer materials are liable to resuspension under high discharges. Turbidity is usually the major factor impeding the growth of phytoplankton in large lowland rivers (usually between third and sixth order in UK; larger rivers outside the UK, such as the Nile, Mississipi, Yangtse, mostly do not support phytoplankton in their lower reaches).

The nutrient load in most UK rivers is usually in excess of the capacity of light-limitation to support phytoplankton growth. No clear instance of phosphorus limitation of plankton growth in lowland reaches has been established in the UK, although there are suggestions that diatoms do sometimes deplete the silicon to fairly low concentrations.

In consequence, the generalisation may be made that in UK rivers, the phytoplankton supportive capacity is set by light rather than nutrients, its achievement depends on the time of travel and the river-specific impacts of gradient and sinuosity. As a result, the “seasonality” relates most to discharge and its ramifications: the onset of “bloom” periods usually depend on falling discharge and decreasing sediment loads. The greatest theoretical potential for phytoplankton production is realised at prolonged low flows, although frequently in these conditions, the plankton may be sparse and the water very clear, thought to be attributable to the filter-feeding activities of benthic unionids.

On the other hand, chlorophyll$_a$ in the Rivers Thames, Severn and, occasionally, the Trent has been observed to have risen to concentrations that would be considered hypertrophic if they occurred in lakes (>100 µg l^{-1}, sometimes >350 µg l^{-1}). These rivers regularly carry more than 0.5 mg l^{-1} P and 3.5 mg l^{-1} N, more than enough to sustain such levels of chlorophyll. Most other UK rivers are just too short and/or steep to support phytoplankton production of this magnitude. The growth potential of high and excess nutrient concentrations tends not to be expressed until the water is impounded in main-stem or pumped riverside storages.

River phytoplankton is dominated by fast-growing, fast replicating nano- and small microalgae rather than the slower-growing, temperature sensitive Cyanobacteria. Potamoplankton assemblages are also biased towards species whose propagules remain in the river, its sediments and bottom vegetation. Centric diatom species, especially *Cyclotella* and *Stephanodiscus* spp. (in continental rivers, *Aulacoseira*, *Thalassiosira* and *Skeletonema* spp are common) are favoured over pennate diatoms. Most of the green algae that are prominent in rivers (especially species of *Scenedesmus* and *Pediastrum*) lead a mero-planktonic life cycle. Abrupt onset of high-clarity and low-flow conditions can, however, still favour an element of the unpredictable, with stochastic, weed-like invasions of unicellular nanoplankters of the X1 and X2 functional groups, potentially able to build up large populations. The behaviour is predictable but the identity of the participating species is not.

4.3.1 River templates

Although the fundamental principles that underpin the distribution of phytoplankton in rivers are prevalent, the issue of base status is less prominent than in lakes (because rivers cross geological and pedological boundaries rather easily) and depth is largely irrelevant (all UK rivers are shallow according to the lake depth typology). The seasonality reduces to the bloom period of reducing flow, summer base flow and prolonged low flows when the river turns into something more closely resembling a linear lake. The finalised template for rivers, therefore, considers only one river type (roughly equivalent to a very shallow, medium-high alkalinity lake, 2S or 3S) under three flow conditions, rather than the five seasons considered for lakes.

Fitting the trait selected groups then depends upon the same principles as for lakes, except that the influence of flow on slow-growing species is applied pragmatically. The representation in the U, O and M categories change little from that of a corresponding 2S or 3S lake. Only in the E and H categories (practically the only ones we can observe in the UK), the retentivity and water clarity factors operate strongly in selecting for D, X, J and S1 traits; cryptomonads (Y) also figuring (Figure 4.1).

As is true of lakes, the templates are probabilistic, no more predicting what will be found than they predict what will be excluded. In the case of rivers, there are two additional complexities. Rivers that run through lakes or reservoirs or receive inflow from bank-side storage, can often carry limnetic phytoplankton some distance downstream (including bloom-forming Cyanbacteria, dinoflagellate crops and large numbers of pennate diatoms). These algae may not grow well but they are present and, for some considerable downstream length, may affect the perceived water quality. The second complexity is the behaviour of the river under prolonged low flows, when it becomes increasingly lake-like, amenable to the growth of more types of algae, and is similarly likely to be subject to intense molluscan (or even crustacean) grazing down to high water clarity. The present scheme does not anticipate these outcomes.

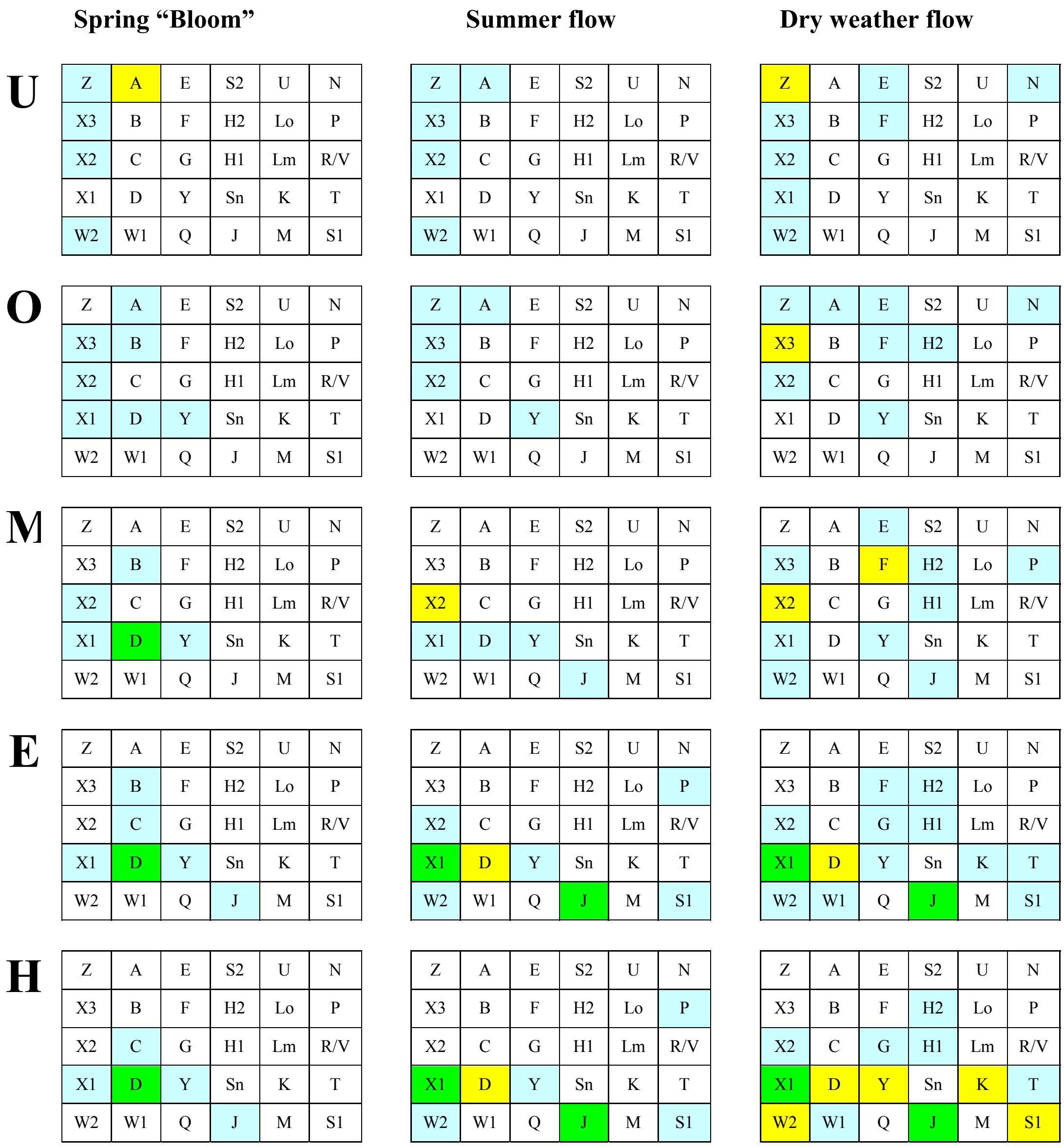


Figure 4.1 Template of phytoplankton functional groups in the slow-moving river type

4.4 Development and application of a working classification tool

In order to develop a simpler working tool for immediate application, a modified approach is proposed. The information in each of the five seasonal boxes (“Season 1”, “Season 2”, etc.) for each of the five trophic categories (“U”, “O”, “M”, etc) and for each of the eight lake types (1S,1D, 2D, etc) has been summarised into a binary-coded score, based on prescribed probabilities (1= predicted presence at greater than 25% probability, else blank). The 200 summary lines can be arranged on an Excel spreadsheet, the rows being identified by a template code (from 1SU1, meaning “lake type 1, shallow, ultraoligotrophic, season 1”, through to 4DH5, for “lake type 4, deep, hypertrophic, season 5), the columns being headed by the functional coda (A, B, H1, H2 etc.) (Appendix 4).

Once a phytoplankton count has been completed and each common species ascribed to its functional group, the count can be compared with the binary-coded scores in the templates. This can be done quantitatively by means of a similarity test.

4.4.1 Approach

A semi-automated Microsft Excel spreadsheet has been produced to calculate the similarity between the functional groups of phytoplankton within a sample and the similarity to the 200 simplified binary-coded templates. On one sheet within the workbook, data are entered either as species names or as phytoplankton codes (based on the Windermere database phytoplankton codes) and these are translated to functional groups within a lookup table. In a second sheet, the similarity of the functional groups to each of the 200 sets is calculated using Sorensen’s similarity index based on presence or absence (Equation 1). In this trial phytoplankton classification tool, no attempt has been made to filter out species at low abundance or to use a similarity index based on abundance or frequency, although these could be developed in subsequent versions.

$$Sorensen's\ index = 100 \times \left(\frac{No.\ of\ \ common\ species}{0.5 \times \left(No.\ species\ in\ set + No.\ species\ in\ data \right)} \right) \qquad (Equn.1)$$

The match between the data and the sets is assessed in three different ways:

1) The set with the closest match is identified. This produces a single matching set but it is possible that other sets are present with the same or very close, similarity (Excel only identifies one match even if there are several with the same similarity).

2) The similarity indices are averaged over all the possible sets which are mesotrophic or eutrophic, or shallow or deep, etc. One disadvantage of this approach is that the average similarity calculated for a given attribute will be very broad and include average sets with low similarity (for example a ‘shallow’ category will be averaged over the entire trophic range).

3) The third approach is a hybrid of the above two. The similarities are calculated over the top X percent ranked similarities, where X can be set by the user on the sheet. The exact value of X does not appear to alter the results greatly and currently the 95 percentile (top 5%) has been adopted.

A second advantage of the two averaging approaches is that the trophy, alkalinity, depth etc can be calculated as an average based on either the complete data or the top X% similar phytoplankton sets. This produces a more robust estimate of the best fit of the phytoplankton data to the phytoplankton sets.

4.4.2 Example application to Windermere long-term phytoplankton data

An example to test the sensitivity and validity of the Phytoplankton Classification Tool was carried out using data from the South Basin of Windermere. This lake basin has experienced a well-documented increase in nutrient enrichment followed by a decline following the introduction of tertiary P-stripping in 1992. The data analysed were the first phytoplankton sample in April of 1950, 1960, 1970, 1980, 1990 and 2000. The simplified binary-coded approach was applied, cell count data not being utilised, and there was no attempt to remove rare species from the analysis.

The analysis in Figure 4.2 shows a clear trend in the trophy index from mesotrophy in the 1950s to 1980s, followed by a dramatic increase in trophy in 1990 and a return to meso-eutrophic conditions in 2000. There is a very strong and statistically significant ($P = 0.001$) relationship between the trophy index and the winter maximum soluble reactive phosphorus (SRP) concentration (Figure 4.3), which is an encouraging first trial.

The alkalinity index varied between 2.3 and 3.4, which equates to an alkalinity in the categories around the 200 m.equiv. m^{-3} border which is similar to the alkalinity of the lake. The depth index varied between 1.2 and 1.7 (i.e. placing the lake on the border between ‘shallow and ‘deep’ which is lower than the actual mean depth of the South Basin of 16.8 m. Finally, the season index varied between 2.2 (about weeks 11-20) and 4.1 (about weeks 31- 40). The first week of April is about week 14, so an expected season index would be 2.

The analysis suggest that the phytoplankton functional groups predict a relatively static alkalinity, mean depth and season for the South Basin of Windermere, but there is clear evidence that the trophy index has changed. The reliability of this index was cross-checked by relating the tropic index derived from phytoplankton functional groups to the winter concentration of soluble reactive phosphorus in each of the six years (Figure 4.3).

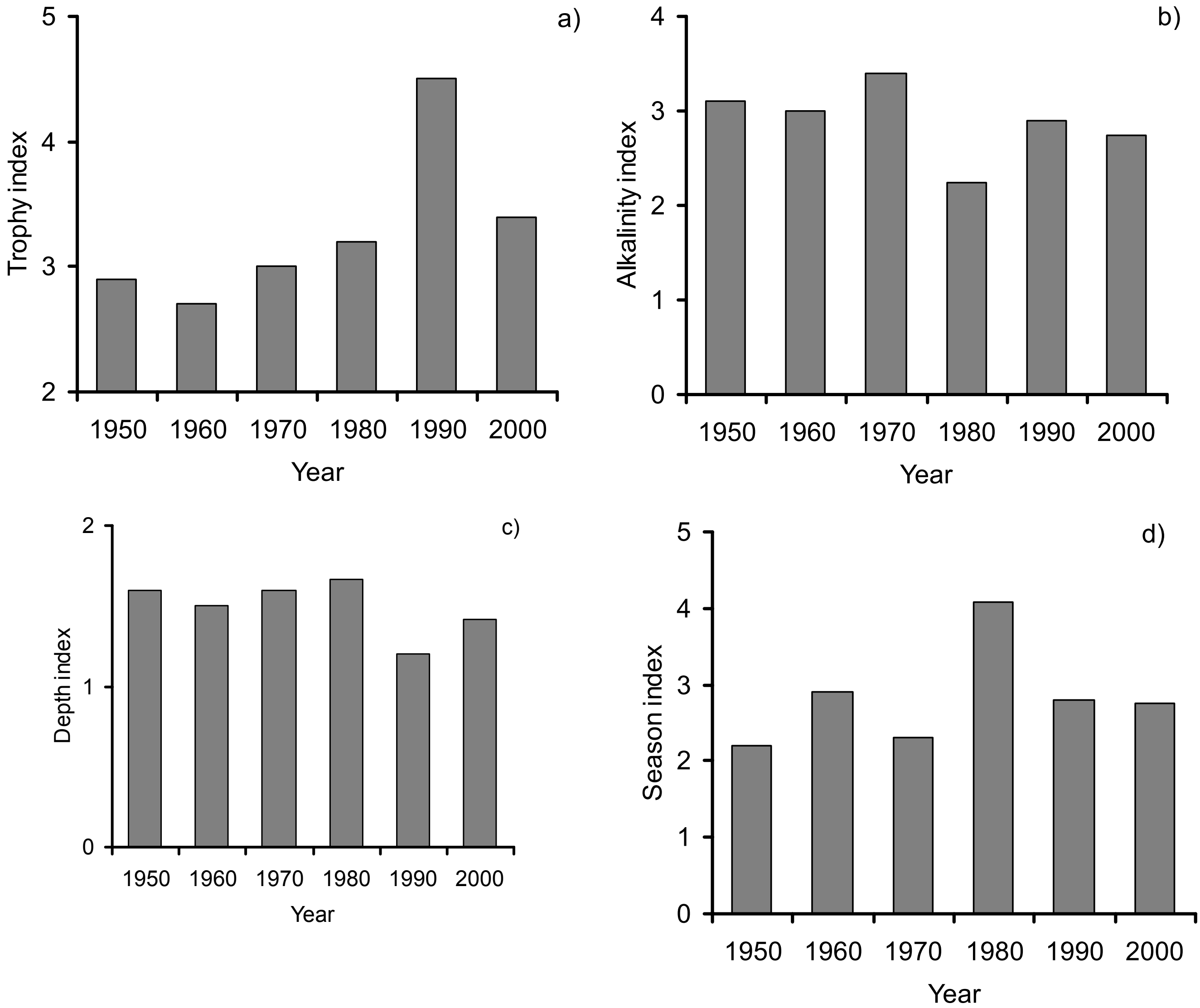


Figure 4.2 Calculation of lake characteristics in Windermere South Basin based on phytoplankton functional groups in April

a) Trophy index (1 = ultra-oligotrophic, 2 = oligotrophic, 3 = mesotrophic, 4 = eutrophic, 5 = hypertrophic); b) alkalinity index (1 = < 0 mequiv. m^{-3}, 2 = 0 – 200 mequiv. m^{-3}, 3 = 200- 2000 mequiv. m^{-3}. 4 = > 2000 mequiv. m^{-3}); c) depth index (1 = <3 m mean depth, 2 = >3 m mean depth); d) season (1 = weeks 1-10, 2 = weeks 11-20, 3 = weeks 21-30, 4 = weeks 31-40, 5 = weeks 41-52).

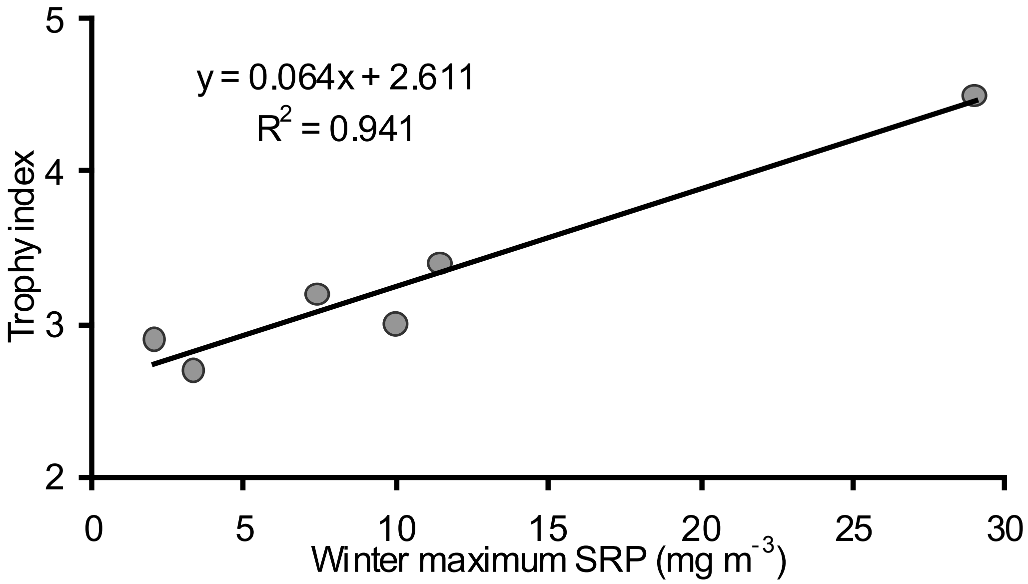


Figure 4.3 Relationship between the Trophy index and winter maximum SRP in the South Basin of Windermere

4.4.3 Calculating EQR values

A provisional approach to calculating an EQR from these similarities is outlined below and illustrated with the data for the South Basin of Windermere. The average similarity for the top 5% of sets is calculated. For example, in 1950 this was 3DM2, i.e. the lake alkalinity class was 3 (200 – 2000 mequiv. m^{-3}), the depth class was D (> 3 m), the trophic group was M (mesotrophic) and the season was 2 (spring). Using a reference condition for this lake of oligotrophic, derived from an average site-specific reference TP concentration of 9 mg m^{-3}, a similarity is automatically calculated between the average set, 3DM2 and 3DO2 (i.e. M is replaced by O). In the example for 1950 the similarity is 44.4% giving an estimated EQR of 0.44.

An example of changing EQR for the South Basin of Windermere using the example dataset from 1950 to 2000 is shown in Figure 4.4. It, however, only shows a slight resemblance to the known nutrient enrichment history of Windermere.

An alternative way of calculating an EQR is to use the trophy score in relation to the reference trophy score. For example, in 1950 the trophy score based on the top 95 percentile values was 2.9, the reference trophy score would be 2 (oligotrophy), so the EQR would be 2/2.9 = 0.69. Figure 4.5 shows that the EQR-trend using this system more closely matches the known history, with a maximum EQR in 1960 of 0.74 before the first sewage treatment works was constructed in the catchment and a minimum in 1990 (EQR = 0.44) at the peak of TP loading, and an improvement by 2000 to 0.59 following the introduction of tertiary treatment on the two main sewage works on Windermere. This alternative approach to calculating EQR , therefore, appears to be preferable, but clearly requires further refinement and validation. Note that if the reference condition had been set at 3 (mesotrophic), the EQR values for 1960, 1990 and 2000 would have been 1.11, 0.67 and 0.88 respectively, highlighting a problem of only having a limited number of fixed integer reference values (1 through to 5).

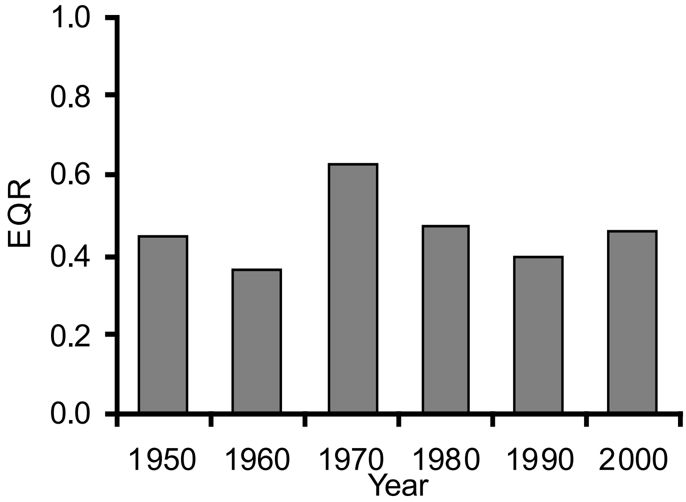


Figure 4.4 Example time-course of EQR based on the South Basin of Windermere phytoplankton data in April for the six identified years, using the average percentile similarity (top 5%) approach

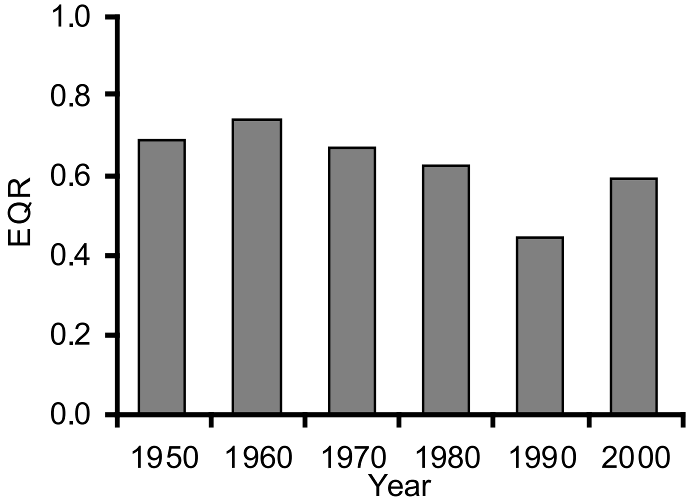


Figure 4.5 Example time-course of EQR based on the South Basin of Windermere phytoplankton data in April for the six identified years, using the trophic similarity approach

4.4.4 Instructions for using the phytoplankton composition tool

1. A summary of these notes is present on sheet '0' of the Microsoft Excel workbook.
2. Sheet '1, sets' contains the phytoplankton functional groups expected in the 200 different lake types. This sheet does not need to be altered, but is needed in subsequent calculations.
3. Sheet '2, lookup' contains lookup tables to convert species names or Windermere phytoplankton computer codes to Functional Groups.
4. Sheet '3 Data entry' here the Windermere algal codes are entered or copied into the grey area (column A) and the corresponding Functional Group is automatically entered into the adjacent column (column B). Note, do not delete the complete sets of functional groups in the yellow area above (currently the rows are hidden)- this is needed in the calculation. Take care to ensure: (i) that the data entered is correct (does not contain data from a previous calculation for example) and (ii) that the lookup codes have been calculated for all the data.
5. Sheet '4 Analysis' contains the bulk of the calculations. First, update the pivot table by placing the cursor within the pivot table and pressing the red exclamation mark to refresh the table. This transfers the data from the data entry sheet and recalculates the similarities.
6. In the name box (lower left toolbar area), select 'RANK' this automatically selects the range of lake sets and calculated similarities. Copy this and Pastevalue it to the indicated yellow range at F31 (only need to place cursor in F31).
7. While the data is still selected, Data Sort the list. The ranks will be automatically calculated according to the percentile range for calculation in S2 (e.g. 95 to calculate from the top 5% similarities).
8. Select 'OUTPUT' from the name box to automatically select the output data. These can then be transferred to an output sheet using Copy followed by Paste and Pastevalue.
9. At present a 'Reference' trophic type can be entered in cell S1 (e.g. U, O, M, E or H) and the percent similarity is calculated in the blue boxes based on the closest match or the average of the top X percent or average of all the data. The average of the three is calculated in the dark blue box below labelled 'EQR' on a scale of 0 to 1. Note that by changing S1, the EQR updates automatically. At present this approach does not seem to be very successful (see text above).

4.4.5 General application of the classification tool

The classification tool has two principal uses. It can be used to check that the phytoplankton in a given water body is approximately of the quality expected for a given lake type. More specifically, it can be used as a sensor of temporal change in the trophic status of a given lake type, particularly if the phytoplankton assemblage consistently represent similar directional change in several seasonal samples. In theory, it could also be used to compare contemporary algal quality with the predicted compositional trends in the same lake but subjected to reduced/increased nutrient loads either that occurred in the past, or following future lake management.

In terms of number of samples required for tool application, clearly the more seasonal samples that indicate similar trends in trophic change, the more confidence can be assumed in the classification. Kadiri and Reynolds (1993) suggest that very little detailed information is required to establish the quality of a lake. Three or four samples during a single calendar year would generally be sufficient to establish the type of plankton to which the system was host. From the point of view of robust assessments, samples taken during the spring before, or very soon after, the onset of seasonal stratification (say, April), should catch the spring diatom bloom and the most predictable response to potential growth conditions.

4.5 Recommendations for Future Development

This chapter has shown that the development of an ecological classification scheme based on phytoplankton composition, to its specific application as a management tool, has no theoretical obstacles. Its development into an unambiguous and reliable management tool may be rather less smooth and needs considerable development. A number of issues require further work:

- Validation of faithfulness of traits/species associated with phytoplankton functional groups and, potentially, development of new functional groups associated with poorly studied lake types (e.g. peaty lakes)
- Validation of functional group associations with lake type, season and trophic conditions
- Development of classification tool to incorporate maximum potential in phytoplankton count data. In particular, use of the number of representative taxa present for each functional group and/or the relative abundance of functional groups in count data
- Estimates of error in classification and potential of misclassification
- Further consideration for developing a reference EQR scale that is continuous rather than nominal.
- Establishment of taxonomic quality assurance procedures for UK laboratories involved in method development and future phytoplankton monitoring

5 POINT-SOURCE NUTRIENT PRESSURES

Linda May and John Hilton

5.1 Introduction

In addition to the nutrient supply to lakes from diffuse sources within the catchment, many lakes also receive a significant proportion of their phosphorus (P) load from point sources. These discharge either directly into the lake itself, or indirectly into one of its feeder streams. In the context of this project, the main point sources of P have been taken to include industrial discharges, sewage treatment works, septic tanks, small package treatment plants and cage fish farms. Methods of assessing input from these sources are described and evaluated, especially in relation to data availability.

5.2 Sewage Treatment Works

5.2.1 Method 1

The most accurate method of estimating the P load to a lake from sewage treatment works (STWs) within its catchment would be to calculate the load from P concentration and rate of flow measurements for each of the STW effluents. Unfortunately, this type of data is not held by the UK Environment Agencies at a national or local level. It was not, therefore, possible to apply this method to a Tier 1 risk assessment approach. This approach should, however, be used in higher tier, site-specific, risk assessments. It should be noted, however, that most recorded flow data is likely to be held by local Water Companies, so collating these data could be a very time consuming task.

5.2.2 Method 2

A simple way of estimating the P load from the total human population within a lake catchment is that used by Bennion *et al* (2002). This method is based on population density data with national coverage from which the total number of people resident in any catchment can be derived. The P load to each lake is then estimated by applying a *per capita* P export coefficient to the resident population. In the first instance, it is assumed that this entire P load comes from sewage-related sources.

This method produces sensible estimates, on the whole, and has the major benefits of being easy to understand and using easily available national coverage data sets. However, it has a number of potential sources of error. Firstly, the accuracy of the P load estimate is totally dependent on the export coefficient used (see later); secondly, it does not distinguish between people connected to mains sewerage systems and those served by septic tanks or small package treatment plants; thirdly, it does not take into account the fact that P removal rates vary enormously from one STW to another, depending on whether secondary or tertiary (P removal) treatment has been applied (Table 5.2); it does not take into account trade effluent discharges made to STWs; finally, no allowance can be made for the transfer of sewage from one catchment to another for treatment.

5.2.3 Method 3

An estimate of the P load from each STW, alone, can be made using the design (or consented) Population Equivalent (PE) value for each works. In England and Wales, this information is held in the Discharge Consent Register, but is not widely held on the Environment Agency's Water Information Management System (WIMS). The latter contains the following information shown in Table 5.1.

Table 5.1 Information on sewage treatment works in the Water Information System (WIMS)

Region	= Environment Agency region
STW name	= name of STW
Consent number	= reference number of discharge consent
Dilution_R	= average ratio by which the receiving river water dilutes the discharge
NGR	= grid reference of discharge point
Receiving_W	= name of receiving water, where available
FW_S_E	= description of receiving water (freshwater, saltwater, or estuary

Additionally, consented dry weather flow and maximum daily flow fields are available for some STWs.

SEPA holds similar information for STWs in Scotland, in many cases with additional information on actual PE connected and level of treatment for each works. However, it was not possible to compile these data for the whole of Scotland within the timescale of this project.

In the current project, the above information was entered into a geographical information system (GIS), which allowed the number, size (expressed as a 'population equivalent' value) and location of all STWs within each catchment to be determined. From this, the total population equivalent (PE) value for all STWs discharging into the river system within any given catchment could be estimated. The P load to each lake from upstream STW discharges was estimated from this value by applying a *per capita* P export coefficient (see below).

This approach also has some limitations. Firstly, the load estimate is, again, totally dependent on the export coefficient used. Secondly, the PE value of any individual STW does not, necessarily, reflect the human population discharging to the STW. There are two main reasons for this. One reason is that, to aid

STW design, industrial effluent that discharges to a STW is also ascribed a PE value, i.e. the theoretical number of people who would contribute the same load to the STW as the industrial effluent. As industrial discharges seldom have the same chemical characteristics as sewage discharges, simply multiplying the PE equivalent of an industrial effluent by a *per capita* P export coefficient determined for sewage may give incorrect results. The other reason is that the most readily available PE value for a given works describes the design capacity of the works and not, necessarily, the number of people connected which may be higher or lower. There is a particularly marked disagreement between these values in situations where a works has been built to accommodate large numbers of visitors during the short tourist season, e.g. in the Lake District. Here, the PE value may be 3 times higher than the resident population that it normally serves (Table 5.4). Data for Scotland clearly show that, although actual and design capacity PE has, on average, an almost 1:1 relationship, the relationship between these figures for individual works may be very different (Figure 5.1). Thirdly, in rural catchments, an estimated P load based on the PE of all sewage works within the catchment does not include the large number of people who live in dwellings that are not connected to the main sewer system, but rely on septic tanks for the disposal of household wastes. In such cases, the P load from these tanks must be calculated separately (see Section 5.3). Finally, the method does not properly reflect the effect of any P removal processes operating within the sewage treatment system. This is especially true of tertiary treatment, which may significantly reduce P discharge.

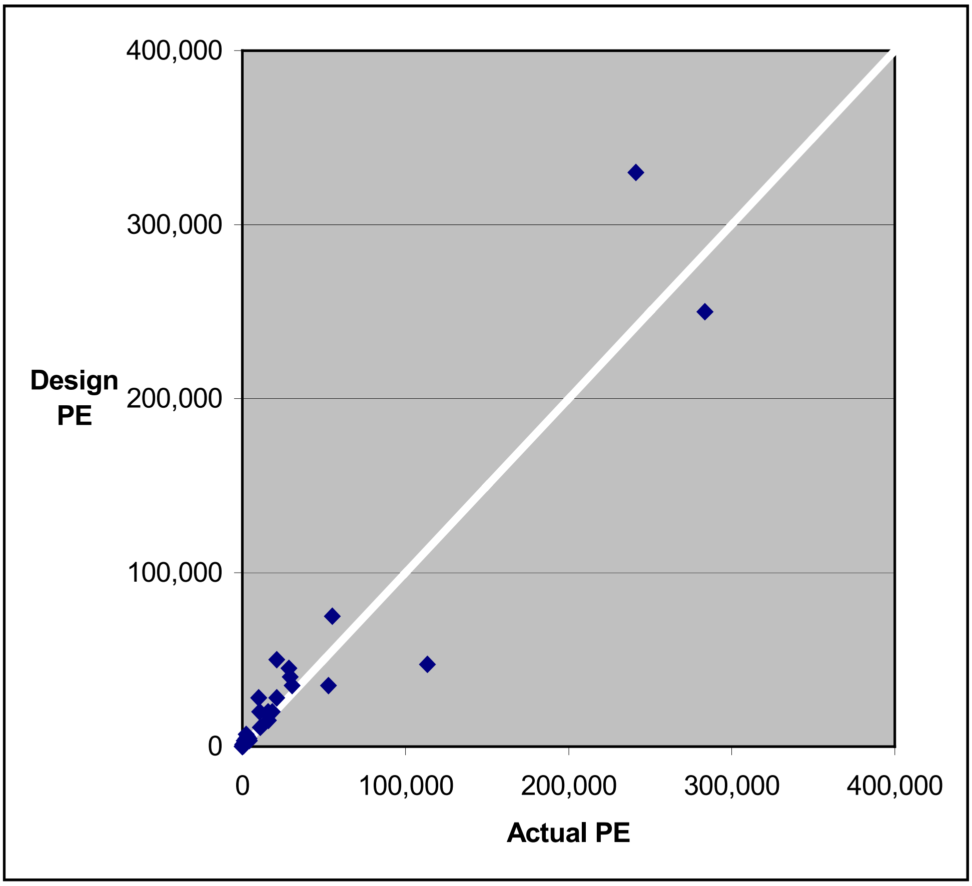


Figure 5.1 Relationship between actual and design (or consented) population equivalent values for sewage treatment works in the west of Scotland

Note: 1:1 relationship is indicated by the white line

5.2.4 Choices of export coefficients

There are, basically, two different approaches to estimating an appropriate P export coefficient for use in Methods 2 and 3, above. One is to estimate the total P load *per capita* input to a sewage works and multiply this by the reduction in P through each treatment process. The second is to estimate, directly, the *per capita* P load discharging from a STW, using either in-stream or discharge pipe measurements of flow and concentration. Investigating the latter approach during this project, using data currently held by CEH, gave values ranging from 0.14 kg capita^{-1} y^{-1} to 1.55 kg capita^{-1} y^{-1}. It seems unlikely that this reflects the true situation at the works that were investigated. Part of the explanation for this wide range of values is probably the use of design capacity PE for these calculations, as this may not reflect the actual number of people connected to the works (see above). Differences in the way the data had been collected and analysed may also be partly to blame. For example, some data were close interval while some were collected very infrequently; some had been collected from an effluent pipe and some from in-stream monitoring; some values had only been determined for soluble reactive phosphorus (SRP) requiring an assumption to be made about SRP:TP ratio before *per capita* TP export from the works could be estimated. In addition, there was variation among the works in their level of wastewater treatment, although detailed information on this was not readily available during the data analyses. During this study it became clear that more detailed research was needed to determine why these figures differed so markedly from one STW to the next. In the meantime, it was decided that the first of the approaches given above should be used in the Tier 1 risk assessment process. This is described below.

Deevey and Harkness (1973) estimated a *per capita* P export value for raw household waste (i.e. untreated waste water) of 1.2 kg *capita*$^{-1}$ y^{-1}. This was based on an estimated 0.5 kg P *capita*$^{-1}$ y^{-1} from human waste and 0.7 kg P *capita*$^{-1}$ y^{-1} from soap and detergents. This value is consistent with current expert opinion, although it should be noted that it may now need to be revised downwards to take account of the increasing use of P-free detergents since the 1970s.

Some reported estimates of dissolved phosphorus removal rates at different stages of sewage treatment are given in Table 5.2. Assuming that the percentage reduction quoted is for that stage only, i.e. the cumulative reduction is obtained by multiplying the inlet dissolved P load by each reduction factor in turn, then it is possible to estimate the annual *per capita* P load after secondary treatment as described below.

If the majority of the human waste component of the wastewater entering the works is assumed to be in solid form, then all of this component will be removed during the primary and secondary treatment processes. If we, then, assume that the majority of the 0.7 kg P *capita*$^{-1}$ y^{-1} from soap and detergents is in dissolved (or hydrolysable) forms, then primary treatment will remove about 7.5% and secondary treatment will remove about 30%. This suggests that the total *per capita* annual discharge from domestic waste following secondary treatment is about 0.44 kg P *capita*$^{-1}$ y^{-1}, which is similar to the value proposed by Johnes (1996) of 0.38 kg P *capita*$^{-1}$ y^{-1} following secondary treatment. The latter estimate, in conjunction with estimates of diffuse P load, has been shown to produce results that are comparable to ‘measured’ in-river P loads derived from in-river flow and concentration measurements in a multiple point-source – diffuse-source catchment (Hilton *et al*, 2002). *Per capita* P export coefficients for septic tanks (see Section 5.3) range from 0.26 - 0.4 kg *capita*$^{-1}$ y^{-1}. These figures are of similar size to that estimated by Johnes (1996) and that estimated above from the values given by Deevey and Harkness (1973). As a result, we suggest that for general risk assessment purposes, an annual *per capita* P load of 0.4 kg is used in the current risk estimates.

There is great confusion in many publications as to whether total P or dissolved reactive P is being assessed. In addition, because many of the published measurements of P loads to sewage treatment works were carried out before 1980, they do not appear in computer-based literature searches and many have been missed by recent reviews. As a result, there is a wide range of values in the literature and considerable lack of clarity as to how they were all obtained. For this reason, we recommend that the Agency commissions a more thorough review of P export coefficients for domestic waste, and of P removal rates during the various stages of sewage treatment within a works, than was possible here. Such a review should combine a detailed literature review (paying particular attention to literature from the 1960-1980s) with an analysis of data from the current Water Company monitoring programmes. Once complete, the Agency should insist that the results are published in the refereed literature so that the basis of future load estimates can be reliably referenced.

Table 5.2 Phosphorus removal rates from wastewater for different levels of treatment within a sewage treatment works

Treatment Level	Phosphorus removal	Reference
Primary	5-10% (dissolved)	SCOPE 1999
Secondary	20-40% (dissolved)	SCOPE 1999
Tertiary: chemical	≥95% (dissolved)	SCOPE 1999
Tertiary: chemical	70-90% (total after secondary treatment)	Cooper *et al*. 1995
Tertiary: biological	40-85% (dissolved)	SCOPE 1999
Tertiary: biological	80% (total after secondary treatment) easily achievable	Brett *et al*, 1997

P removal at STW

None of the methods described above take account of the significant reduction in P loads from STWs brought about by the application of P removal processes (tertiary treatment) at some works. These were identified and mapped for England and Wales from data supplied by the Agency detailing where a P removal plant had been installed as a result of the Urban Waste Water Treatment Directive (UWWTD). This allowed us to identify which works in each catchment had only secondary treatment and which had tertiary treatment for P removal (e.g. Barton Broad, Bassenthwaite, Loch Leven and Windermere catchments).

There are, potentially, two ways of adjusting the P load to a lake for the P removal processes at any plant with UWWTD consents for P. These are

1) replace the population-based load estimate after secondary treatment with an estimate obtained by multiplying the consented maximum concentration by the average annual discharge flow or,
2) assume 80% removal of the load estimated for secondary treatment.

The UWWTD sets average levels of P concentration in the discharge (2 mg l^{-1} for works with a PE of ≥10,000 and 1 mg l^{-1} for works with a PE of ≥ 100,000). The most appropriate of these concentration values, multiplied by the average annual discharge volume, could be used to provide an estimate of P load for any given works. However, it should be noted that the average annual discharge data required for this calculation are not readily available, but are usually estimated as 1.25 times the dry weather flow (Paul Simmons, personal communication). It would be possible to use the maximum design flow, instead, but these data are not readily available, either, and also generally relate to a storm flow rather than the average annual flow. So, this approach is not a practical option for use at the national scale. Because the PE equivalent for a STW often bears no resemblance to the census population in a catchment (Table 5.4) it is difficult to ascertain what proportion of the sewage from humans in a catchment has been treated. As a result, we recommend that the effect of currently applied, or proposed phosphorus removal from a STW is not included in the assessment until tier 2, when individual catchments are assessed using more detailed data than can be used at the national (tier 1) level. However, if an approximation is required at the tier 1 level then it would only be possible by applying an 80% reduction to the estimated total P load from ALL humans in the catchment. This approach has the potential to greatly over estimate the effects of P removal and we do not recommend it.

5.3 Septic Tanks

Septic tanks are the most commonly used method of disposing of human waste in rural parts of Britain. The effluent from these tanks usually drains to a soakaway where, it has always been assumed, the nutrients that it contains are dissipated into the soil (Harper, 1992). However, there is now mounting evidence that this is not true. All septic tanks contribute, to some extent, to the pollution of nearby surface and underground waterbodies and many older installations present a particular problem because they discharge directly into watercourses (Frost, 1996). Although there have been no intensive studies on the use and effectiveness of septic tanks in Britain, a recent survey of 24 septic tanks within the Lough Leane catchment, in Ireland (LLCMMS, 2000), suggests that the impact of these systems on the environment may also be higher than expected due to lack of maintenance. Most of the tanks surveyed were over 20 years old, and consisted of a one-chamber blockwork design with the sewage discharge going to a soakaway. Inspection showed that accumulated sludge had reached the level of the outlet in 88% of these tanks, suggesting that tanks were only rarely, if ever, de-sludged. In addition, 20% of the tanks failed to meet the capacity requirements for the dwellings that they served and 47% failed to meet the minimum required in percolation tests. It is unclear to what extent these results reflect the situation in rural areas of England, Scotland and Wales because there have been no similar studies in these areas.

Table 5.3 Site-specific estimates of phosphorus loads to lakes from septic tanks within their catchments

Waterbody	Estimated P load to waterbody from septic tanks (tonnes)	Proportion of external P load to waterbody attributable to septic tanks (%)	Reference
Bassenthwaite Lake	2.3	18.0	May *et al*. (1998)
Loch Earn	0.07	1.2	Weller (2000)
Loch Flemington	0.02	17.5	May *et al*. (2001)
Loch Leane	1.5	12.0	LLCMMS (2000)
Loch Leven	1.5	10.0	Frost (1996)
Loch Ussie	0.03	22.0	May and Gunn (2000)
Lough Erne		12.0	Foy (pers. comm.)
Lough Neagh	56.0	14.0	Foy (pers. comm.

There are only a few, site-specific, studies on the impact of septic tanks on standing waters within the UK and Ireland. Most of these have been desk studies based on population estimates and *per capita* phosphorus (P) export coefficients derived from the literature. Only that on Lough Leane has involved field measurements and monitoring. The results of all these studies are shown in Table 5.3. They suggest that P discharges from septic tanks may account for 10-20 per cent of the P load to some lakes from external sources.

There are usually many septic tanks scattered across rural catchments in Britain. In terms of assessing their impact on water quality, these individual tanks are usually viewed collectively as a diffuse source of pollution. However, for the purposes of this report, septic tanks are considered as individual, point sources of pollution that can be scaled up to the catchment level using a *per capita* P export coefficient and a suitable multiplication factor that reflects the number of individuals served by septic tanks within the catchment.

It is very difficult to provide an accurate P export coefficient for septic tanks that is universally applicable in every situation. This is because P output from such systems depends upon a wide range of factors including the quality of the influent wastewater (e.g. whether phosphorus-based detergents are permitted) and the age, management and efficiency of the tank itself. Also, the fact that many septic tanks serve properties that are occupied for only part of the year must also be taken into account.

The extent to which P-laden effluent from these systems reaches adjacent water bodies is also difficult to assess. It is affected by local conditions that include the nature, length and slope of any drainage culverts, the type and depth of soils (Ellis and Childs, 1973; Ebers and Bischofsberger, 1987), the depth of the water table, and proximity to and size of the nearest watercourse (Harper, 1992; Ellis and Childs, 1973). For older installations, whether or not the surrounding soils have already become saturated with P is also an important factor (Frost, 1996).

In spite of the limitations outlined above, some coefficients for the export of P from septic tanks to surface waters have been published for the UK. These include 0.33 kg P *capita*$^{-1}$ ann^{-1} for septic tanks within the catchment of Loch Leven, Scotland (Frost, 1996), 0.4 kg <u>SRP</u> *capita*$^{-1}$ ann^{-1} for septic tanks within the catchment of the River Main, Ireland (Foy and Lennox, 2000) and 0.26kg <u>MRP</u> *capita*$^{-1}$ ann^{-1} in the Lough Leane catchment. Although these values provide little more than a general overview of the magnitude of the P load to surface waters from such systems, they are probably good enough to be used in a general screening tool such as that being developed in this project. A much greater challenge is to estimate the number, size and location of septic tanks within each catchment, as there are no central records of these. In view of this, various possible methods of deriving the required information from readily available data are reviewed below (Sections 5.3.1 to 5.3.4).

Perhaps a better approach for the present study would be to assume that the P input to the septic tank would be the same as that given for raw domestic waste (i.e.1.2 kg *per capita* y^{-1}, see Section 5.2.2) and multiply this by a transfer coefficient that reflects P removal by the tank itself and the soakaway system. Such a value (60%) has been calculated for septic systems in the River Main catchment in Ireland (Smith 1977). This suggests that the P export from a septic tank, after passing through the immediate soakaway system, would be about 0.7 kg *per capita* y^{-1}. Of this, only a proportion would travel to the nearest watercourse, depending on the soil type and distance travelled across the catchment. Ellis and Childs (1973) showed that significant amounts of phosphate (50µg l^{-1} PO_4-P) could still be detected in sub-surface flow more than 100 m down slope from a septic tank discharge in the USA. Similarly, phosphate concentrations in a borehole adjacent to a septic tank in Ireland were found to be 140µg l^{-1} higher than were recorded in a control borehole further up the catchment (LLCMMS, 2000). The same study also recorded an additional 36µg l^{-1} of phosphate in a borehole 10m from the septic tank by comparison with the control borehole.

Until more information becomes available on the transfer of septic tank effluent across catchments, it is suggested that septic tank discharge is treated, for risk assessment purposes only, in one of two ways. The first is as an additional P transfer value of 0.7 kg P *per capita* y^{-1} within Layer 2 of the PIT tool (see Section 6.2.3). This will allow the transport of P from septic tank discharges to be treated in the same way as P from any other source. The second is to apply a P export value of about 0.3 kg *per capita* y^{-1} to the unsewered population living within a catchment in order to estimate the delivery of septic tank effluent directly into watercourses. This follows the approach applied by Frost (1996) to the Loch Leven catchment (see above).

It should be noted that, although septic tanks may be at some distance from any given lake, it should not be assumed that they have no influence on the water quality within that lake. This is because the properties that they serve are often built on level ground near rivers or streams, or sited close to streams or groundwater from which they can obtain a private water supply. So, the theoretical ‘self-cleansing’ route between the septic tank and the lake is often short-circuited through the transfer of P-laden effluent into a nearby watercourse, which then carries it very rapidly and directly to the lake. It is impossible to determine the extent of this problem without further research.

5.3.1 Method 1

It has been suggested that discharge from septic tanks could be estimated from the number of people served by these systems, their average water consumption, and the P concentration of the effluent (SNIFFER, 2003). However, it seems unlikely that the information required to implement this method would be readily available at the national scale. This method is, therefore, considered to be unsuitable for national risk assessment purposes.

5.3.2 Method 2

An alternative method for estimating P loads from septic tanks is described in May *et al*. (1997b). The method is based on the estimated number of dwellings within a catchment using septic tanks for sewage disposal, an assumed number of occupants per dwelling (3, for the Bassenthwaite catchment) and a *per capita* P export coefficient (see below). The density of septic tanks per catchment was obtained by difference between the number of dwellings within the catchment and the number of those dwellings that paid sewerage connection charges. A postcode approach was used to estimate this value so that individual households could not be identified. This approach was found to be sufficiently accurate to allow the approximate geographical location of each septic tank system to be determined.

Figure 5.2 shows the results of applying this approach to the catchment of Bassenthwaite Lake, Cumbria. The *per capita* P export coefficient used in this study was a general value of 0.7 kg P ann^{-1} (Harper 1992) which was taken to represent the worst case scenario that almost all septic tank effluent reaches the lake very rapidly in this steep, upland catchment with many small drainage channels. For more general application, a lower value of 0.3 kg P ann^{-1} (see Section 5.3, above) is suggested.

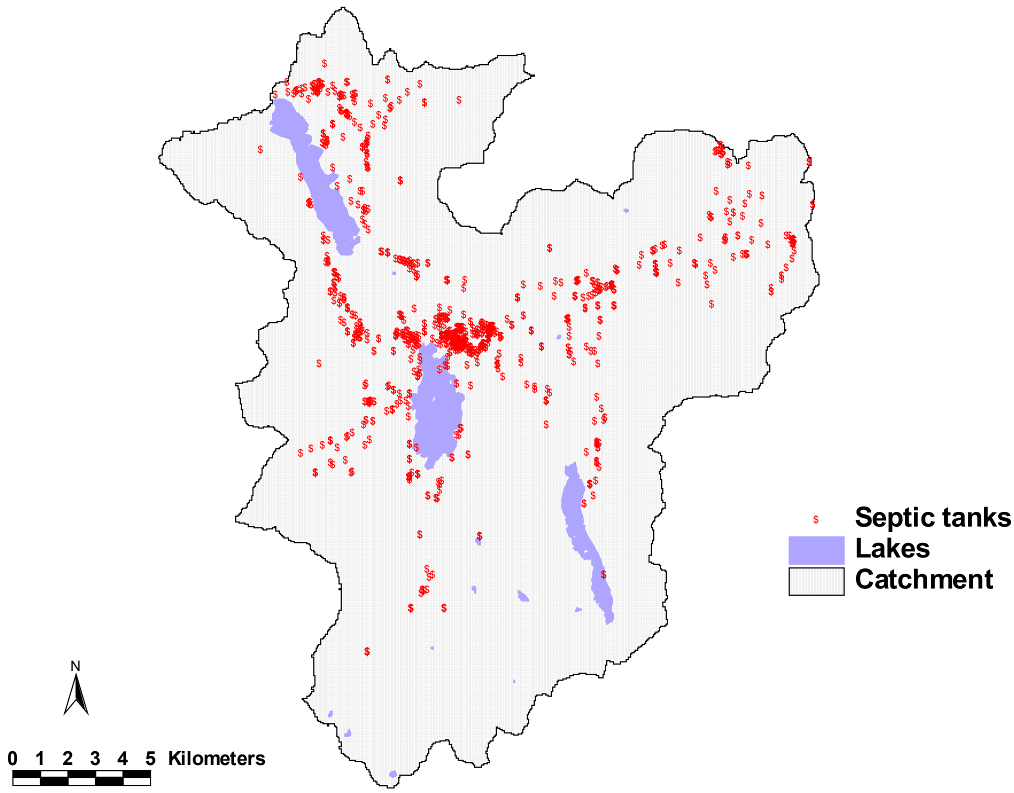


Figure 5.2 Location of septic tanks in the catchment of Bassenthwaite Lake

The method described above is probably the most accurate method of estimating the number and location of septic tanks, but would be expensive and time consuming to implement on a national scale. As such, it is unlikely to be of any practical use in the development of a national screening tool within the context of this project. However, it could provide important, site-specific information when management strategies are being developed for individual lakes and their catchments, especially if more accurate values for P transport from septic tanks can be determined from experimental studies or the application of the PIT tool.

5.3.3 Method 3

May and Gunn (2000) and May *et al*. (2001) employed a more labour intensive, map-based method for estimating the number of rural households within the catchments of Lochs Ussie and Flemington, respectively, in order to estimate the P load to these water bodies from septic tanks. Individual dwellings were identified by eye from a 1:50 000 scale Ordnance Survey Landranger map. These figures were then used to estimate the likely P load from this source using an estimated household size of 2 persons and a *per capita* P export coefficient of 0.7 kg ann^{-1} (Harper 1992). It should be noted that a lower *per capita* P export coefficient value of 0.3 kg ann^{-1} is recommended for use in the current project (see above) and that standard consenting procedure in England

and Wales assumes 1 person per bedroom plus 0.5 persons per house to estimate discharges to septic tanks, so that for example a two bedroomed house uses an average of 2.5 people, not 2.

Although this method was applied manually in the Ussie and Flemington catchments, it would be possible to use the Ordnance Survey (OS) mastermap data in a GIS to make an estimate of the number of isolated buildings in each catchment at the GB level. The resulting dataset of shapes would be significant (10Gb), but there are a series of methods by which the data could be made more usable, allowing the estimates to be made in realistic machine processing times. Although this approach is outside the scope of the current work, we recommend that the Agency consider obtaining these data for future risk assessments.

5.3.4 Method 4

Another method that could be used to estimate the population served by septic tanks within a catchment is to subtract the population equivalent (PE) values of all sewage treatment works within a catchment from the total population resident within the catchment. A potentially significant limitation of this approach is that the PE value of a sewage works includes the PE of any industrial effluents that it receives and may not correctly reflect the size of the sewered population within the catchment. However, a comparison of the PE estimates and 1991 census population data for ten test catchments within this project revealed an unexpected and a much more serious problem. Table 5.4 shows the sum of all STW PE values within each of the 10 catchments and the corresponding population identified in the census. In two of the ten catchments (Derwent Water and Marsworth Reservoir), the two estimates are within 8% and 3%, respectively, suggesting that almost all of the population are sewered. In the catchments of Bala Lake, Barton Broad, Grasmere and Slapton Ley the census estimate exceeds the PE estimate by 32%, 20%, 17% and 83%, respectively, which may indicate the proportion of the population linked to septic tanks. The discrepancy at Barton Broad is, however, likely to be due to the fact that a proportion of the catchment population are served by South Walsham STW, which discharges to sea, not to within the catchment; this may be true for other sites too. In four of the catchments (Bassenthwaite, Coniston, Esthwaite and Windermere), the population equivalents (PEs) were found to be significantly greater than the census population estimates by 26%, 111%, 230% and 60%, respectively. As these catchments all contain major tourist centres, it seems likely that some of these STWs have been designed to cope with much higher loads than those produced by the local population, alone, although we have been unable to verify this to date. Whatever the reason, these large negative discrepancies suggest that the subtraction of the PE value from the census population estimate is not a reliable method of estimating the size of the population served by septic tanks within a catchment.

Many site-specific studies have suggested that it is important to estimate the P load to water bodies from septic tanks, as this may account for up to a quarter of the P load from external sources (Table 5.3). In order to achieve this on a GB scale, **Method 3** is probably the most cost-effective and practical means of estimating the size of the population served by septic tanks within a catchment. This value should then be combined with a *per capita* P export coefficient for septic tanks, such as that recommended above.

5.4 Small package treatment plants.

In addition to septic tanks discharging to soak-away, one other form of sewage treatment in rural areas not served by mains sewer is through small package treatment plants. These treatment plants confer basic treatment to household sewage and produce a final effluent, which is discharged to a watercourse under a consent to discharge provided by the Environment Agency (in England and Wales). The size of plant installed and therefore the volume of effluent discharged is determined by the number of households it serves, which can vary from one to many, although one or two households is more usual. The effluent discharged is generally of good quality in terms of low biological oxygen demand and ammonia concentrations, although TP and total nitrogen concentrations may still be high. Within England and Wales there are many such discharges and this type of treatment plant is often installed for new developments, or where septic tanks are being replaced. Unlike septic tanks, as a consent is required for the discharge, the Agency has a record of the location of each package treatment plant. There can be many hundreds of such discharges in catchments depending on their size and the availability of mains sewage treatment.

The significance of small package treatment plants when determining the per capita phosphorus export from catchments is two-fold. Firstly the effluent, although of good quality in terms of sanitary determinands, receives no intentional phosphorus removal and secondly it is discharged directly to a watercourse – there is no soakaway involved. Therefore, the phosphorus content of the effluent is likely to be relatively high compared with a septic tank (probably similar to that from STWs with secondary treatment) and will be dissolved (i.e. readily available for biological uptake). There appears to be no data available on the likely phosphorus content of the effluent from such plants, although, the type of detergent used and the effectiveness of the plant operation will affect it. Most plants require sludge removal at regular intervals and it is likely that phosphorus will be removed in this process. However, some plant designs are such that sludge production is minimal which implies the effluent will contain more of the nutrients usually found in the sludge. Taking a precautionary approach and assuming no phosphorus removal from the effluent may well lead to an over-estimate of the per capita phosphorus export from this subsection of the population. Alternatively, assuming a similar export to STWs with secondary treatment is a pragmatic and sensible solution in the absence of more specific information and will probably enable the phosphorus contribution from small package treatment plants to be quantified effectively (see Section 5.5 below).

5.5 All sources of sewage-related phosphorus load

Although several methods of estimating the P load to lakes from sewage works and septic tanks within their catchments are discussed above, most are difficult to implement in the Tier 1 assessment process due to the problems associated with obtaining the required data at the national level. The only practical approach to this process at present is to apply *per capita* P export coefficients of 0.4 kg y^{-1} for people connected to the sewage works or package treatment plants and 0.3 kg y^{-1} for those served by septic tanks. As these values are very similar, and it is at present impossible to differentiate between people who are connected to a sewage works and those that rely on septic tanks for sewage treatment, a very simplified method of multiplying the number of people resident in the catchment by a general export coefficient of 0.4 kg $capita^{-1}$ y^{-1} is recommended for a more precautionary Tier 1 risk assessment approach. It is, however, strongly recommended that the more detailed data and approaches outlined above are used for the site-specific analyses required later in the risk assessment process for lakes that have been identified as 'at risk' during the Tier 1 assessment.

Table 5.4 Data for ten test catchments showing the relationship between the population equivalents of all sewage works within the catchment and the resident population as estimated **from the 1991 population census**

Lake	Lake ID	EA Region	STW name	PE	UWWT-PE	Population (1991 Census)
Bala Lake	34987	Wales	Llanuwchllyn	1352	1352	2005
Barton Broad	35655	Anglian	Southrepps	795	795	
			Stalham	10932	10932	
			Worstead	358	358	
			Total	12085	12085	15253
Bassenthwaite Lake	28847	North West	Bassenthwaite	710	710	
			Embleton	530	530	
			Grange in	340	340	
			Keswick	7032	25100	
			Rosthwaite	190	190	
			Seatoller	161	161	
			Stonethwaite	310	310	
			Thornthwaite	700	700	
			Threlkeld	2000	2000	
			Total	11973	30041	8293
Coniston Water	29321	North West	Coniston	2336	2336	1107
Derwent Water	28965	North West	Grange in	340	340	
			Rosthwaite	190	190	
			Seatoller	161	161	
			Stonethwaite	310	310	
			Total	1001	1001	925
Esthwaite Water	29328	North West	Hawkshead	1723	1723	522
Grasmere	29184	North West	Grasmere	676	676	812
Marsworth Reservoir	40608	Thames	Tring (2m -	11800	11800	11441
Slapton Ley	46472	South West	Slapton	100	100	607
Windermere	29233	North West	Ambleside	3368	3368	
			Grasmere	676	676	
			Hawkshead	1723	1723	
			Langdale	1752	1752	
			Windermere	7291	15591	
			Total	14810	23110	14420

5.6 Fish Farms

Fish farms are a significant and increasing source of nutrient loads to freshwater systems through waste material that comprises, mainly, uneaten food and fish excreta. Most fish farms within Scotland are located in the Highlands and Islands (Figure 5.3), although small numbers are also located in other parts of Scotland. In total, there are about 380 consented fish farms in Scotland. Of these, about 130 are cage fish farms, and the remainder are tank fish farms. Equivalent information for England and Wales is difficult to obtain, but a report on rainbow trout production in England and Wales suggests that production may be quite high in some areas (CEFAS, 2000). This report shows that, in 1999, the total biomass of rainbow trout produced in England and Wales was 6,710 tonnes – mostly in the old Environment Agency Regions of Southern Region (28%) and Wessex Region (35%).

Three studies completed by CEH (Bailey-Watts *et al*., 1993; Hall *et al*., 1993; May *et al*., 1997), that have compared the estimated TP load to lakes from cage fish farms with the total TP load to the lake, have shown that it is important to consider the contribution of P from fish farms when calculating a P budget for a lake. The results of these studies, on Lochs Earn and Shiel in Scotland, and on Esthwaite Water, in England, have indicated that cage fish farms could be the source of up to 60 per cent of the external P load to a lake (Table 5.5).

Table 5.5 Site-specific estimates of phosphorus loads to lakes from fish cages.

Waterbody	Estimated P load to waterbody from fish cages (tonnes y^{-1})	Estimated proportion of total P load to waterbody from fish cages (%)	Reference
Esthwaite Water	0.8	51.4	Hall *et al*. (1993) May *et al*. (1997a)
Loch Earn	3.6	59.7	Weller (2000)
Loch Shiel	0.9	12.0	Bailey-Watts *et al*. (1993)

There are two main types of fish farm. The first is a cage fish farm, in which fish cages are usually suspended from flotation collars in the open water of lakes. The second is tank fish farms, which tend to be alongside the lakes or rivers from which they abstract water and to which they discharge effluent. The P load to lakes from tank fish farms within their catchments is relatively easy to estimate since their discharges are usually consented and monitored. Although these data were unavailable to the project at the stage of writing this report, SEPA are currently compiling a central database of this information.

In contrast, the P load to lakes from fish cages suspended in the open water is more difficult to determine. This is because, despite being consented, the effluent from these cages discharges directly into the lake water and their load cannot be measured directly. So, the P load to lakes from these fish cages must be derived from other information that is readily available for each fish farm. At the site-specific level, such information may include the feed input to the cages and/or the biomass of fish removed. However, such detailed information is unlikely to be available at the national level. Examination of information on fish farm discharge consents provided by SEPA showed that the only parameter available from which P inputs to lakes could be estimated was annual biomass production. Even then, these data were only available for 57% of the cage fish farms listed.

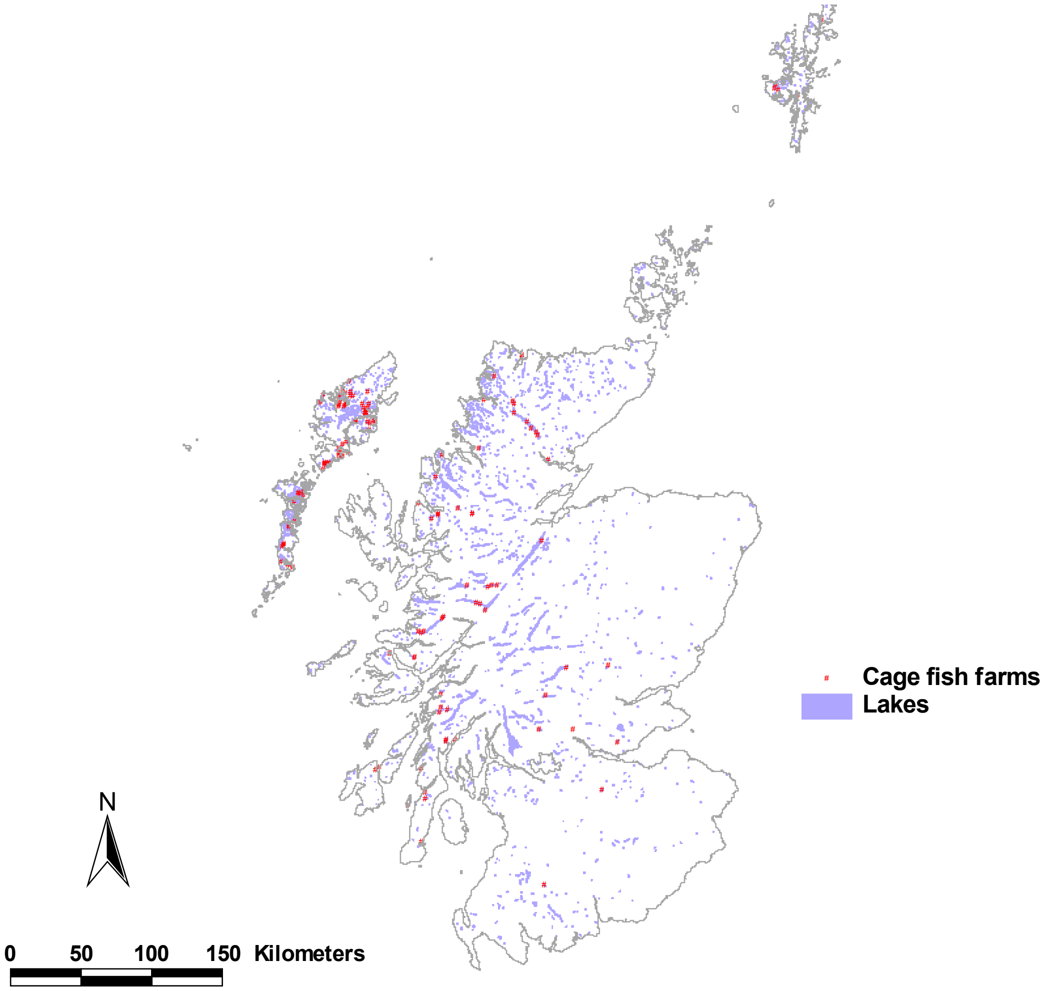
5.6.1 Method 1

The method used to estimate the P loads from fish cages to Lochs Earn and Shiel was based on confidential information on fish production rates and feeding rates provided by the fish farm operators (Bailey-Watts *et al*., 1993; Weller, 2002). Such information is unlikely to be available nationally for use on a GB scale.

5.6.2 Method 2

Another method of estimating the P load from fish cages to the lake water is that based on the annual production of fish per year from the fish farm. This figure can be multiplied by an annual P export rate determined from the literature, although it should be noted that P export per tonne of fish produced differs between salmon and trout production. For salmon, Hennessy *et al*. (1996) estimated a value of 9.1-10.0 kg TP per tonne of fish per year for fish reared in tanks; Kelly *et al*. (1996) produced a similar figure of 11 kg P per tonne of fish per year, also for tank reared fish. These results suggest that 10 kg P per tonne of fish per year would be suitable figure for estimating the P load to lakes from salmon farms within their catchment.

Figure 5.3 Distribution of cage fish farms in Scotland



The equivalent P export value for trout is generally much higher than that for salmon. Phillips (1985) determined a value of 27 kg P per tonne of fish produced per year for rainbow trout in Scotland, while Foy and Rosell (1991) estimated a value of 25.6 kg P per tonne of fish produced per year for tank reared rainbow trout in Northern Ireland. An average value of 26 kg P per tonne of fish produced per year is, therefore, recommended for use in this project. Interestingly, Foy and Rosell (1991) also make the observation that the P content of effluent from a fish farm producing 50 tonnes of fish per year is equivalent to the P content of treated domestic sewage from a population of about 1400 people. *Pro rata*, this suggests that the P load to lakes from larger fish farms with consent to rear almost 400 tonnes of trout per year (e.g. that in Ardveich Bay, Loch Earn) have an annual P discharge equivalent to that of a sewage works serving a population of about 11,000 people.

5.7 Recommended Guidance for Risk Assessment

5.7.1 Sewage works and septic tanks

It is difficult to differentiate between the P load to surface waters from STW effluent and that from septic tanks at the national scale due to limitations in the available data. Until this situation improves, and per capita P export coefficients can be determined more accurately, the following approach is recommended for estimating the P load to lakes from these sources:

Step 1. Estimate the resident population (N_{1991}) in each catchment from the 1991 population census data

Step 2. Multiply N_{1991} by a *per capita* P export value of 0.4 kg y^{-1}

5.7.2 Fish farms

Most information on the way that fish farms are managed, such as the amount and type of feed used and the mortality and production rates of fish, is confidential and only available from individual fish farm operators. The only widely available data on fish farms is the consented annual rate of fish production or biomass, which is held by the regulatory agencies. A methodology based on these data is probably the best approach to assessing the P load from cage fish farms, at present:

Step 1. Obtain information on the location and annual biomass production of each cage fish farm; enter these into a GIS

Step 2. Determine the annual tonnage of fish produced by cage fish farms in each lake (B_{CAGE})

Step 3. Multiply this value by a P export coefficient of 10 kg P per tonne of fish per year for salmon and 26 kg P per tonne of fish per year for trout.

The data required in Steps 1 and 2 are incomplete at present and require further compilation for application on a national basis.

5.8 Recommendations for Future Development

5.8.1 Sewage works

Future development of the methodology for determining the P export to waterbodies from STWs within their catchment should be based on the following approach:

Step 1. For each STW in GB, compile information on the location of the discharge, the actual (rather than design/consented) PE value, and the level of treatment; enter this information into a GIS system.

Step 2. Estimate *per capita* P export coefficients from concentration and flow monitoring data for STWs with different levels of treatment;

ideally, the monitoring data should relate to the discharge from STW effluent pipes, rather than in-stream samples collected up- and down-stream of the effluent pipe. This will avoid the complications associated with in-stream P-processing and any additional contribution of P from diffuse sources between the up- and down-stream monitoring sites.

Step 3. Estimate the P load from each STW within each catchment by multiplying the actual PE value for each works by the *per capita* P export coefficient appropriate to the level of treatment at that works.

Step 4. Sum the values for all STWs across each catchment.

5.8.2 Septic tanks

A future development of the methodology for assessing the impact of septic tank discharges on water quality should be based on the following approach:

In the short term use the Mastermap 1:10,000 OS data to identify potential dwellings with septic tanks and use a figure of 2.5 persons per household to estimate the contributory population. A more accurate method is, however, outlined below and should be pursued in the longer term:

Step 1. Obtain postcode address data for all premises in GB from the Ordnance Survey (Dataset A)

Step 2. Obtain the addresses or postcodes of all premises within GB that pay sewerage charges from the relevant water companies or local authorities (Dataset B)

Step 3. Subtract Dataset B from Dataset A to identify those premises not connected to the main sewerage network; and using digitised catchment maps, estimate the number of such premises in each catchment (P_{SEPTIC})

Step 4. Estimate the average number of residents at each of these premises (N_R), taking into account the proportion of the year that they are likely to be resident there, if many of the properties are likely to be used as holiday homes.

Step 5. Estimate the P load to the lake as the product of P_{SEPTIC}, N_R and a suitable *per capita* P export coefficient for septic tanks

The accuracy of this approach and that outlined in Section 3.5.2 depends upon good estimates of P export to receiving waters from septic tanks. These are not available at present. Many of the values currently in use are based on North American studies, such as that by Ellis and Childs (1973), which were undertaken under different climatic and soil conditions to those found in GB. Research is needed to determine values that are applicable across a range of soil types and geographical areas of GB.

5.8.3 Fish Farms

Further development of the methodology for estimating the P load from cage fish farms to lakes requires the compilation of SEPA data on fish farm biomass production levels to be completed and the availability of similar datasets for England and Wales to be investigated. A method for determining the impact of tank fish farms on water quality also needs to be developed once the necessary discharge monitoring data become available.

5.9 Data Availability, Harmonisation and Quality Assurance

The data required to perform the initial risk assessment for lakes is, or will shortly be, available for most areas of the country. Those data necessary to achieve the future development tasks outlined above are, currently, incomplete at the national scale and require further investigation.

6 DIFFUSE NUTRIENT PRESSURES

Richard Brazier, Louise Heathwaite, Shuming Liu, Linda Pope, Rachael Dils, Mike Hughes, Laurence Carvalho and Linda May

6.1 Introduction

The assessment of nutrient enrichment on both terrestrial and aquatic ecosystems is extremely difficult with respect to the scale of impact or the likely costs involved. Whereas most ecosystems respond in a similar way to an increase in N supply, which causes a reduction in species diversity and an increase in productivity, it is difficult to isolate the effects of N because other nutrients, such as P, are commonly enhanced. As a consequence, many of the effects of nutrient enrichment are chronic in nature and substantial lags are likely between implementation of restrictions and some reduction in nutrient loss.

Research on diffuse pollution has commonly taken one of two routes: (i) small-scale process studies conducted at the lysimeter, field plot or small catchment scale (up to 1km^2), and (ii) deterministic, physically-based process models. Empirical research has made some inroads in understanding the mechanisms of nutrient transport and delivery from diffuse sources to receiving waters. However, the temporal and spatial complexity of diffuse catchment sources means it is difficult to see how mitigation can be developed strategically without recourse to predictive models. The available research on diffuse nitrogen pollution, for example, highlights the need for an improvement in predictive capacity in order to test various proposals for changes to, for example, agricultural land use and fertiliser management (Edwards *et al.* 2000).

6.1.1 Sources of diffuse nutrients

The most significant diffuse sources of nutrients to surface waters are associated with agricultural activities. Contribution of diffuse agricultural sources to the overall phosphorus load to a water body increases with percentage of agricultural land in the catchment. Although high phosphorus losses recorded from agricultural land may come from farmyards, most are attributed to excessive accumulation of phosphorus in soils because of long-term inputs of inorganic fertilisers and manures.

Two major changes have taken place in UK agricultural practices since the second World War: (i) intensification and (ii) an increase in average size of farm holding. Between 1960 and 1990 in the UK, the average farm holding size doubled, the area of arable crops and temporary grass increased by 36% (cereal cultivation 60% increase), cattle numbers increased by 70% and poultry by 104% (Defra, 2002). The current P balance for UK agriculture is shown in Figure 6.1. The imbalance in inputs-outputs to the agricultural system has resulted in the accumulation of P in agricultural soils (Heathwaite and Sharpley 1999). In many developed countries, elevated diffuse nutrient loads result from the shift towards specialised and intensive farming systems that import a lot more nutrients in feed and fertilizer than are output in produce. The net result is an increase in the potential for nutrient export in agricultural runoff and the accelerated eutrophication of receiving waters.

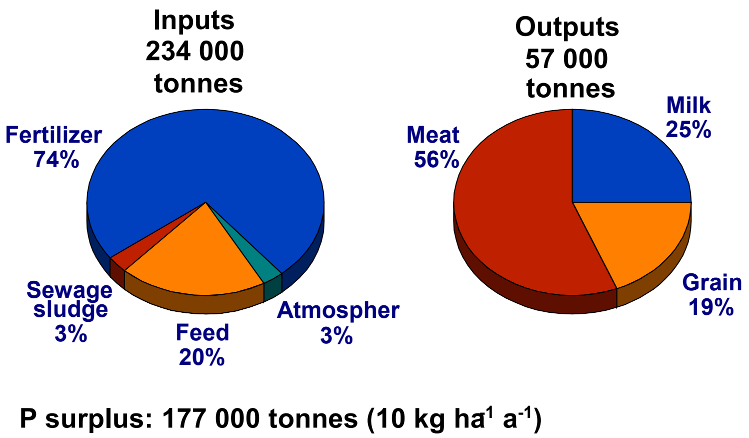


Figure 6.1 External phosphorus fluxes in UK agriculture

The imbalance has been largely blamed on intensive livestock production. Table 6.1 below shows some recent trends in livestock numbers and P output for the UK.

Table 6.1 Phosphorus export from livestock manure

	Livestock number per holding		P output kg/holding	
	1965	1993	1965	1993
Dairy	20	70	390	1130
Pigs	70	290	200	1370
Poultry	9300	29500	2700	8900

The importance of managing agricultural nutrients in a manner that both sustains agricultural profitability and protects the quality of our environment is clear. But it is virtually impossible to produce food and other agricultural products economically without some nutrient losses to ground and surface waters and to the atmosphere (SSSA, 2000).

6.2 Approaches to diffuse pollution modelling

Diffuse pollution modelling requires accurate and sensitive treatment of spatial data. The spatial variations in catchment characteristics may be modelled using lumped, distributed or topological representations. The extent to which models based on any of these representations may be validated depends on the quality of the available data, with distributed models requiring detailed field data to accurately capture the true variation in the catchment, while lumped approaches assume the point scale collection of catchment data are representative of internal catchment processes which are not calibrated. Topological representations of catchment hydrology are becoming more popular through the construction of GIS-based models of catchment structure and function for use as decision support systems for catchment management (e.g. Viney *et al.* 2000; Cassell *et al.* 2001; Quinn 2002). More recently, the topological approach has been applied to diffuse nutrient pollution (Heathwaite *et al.* 2003).

Effective modelling of diffuse pollution must marry terrestrial and aquatic processes. Whereas patch dynamics are critical in terrestrial habitats (Wu and Loucks, 1995), temporal dynamics (e.g. rainfall, drought) dominate the aquatic environment. Understanding diffuse pollutant export from catchments needs to accommodate <u>both</u> but many simulation models do not adequately represent the spatial controls on nutrient export (Harris, 1998). Thus in current diffuse nutrient modelling research, there appears to be a move from sophisticated, data-hungry models towards simple semi-distributed models that estimate nutrient loss on the basis of the limited data available. Such models commonly operate with a spatial resolution around 1 km and are driven by data availability, which in most developed countries includes land cover, livestock numbers, crops grown, climate, and physical properties of soils.

6.2.1 Basic Export Coefficient Approach

Empirical nutrient export models may be used to develop indicators of diffuse nutrient pressures at the small catchment through to regional or national scale. These so-called ‘black-box’ models have long been used to provide simple budgets of nutrient loads entering water bodies. Such models make no attempt to explain the processes involved in generating nutrient outputs from a set of input parameters but allow some evaluation of the impact of changing inputs or managing outputs within, for example, agricultural systems.

Existing empirical modelling approaches used in the UK include the simple but well-tested Export Coefficient Model (Johnes, 1996; Johnes and Heathwaite, 1997), which has an implicit water quality bias and the P-Expert System (Fraser and Harrod, 1998), which has a soils bias but includes hydrological drivers. In the US, the modified P Index (Gburek *et al.* 2000) has been refined for use in a number of states where legislation relating to livestock production requires nutrient management planning for P (Weld and Beegle 2001). Heathwaite *et al*. (2000) describe an integrated N and P Index based on US research.

The export coefficient model calculates the mean annual total N and total P loading (kg a^{-1}) delivered to a surface water body (lake or river) as the sum of the nutrient loads exported from each nutrient source in the catchment. Initially based on models developed in North American studies of eutrophication during the 1980s, the export coefficient modelling approach has been developed, refined and tested on a number of UK catchments (Johnes *et al.* 1996; 1998; Johnes and Heathwaite 1997; Johnes 1999).

The model calculates the total load of N or P arriving in a water body as the sum of the nutrient loads exported from each nutrient source in the catchment, as a function of the rate of nutrient input to that source, and the nutrient export potential of each land use type, livestock variety or people. It also takes into account atmospheric deposition inputs to any system. Thus:

$$L = \sum_{i=1}^{n} E_i (A_i (I_i)) + p$$

Where

L = Loss of nutrients
E = Export coefficient for nutrient source i
A = Area of catchment occupied by land use type i, or Number of livestock type i, or of people
I = Input of nutrients to source i
p = Input of nutrients from precipitation

The export coefficient (E_i) expresses the rate at which N or P is exported from each land use type in the catchment. For animals, the export coefficient expresses the proportion of the wastes voided by the animal which will subsequently be exported from stock houses and grazing land in the catchment to the drainage network, taking into account the time spent by each livestock type in stock housing, the proportion of the wastes voided subsequently collected and applied to land in the catchment, and the loss of nitrogen through ammonia volatilisation during storage of manures. For human wastes, the export coefficient reflects the use of phosphate rich detergents and dietary factors in the local population, and is adjusted to take account of any treatment of the wastes prior to discharge to a water body using the following equation:

$$E_h = D_{ca} * H * 365 * M * B * R_s * C$$

Where

E_h = Annual export or N or P from human population (kg a^{-1})
D_{ca} = Daily output of nutrients per person (kg d^{-1})
H = Number of people in the catchment
365 = Days per year
M = Coefficient for mechanical removal of nutrients during treatment (range 0.85 - 0.9, reflecting removal of 10 - 15 % of the nutrient load)
B = Coefficient for biological removal of nutrients during treatment (range 0.8 - 0.9, reflecting removal of 10 - 20 % of the nutrient load)
R_s = Retention coefficient of the filter bed (range 0.1 - 0.8, reflecting retention of 20 - 90 % of the nutrient load)
C = Coefficient for removal of P if phosphorus stripping takes place (range 0.1 - 0.2, reflecting removal of 80 - 90 % of the P load)

At its finest scale of resolution, the model has been applied to individual catchments, from 5 to 500 km^2 in area, using the field scale as the spatial unit for data input, providing output on an annual basis. For any one land use type, the total export of N and of P is calculated as the area of each field, multiplied by the input of nutrients to that field from fertiliser applications and nitrogen fixation, and then multiplied by the export coefficient selected for that nutrient source type. Calculations for each livestock type are conducted on a farm by farm basis, assuming that the livestock are equally distributed over the grazing land on the farm, taking into account whether manure collected in stock housing is stored prior to application to the land. Calculations for the human population are conducted for each separate sewage treatment or septic tank system within the catchment, taking account of the degree of sewage treatment prior to discharge.

More recently, attempts have been made to derive a generic export coefficient model that may be applied to a range of catchments with similar environmental characteristics (Johnes *et al.* 1998). Generic sets of export coefficients were derived for each characteristic land use region type which could then be applied to parish scale Agricultural Census data for any part or whole parish lying within each land use region type.

This approach has not been used in NUPHAR; instead a similar approach applicable to all GB based on CEH land cover and animal stocking data held in the GB lakes inventory was applied.

6.2.2 The GB Lakes Inventory approach (GBI) – NUPHAR Tier 1

A system was developed to assess nutrient pressure in all lakes in Great Britain with a surface area greater than 1 ha. Loadings of TP were chosen as the relevant nutrient parameter to define the level of pressure that each lake is exposed to. TP loads were estimated from GIS-derived catchment land use (CEH land cover classes), animal stocking densities and catchment population data.

TP loads from agricultural and human sources were estimated using a simplified set of P export coefficients from the literature (see Hilton *et al,* 1999) based on the coefficients developed by Johnes (1996). The values used for the NUPHAR project and the previous risk assessment project (Bennion *et al*., 2002) are shown in Table 6.2 for a range of land cover types, animals and people.

Table 6.2 highlights a few revisions of some of the export coefficients that were made during the course of this project. These include:

- Unclassified land: a variable catchment-specific export coefficient based on weighted average of known land-cover types in catchment
- Coniferous woodland: increase from 0.02 kg ha^{-1} a^{-1} to 0.15 kg ha^{-1} a^{-1} based on figures from May *et al*. (1996)
- Felled forest: increase from 0.02 kg ha^{-1} a^{-1} to 0.20 kg ha^{-1} a^{-1} based on figures from May *et al*. (1996)
- People: increase from 0.38 to 0.40 kg capita^{-1} a^{-1} [see chapter 5 for discussion].

In general the approach in England and Wales is comparable with the approach in Scotland, although the animal density data in these two regions differs in resolution; Scotland data is based on data at 2 km^2 resolution (Edinburgh data library), whilst data from England and Wales is at a 1 km^2 resolution (Lord and Anthony, 2000). Additionally agricultural TP loads from Scotland include data from farmed and wild deer.

Total P load, expressed as kg a^{-1}, was then calculated for each lake catchment by summing the total contribution from land use, animals and humans. This was carried out for both the original and revised set of coefficients.

The discharge of water into each lake was calculated from the runoff depth multiplied by the catchment area. The TP loads can then be converted into in-lake annual mean TP concentrations (µg l^{-1}) and, if required, into annual mean chlorophyll$_a$ concentrations (µg l^{-1}) using the relevant OECD regression equations which take account of retention time (OECD, 1982).

A risk assessment is then possible by comparing modelled TP concentrations with either the site-specific or type-specific reference conditions to derive a predicted EQR (see Chapter 2).

Table 6.2 Tier 1 export coefficients (kg ha^{-1} a^{-1}) applied to CEH land cover classes, animal types and humans in current study (revised coefficients) and earlier study (Bennion *et al.*, 2002)

LCID	LCNAME	Original coefficients (Bennion et al., 2003)	Revised coefficients
0	Unclassified	0.48	variable
1	Sea / Estuary	0	0
2	Inland Water	0	0
3	Beach / Coastal	0	0
4	Saltmarsh	0	0
5	Grass Heath	0.02	0.02
6	Mown / Grazed Turf	0.2	0.2
7	Meadow / Verge / Semi-natural	0.2	0.2
8	Rough / Marsh Grass	0.02	0.02
9	Moorland Grass	0.02	0.02
10	Open Shrub Moor	0.02	0.02
11	Dense Shrub Moor	0.02	0.02
12	Bracken	0.02	0.02
13	Dense Shrub Heath	0.02	0.02
14	Scrub / Orchard	0.02	0.02
15	Deciduous Woodland	0.02	0.02
16	Coniferous Woodland	0.02	0.15
17	Upland Bog	0	0
18	Tilled Land	0.66	0.66
19	Ruderal Weed	0.02	0.02
20	Suburban / Rural Development	0.83	0.83
21	Continuous Urban	0.83	0.83
22	Inland Bare Ground	0.7	0.7
23	Felled Forest	0.02	0.2
24	Lowland Bog	0	0
25	Open Shrub Heath	0.02	0.02
26	cattle	0.22	0.22
27	pigs	0.14	0.14
28	sheep	0.045	0.045
29	fowl	0.0054	0.0054
30	humans	0.38	0.64

It must be noted that the approach is limited by the quality of the input and output data that are used, as the method relies entirely on the data being representative of the 'real' inputs and outputs from the system. Most of the coefficients used are based on literature reviews and, at this stage, no consideration is made of the error or uncertainty surrounding the input or output data and no attempt has been made to calibrate the coefficients for GB.

6.2.3 The Phosphorus Land Use and Slope (PLUS) model

The PLUS model developed by MLURI for application to Scottish catchments is based on two key assumptions:

- Diffuse P loss from a catchment can be estimated using representative loss coefficients
- P loss coefficients are explained by the land use and slope of an area.

Thus the key parameters in the PLUS model are slope and land use. A transfer function based on P loss coefficients is used to link the land and water phases. The model output is given as P load in terms of a range between upper/lower 'confidence limits'.

The inputs to the PLUS model are: catchment boundary coverage, land use coverage (MLURI 1988 Land Cover of Scotland), slope coverage (DEM 50m resolution), and P loss coefficients (MLURI/FRPB values). The platform used to run the PLUS model is ARCINFO. Table 6.3 below gives the P loss coefficients used in the PLUS model.

Table 6.3 PLUS phosphorus loss coefficients (kg/yr)

Slope class P loss range	FLAT PLOAD MIN	FLAT PLOAD MAX	MEDIUM PLOAD MIN	MEDIUM PLOAD MAX	STEEP PLOAD MIN	STEEP PLOAD MAX	TOTAL PLOAD MIN	TOTAL PLOAD MAX
water	80.01	228.61	0.00	0.00	0.00	0.00	80.01	228.61
wetland	0.41	3.04	0.00	0.00	0.00	0.00	0.41	3.04
blanket bog and peatland	9.26	27.79	9.58	19.16	0.50	0.83	19.35	47.79
improved grassland	1.13	2.49	3.51	6.81	0.00	0.00	4.65	9.30
coarse grassland	15.15	30.29	43.33	86.67	25.75	48.29	84.23	165.25
smooth grassland	0.84	1.69	5.41	10.83	0.03	0.07	6.29	12.58
heather all types	16.23	32.47	127.74	255.48	115.50	216.56	259.	504.51
bracken	0.01	0.02	0.09	0.19	0.59	1.11	0.69	1.31
cliffs	0.00	0.00	0.00	0.00	0.50	4.03	0.50	4.03
montane vegetation	0.18	0.73	1.97	5.91	4.75	9.51	6.91	16.14
mixed woodland	0.02	0.04	0.35	0.67	0.00	0.00	0.37	0.71
coniferous plantation	76.61	178.76	162.65	309.03	38.02	67.90	277.28	555.68
recently ploughed land	24.53	44.15	113.29	194.22	11.81	19.68	149.63	258.05
open canopy young plantation	38.65	82.12	65.66	125.85	2.15	3.80	106.46	211.77
duneland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SUB-TOTALS	263.03	632.18	533.60	1014.81	199.62	371.77	996.25	2018.77
TOTAL-PLOAD	263.03	632.18	533.60	1014.81	199.62	371.77	996.25	2018.77

The PLUS model has not been used in the NUPHAR project.

6.2.4 Pressure-Delivery Risk Screening matrix (PDRS) – NUPHAR Tier 2

Given the tight timescales for the WFD Risk Assessment, an interim intermediate approach for assessing the pressure on surface waters from agricultural sources of phosphorus has been developed that is more sophisticated than export co-efficient modelling, but less sophisticated than the PIT model. The approach has been termed the 'pressure-delivery risk screening matrix'. The structure of the approach was agreed following a meeting in Sheffield on 30th June 2003 and subsequent development was led by the Environment-Policy Risk and Forecasting team in the Environment Agency. The approach could not be applied to Scotland in the timescales, largely due to the lack of readily available datasets.

The draft pressure-delivery risk screening matrix method has been developed based on the document 'Guidance for the analysis of Pressures and Impacts in Accordance with the Water Framework Directive IMPRESS – Guidance Dec 2002'. Table 6.4 outlines the framework for defining pressures and impacts for the Water Framework Directive.

Table 6.4 The framework used in the pressures and impacts analysis

Term	Definition
Driver	an anthropogenic activity that may have an environmental effect (e.g. agriculture, industry)
Pressure	the direct effect of the driver (for example, an effect that causes a change in flow or a change in the water chemistry.
State	the condition of the water body resulting from both natural and anthropogenic factors (i.e. physical, chemical and biological characteristics)
Impact	the environmental effect of the pressure (e.g. fish killed, ecosystem modified)
Response	the measures taken to improve the state of the water body (e.g. restricting abstraction, limiting point source discharges, developing best practice guidance for agriculture)

The method described in this section is intended for use in the first Pressures and Impact Analysis for river basins which the EU requires member states to complete by the end of 2004. In practice this means that the agencies must complete this review in early 2004 to allow sufficient time for consultation and submission.

Methods developed for this analysis must therefore use available data and/or models and be applicable to all river basins. It should be noted that the method described here is a draft method and the details of the method, such as the boundaries between high medium and low pressure categories may change during the validation of the method.

The data sources used in the approach for Tier 2 are shown in Table 6.5 below:

Table 6.5 Datasets used for the Tier 2 model

DATASET NAME	USED FOR	SUPPLIER
Agricultural census returns 2000	Calculation of number of animals/km^2 and area of crops/km^2	ADAS/DEFRA
P application rates to crops	Calculation of P applied to arable and grassland/km^2	NUPHAR project
P production by livestock	Calculation of P produced by livestock /km^2	NUPHAR project
Base Flow Index	Classification of waterbody into low-medium-high groundwater influence	CEH Wallingford
Sediment Delivery map	Potential for P delivery to watercourses from sediments	Environment Agency Project (EA, 2003)

The PDRS approach to evaluate phosphorus pressures is based on a conceptual model for surface waters that links: (i) the pressure from P applied to land as inorganic fertilisers or organic manures, and (ii) the likelihood of P reaching watercourses attached to sediment particles. These are combined using a pressure matrix to give a final pressure map, which assigns a High, Medium or Low diffuse P pressure to a grid square. The pressure maps are currently based on a 1 km^2 grid, but these will be averaged over individual WFD catchments. The surface waters conceptual model is linked to a groundwater model to provide a means of accounting for P delivery to water in groundwater-driven river basins in England and Wales. Concentrations of P in groundwaters are not routinely monitored across England and Wales, and they have been considered of secondary importance when compared with nitrates and Dangerous Substances. A monitoring programme will be put in place from 2004, but there is at present very little information on which to base a risk map, as there are no standards for P in groundwater to define catchments that have levels considered high. Consequently, the pressure map for P and groundwater has been developed to show areas where there is a high application rate of P combined with areas where there is a high connectivity between the surface and groundwaters. The Base Flow Index (BFI) has been chosen to represent areas where there is a high degree of connectivity between the surface and groundwater, and where surface water bodies are highly influenced by the quality of the groundwater. The BFI is based on soil data (HOST soil classification) and estimates the proportion of the effective rainfall which will flow overland directly to a watercourse, and the proportion which will reach the watercourse via the groundwater. A river which is above an ecologically acceptable P level and which has a high BFI would be likely to have a high pressure from P in groundwater. This cannot, however, be tested until there is more information on the location of groundwater bodies which have a high degree of connectivity with surface waters, and where measured levels of P in the groundwater are high

Background and Method Description

Agricultural census returns are collected annually by DEFRA from individual farmers who report the numbers of livestock they keep and the area under each crop on their farm. The data are confidential but may be used in a non-disclosive format, so that individual farmers and their livestock and arable holdings cannot be identified.

This information is a valuable source of data for the application of nutrients to farmland but it should be remembered that:

- Farmers report total area under crops, not locations where they are grown, so figures averaged across a farm will have some elements of over and underestimation for crops
- Livestock may also be grazed anywhere on the farm, and may be moved during the year and spend time in housing. Their manure may be used on the farm or it may be exported

A derived gridded 1 km^2 was used to calculate areas of crops and numbers of livestock in each 1km^2 grid. Standard numbers for the amount of P applied to crops or produced by animals were taken from layer 1 of the PIT model (Heathwaite *et al.* 2003).

The categories from the agricultural census returns were grouped and a standard application or production rate applied to each category.

From the gridded agricultural census data and the standard application rates, an estimated application rate per km^2 could be calculated, and the production rate from livestock could be calculated separately using the same method. For the overall assessment the P from crop application and from livestock has been summed.

This P application rate was then ranked into No-Low-Medium-High categories based on a log-normal distribution, which was the best-fit distribution to the application data (Table 6.6). Maps of P application pressure divided into No-Low-Medium-High bands based on splitting the distribution at the 10%ile (upper and lower), 20%ile and 30%ile have been proposed, however, the final break points will be agreed following comparison with impact data.

Table 6.6 Ranked P pressure from livestock and crops

P from agriculture kgP/ha/year (log10 30%iles)	Risk	Risk Class
None	No (urban)	0
0-2.92	Low	1
2.92-3.52	Medium	2
3.52-4.1	High	3

Surface pathways: In many catchments, the predominant route of phosphorus to surface water is via attachment to eroded soil particles. The likelihood of sediment-bound phosphorus reaching watercourses from any area is related to the soil type, slope of the land, rainfall and the distance of the land from a watercourse. These factors have been combined to produce a map giving the estimated sediment load that could erode based on a 1 km grid square (Environment Agency, 2003). This map was classified into low-medium-high risk of erosion as recommended in the report. The sediment delivery pressure map was prepared as an ArcView Grid File. This was reclassified so that Low risk was numbered 1, medium risk was numbered 2 and high risk was numbered 3.

The two grids (P application pressure and sediment delivery) can then be multiplied together to give a final ranked pressure map, as in Table 6.7. In this combined matrix, 0 is no risk (urban areas), low is combined pressure 1-2, medium is combined pressure 3-4 and high is combined pressure 5-9.

Table 6.7 Combined total P Pressure Matrix

		sediment delivery		
		1	2	3
P Pressure	0	0	0	0
	1	1	2	3
	2	2	4	6
	3	3	6	9

Note that the level of pressure in catchments can be altered by adjusting the boundaries between the pressure classes, for example if category 4 is reclassed High Pressure, then the number of catchments where there is high pressure is increased. The position of these boundaries is being reviewed using monitoring data from catchments without significant point sources, but the classification produced by this method is a first level assessment, and the Water Framework Directive implementation is an iterative process, which allows assessments to be refined and reviewed as River Basin Management Plans are developed.

Groundwater pathways: The procedure used to generate P loading for each 1 km grid square for P pressure used for surface waters was also used for groundwater (Table 6.8). A 1km^2 grid of BFI values was then reclassified into High groundwater to surface water connectivity (3), medium or mixed connectivity (2), and low connectivity (1). The classification is shown in Table 6.9.

Table 6.8 Classification of BFI

BFI	Surfacewater to Groundwater Connectivity
0-0.65	low
0.65-0.8	moderate
0.8-1	high

The reclassified P pressure grid and the BFI grid were then combined using the matrix below (Table 6.9) to give a pressure map for P and groundwater.

Table 6.9 Matrix of BFI and P Input

Pressure matrix for BFI * P input				
	NONE	LOW	MOD	HIGH
NONE	0	0	0	0
LOW	0	1	2	3
MOD	0	2	4	6
HIGH	0	3	6	9

The DPSR method for groundwater catchments requires further work when the data are available to decide the proportions of low-medium-high pressure within a groundwater catchment that should be used to give the appropriate pressure band. The approach would be improved significantly by inclusion of information on the connectivity between the land surface and the groundwater body, especially as borehole readings in some groundwater bodies amalgamate information on surface waters that originated some distance away from the borehole. Furthermore, the uncertainty of the approach increases for small catchments; this also applies to the surface water pathway.

6.2.5 The Phosphorus Indicators Tool (PIT) – NUPHAR Tier 3a (SILVER STANDARD) and Tier 3b (GOLD STANDARD)

The PIT model uses a simple lumped/semi-distributed approach on an annual time-step. It is designed to have low data requirements and is applicable at national, regional and catchment scales.

The model has a three layer structure with each layer containing a set of parameters that act as the most sensitive indicators for that model layer. Layer 1 calculates the stores of and fresh addition to P in the landscape as a result of human activities and calculates their potential for loss. Layer 2 describes the processes initiating the mobilization of P from the landscape stores. It accounts for both the background losses of native P that would occur irrespective of changes to the inputs to Layer 1 (because the natural environment will always drive some P loss), and any enhanced mobilization of P as a consequence of land management. Layer 3 defines the connectivity of the landscape and the pathways by which the leakage from layer 2 plus any stored potential mobilized from layer 1 into layer 2 is routed to the watercourse. Included in layer 3 are infrastructure features such as farm tracks and roads that may facilitate the rapid delivery of transferred P to water. A full description of the PIT model is given in Heathwaite *et al.* (2003). A summary of each layer is provided here:

Layer 1 comprises the indicators of Potential P Loss and represents the store of P input to each land unit as a function of the P content of the soil plus added materials in kg P. For added materials, the P loss potential is obtained by multiplying each land use unit area by P input from fertilizer P applications added to manure P amendments to each land use unit. Topsoil total P concentration is used to estimate P moved due to sediment loss.

Layer 2 comprises the P transfer Indicators used to calculate the risk that soil P or freshly applied P is moved via particulate P transport, P solubilisation or direct detachment of freshly-applied P. The main drivers for the P transfer Indicators are soil physical properties, climate and slope. The Hydrology and Soil Types (HOST) classification is used in Layer 2 to apportion P transfer to surface and subsurface flow pathways by calculating the volume of water moving along each pathway from Hydrologically Effective Rainfall (HER) multiplied by the proportion of flow along that pathway, defined using HOST. Flow along each HOST pathway is then allocated according to land use.

Layer 3 describes the P delivery Indicators and links the P load moved within fields to P reaching adjacent water course. This layer describes the efficiency of P delivery from diffuse sources, where the key controls included in PIT are: the degree of sediment retention within fields and ditches, the extent of artificial drainage, and the distribution of routes of high connectivity which may increase the efficiency of the transport of sediment and associated P to watercourses.

Currently, the PIT model runs on an annual time step and at a 1km^2 grid scale. The input to and output from each layer are shown in Table 6.10.

The modelling framework for the PIT has been implemented within the ESRI ArcGIS framework to enable rapid algorithm calculation by incorporating gridded averages. For the trial runs of the model in PIT phase 1 reported in Heathwaite *et al.* (2003), the code was written using the Arc Macro Language (AML). The outcome from the model was displayed within ArcView. It was developed using raster- or grid-based data because calculations performed on gridded data are quicker than the same calculations performed on vector-based data. However, because AML is not embedded in the ArcView working environment, the interaction between the model and the user is restricted and the model runs and display of results must be manipulated separately. This approach decreases the transparency and integrity of the model and is not being continued in the current phase of PIT model development. Furthermore, although AML supports user interface design, its capability is simple.

To overcome the limitations of the AML/ArcView platform used in phase 1 we have re-written the model code using VBA and the ArcInfo 8.2 platform. The functionality of the re-written code in VBA has been validated against the original model runs and been shown to be satisfactory. The advantage of ArcInfo 8.x is it not only covers all the functionality of ArcView 3.2 but also supports GIS-based application development using the embedded VBA programming language. By using ArcInfo-VBA, we have significantly improved the interface of the PIT model to make it more user-friendly and to ensure that the interaction with the user is enhanced. For example, we are currently devising a system whereby the value of the coefficients used in the PIT model may be input by the user from a look-up table. In this way we are able to allow catchment-specific model runs to be undertaken. Furthermore, using ArcInfo-

VBA, means the internal model calculations and the display of the model results may be managed within one platform.

Table 6.10 PIT model layers, input data and output calculation (after Heathwaite *et al.* 2003)

MODEL LAYER	SPATIAL INPUT DATA	MODEL OUTCOME
LAYER 1 **Phosphorus Loss-Potential**	Livestock Numbers Crop areas Soil texture, crop area Crop area	Total excretal P, kg Total fertiliser P, kg Average soil Total P, mg/kg Average soil Olsen P, mg/kg
LAYER 2 **Phosphorus Transfer**	Soil HOST class, HER Soil texture, average slope average Olsen P (grass/arable), average flow per pathway Total fertiliser P (grass/arable), average flow per pathway Total excretal P (grass/arable), average flow per pathway Erosion risk class, av. soil total P (grass/arable), av. flow per pathway	Average flow per pathway, mm Erosion risk class Soil P solubilised per pathway, kg Fertiliser P solubilised per pathway, kg Excretal P solubilised per pathway, kg Sediment P mobilised per pathway, kg
LAYER 3 **Phosphorus Delivery**	Surface P mobilised Surface P leaving field, HOST class, slope Drain P mobilised Seepage P mobilised P excretion (by type), HER, HOST class, slope	Surface P leaving field, kg Surface P delivered to stream, kg Drain P delivered to stream, kg Seepage P delivered to stream, kg Voided P delivered to stream, kg

The inputs to the PIT model are made using the ArcGIS grid format. Most of the input data, with the exception of the coefficients, are available in Excel spreadsheet format. Therefore, before being input to the model, the input data are converted from Excel spreadsheet to ArcGIS grid using ArcTool. Currently, the input data for the PIT model utilise the MAGPIE database (Lord and Anthony, 2000) but the model is not confined to this database. The input data comprise land use, crop areas, livestock numbers, climate and dominant soil association. The crop and land use data in MAGPIE are a synthesis of satellite-derived land cover data and the 1995 DEFRA parish agricultural census data. Hydrologically effective rainfall (HER) is based on climate data averaged over the period 1961-1990, and is adjusted for land use and soil type. Soil data with associated textural and HOST classes are derived from the NSRI NATMAP system.

Currently work on the Phosphorus Indicators Tool is taking place with the following objectives:

- DEFRA project PE0112 is calibrating the coefficients in the model and examining a range of water quality datasets not considered in the previous model development. The objective is a 'calibrated' and 'validated' model.

- DEFRA PE0113 is focussing on Layer 3 in the model – P delivery. The outputs from this project, which started in October 2003 will produce refined coefficients for layer 3 and a better understanding of the way in which P is delivered to water from agricultural catchments.

6.3 Test catchments

6.3.1 50 test lakes and catchments areas

An initial selection of 50 test lakes was made to represent the broad range of geology, land use, hydrological connectivity and climate that characterise catchments across the UK (Table 6.11). It was recognised that not all four tiers of assessment would be applied to all catchments as the different modelling methodologies are suited to different catchment scales and are driven by different (input) data requirements. Tier 1 (GB Lakes Inventory) is applied to all test lakes outlined in Table 6.11, Tier 2 (Pressure delivery risk screening matrix) is applied to 8 test catchments and Tier 3a (PIT uncalibrated) is applied to 21 lakes with catchment areas of over $3km^2$, including Loch Leven, but excluding the other Scottish catchments as input data have not been made readily available within the timeframe of the project) and Tier 3b (PIT calibrated) is applied to five catchments as outlined in section 6.3.2.

Table 6.11 50 Lakes chosen for Tier 1 – 3 evaluations

Lake catchment	Catchment area (km^2)	Lake catchment	Catchment area (km^2)
Albury Mill	16	Kinord	9
Barton	109	Leven	160
Bassenthwaite	358	Llangorse	23
Blelham	4	Llech Owen	<1
Boardhouse	39	Lomond	763
Bolder	3	Lower Talley	<1
Bugeilyn	1	Loweswater	8
Bwch_llyn	3	Maes llyn	<1
Cinder Hill	9	Malham	4
Coniston	64	Maree	440
Coron	22	Marsworth	14
Davan	35	Mere, The	3
Derwent	90	Ness	1782
Dinam	3	Oak	5
Earn	140	Rostherne	10
Eela Water	3	Slapton	46
Eiddwen	<1	Tatton	4
Esthwaite	18	Tegid	147
Fach	1	Ullswater	150
Girlsta	4	Upper Talley	2
Glanmerin	<1	Upton Broad	<1
Grasmere	29	Wast Water	45
Gynon	2	White Mere	2
Hickling Broad	21	Windermere	253
Hir	<1	Wyth Eidion	2

6.3.2 Test lakes for the Tier 3b standard

Five test lakes were selected for PIT model runs to the Tier 3b standard based on the requirement that the data necessary to run the PIT model were more or less readily available for these sites. For detailed methodology see section 6.5. The five selected lakes were:

Barton Broad
Slapton Ley
Windermere
Esthwaite Water
Blelham Tarn

These lakes are located within a range of different catchment types and were therefore thought to provide an excellent opportunity to test the flexibility of the PIT approach and its' ability to model both upland and lowland catchments. This exercise also demonstrates the potential of such a modeling approach to capture the range of P loss behaviour in the UK, which will be possible for all UK sites in the near future.

6.4 Modelling methodologies

Details of the Tier 1 and 2 methodologies have been provided in sections 6.2.2 and 6.2.4. The following 2 sections describe the 2 different ways in which the PIT approach is used (both calibrated and uncalibrated) to provide results for a range of test lakes

6.4.1 PIT Tier 3a methodology

Data poor catchments (i.e. those catchments with limited or no observed output data) pose a significant problem for all the above modelling approaches as the results cannot be truly evaluated and subsequently, the models cannot be calibrated to improve the quality of predictions. Accepting this fact (as many catchments are not monitored in sufficient detail to improve model predictions) models such as PIT can still be applied in a meaningful manner as long as results are interpreted without reliance upon observed datasets. This approach is justifiable as, where observed data exist, model predictions have been shown to be reasonable without *a priori* calibration of the coefficient sets.

The Tier 3a methodology is an application of the PIT approach to 21 of the test lake catchments that cover catchment areas of >3 km^2. Coefficient values are estimated (as described in Heathwaite *et al.* 2003) and model results are driven by the nationally available spatial datasets for each catchment. No calibration is performed and it is demonstrated that this approach can be applied to all catchments within the UK given the necessary computational time and input datasets.

This approach equates with the ‘silver’ standard described in the original project brief for the NUPHAR project.

6.4.2 PIT Tier 3b methodology

In the Tier 3b methodology it is recognised that model predictions may be improved by calibration of the coefficient sets. Though this is not always the case, where data are available to perform calibration it is a useful exercise as it also demonstrates which coefficients drive the model within the catchment in question (a discussion of this is made in results section 6.5.4).

As a first step, a simple univariate sensitivity analysis of each model coefficient is performed. This involves multiple model runs, varying coefficient values by 10% for each run and retaining the model output for each layer as shown in Figure 6.2 for the cw (proportion of P moving to stream) coefficient.

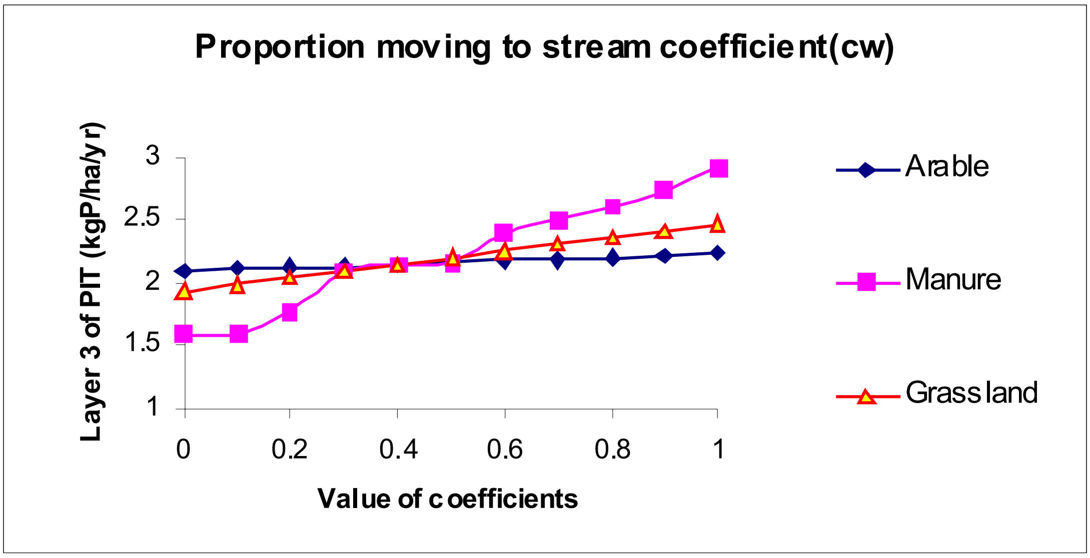


Figure 6.2 Influence of coefficient CW on Layer 3 results

Results from each model run can then be compared to observed data (in this case P flux at the catchment outlet from the Slapton catchment) in order to establish both the sensitivity of the model to variations in that coefficient value and also the optimum value of that coefficient to predict the observed data.

For the Slapton example, observed loss of P from the catchment outlet is 2.19kg/ha/yr, though predicted losses can vary considerably with variations in the coefficient value of CW, from 1.51 to 3kg/ha/yr in the case of manure. It is clear (from Figure 6.2) that the optimal coefficient value lies around 0.4.

Secondly, the optimal values for each coefficient can then be used to parameterise the model for a calibrated model run, also against observed data from the catchment in question. In most cases this approach allows the user to improve model results, based upon available observed data. Where improvements are not made, reliance upon estimated coefficient values is recommended. This method does not consider the parameter interaction or equifinality (i.e. numerous combinations of coefficients give reasonable results – Beven, 1992) that is inherent within such a modelling structure, nevertheless, it permits a straightforward (and transparent) means by which model predictions can be improved where observed data are available. Furthermore, this approach highlights which coefficients should be estimated as accurately as possible for similar catchments where observed data are not available.

6.5 Results for the test catchments

6.5.1 Tier 1

Comparison of Tier 1 model outputs using original and revised coefficients showed the latter estimated slightly higher loads, although differences were small (Figure 6.3). Comparison of modelled TP output with measured TP data showed no discernible difference in using original or revised coefficients. For risk assessment purposes it is, therefore, recommended the revised coefficients are adopted and only output using the revised set of coefficients is described further.

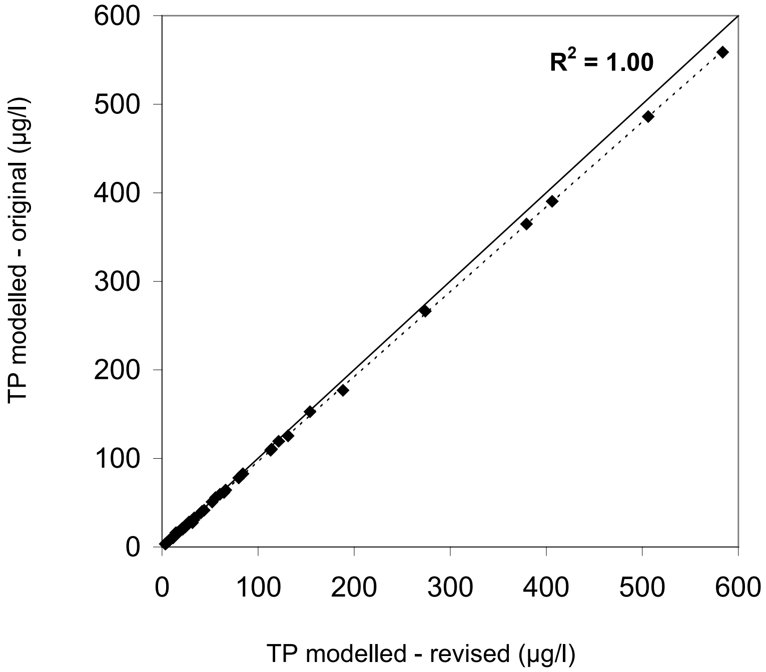


Figure 6.3 Comparison of tier 1 model output using original and revised export coefficients

Catchment TP loads (kg a^{-1}) for the 50 test lakes are presented in Table 6.12. On average, the model suggests 37 % of loads originate from farm animal stocks (plus wild deer in Scotland), 33 % from land cover and 30 % from the human population in the catchment. The combined figure of 70 % from agricultural sources reflects the predominantly rural locations of the test lakes.

Using these catchment TP loads, in-lake TP concentrations and other lake parameters were calculated and are presented in Table 6.13. A comparison of the modelled TP concentrations for all 50 lakes with actual measured data shows a poor relationship, largely due to a single outlier site, White Mere (Table 6.13, Figure 6.4). White Mere is recognised as a rather unique site, with several characteristics, which make it a trap for TP with very efficient internal cycling of nutrients (e.g. no outflow) (Moss *et al.*, 1997). In fact a detailed nutrient budget (Kilinc, 1995), which included measurements of groundwater inputs, support the low modelled TP loads estimated here. If White Mere is removed from the comparison a highly significant correlation (r^2 = 0.59, 47 d.f., $p<0.01$) and more or less 1:1 relationship is observed between modelled and measured TP concentrations (y= 1.09x, Figure 6.5). As would be expected for a national-scale model, there is still variability about this line, particularly as TP concentrations increase (a log-log plot would hide this to some extent). Improvements to the modelled TP concentrations for five of the 50 test sites due to site-specific point-source considerations are presented in Section 7.4.2. These could be accommodated within the GB lakes database, but are not considered here in this generalised application. The tier 1 approach should not, however, be considered a tool appropriate for site-specific studies, but rather as a risk screening tool for the whole of GB.

Table 6.12 Catchment TP loads (kg a^{-1}) from land-use, animals and human population

WBID	NAME	Land Cover	Animals	Population	Total TP Load
704	Eela Water	8	29	0	37
1271	Loch of Girlsta	9	43	0	52
1694	Loch of Boardhouse	380	679	39	1098
14057	Loch Maree	3603	1264	60	4928
18767	Loch Ness	11635	6528	1442	19605
21123	Loch Davan	492	618	68	1178
21189	Loch Kinord	37	59	0	95
24132	Loch Earn	365	1358	153	1876
24447	Loch Lomond	5357	9407	4084	18847
24843	Loch Leven	5309	4410	3266	12985
28847	Bassenthwaite Lake	3797	8013	3317	15127
28955	Ullswater	1129	2575	340	4044
28965	Derwent Water	508	1238	370	2116
28986	Loweswater	111	302	0	413
29183	Wast Water	307	583	42	932
29184	Grasmere	188	619	325	1132
29233	Windermere	2795	3872	5768	12434
29270	Blelham Tarn	74	93	81	247
29321	Coniston Water	587	1026	443	2056
29328	Esthwaite Water	363	364	209	935
29844	Malham Tarn	65	59	30	154
32650	Rostherne Mere	300	247	254	802
32744	The Mere, Mere	109	87	0	196
32761	Llyn yr Wyth-Eidion	32	72	24	128
32804	Tatton Mere	129	72	1723	1924
32948	Llyn Dinam	89	150	169	408
33337	Llyn Coron	475	1461	182	2119
33474	Oak Mere	173	92	86	350
34987	Llyn Tegid (Bala Lake)	1578	4930	802	7310
35091	White Mere	49	28	0	77
35640	Hickling Broad	869	110	371	1351
35655	Barton Broad	5826	1663	6101	13590
36202	Upton Broad	28	23	0	51
37080	Llyn Glanmerin	6	15	0	21
37437	Bugeilyn	6	50	0	57
38394	Llyn Hîr	2	5	0	7
38422	Llyn Eiddwen	4	23	0	27
38525	Llyn Gynon	8	73	0	81
38623	Maes-Llyn	2	2	0	5
39267	Llan Bwch-llyn Lake	32	162	0	195
39796	Upper Talley Lake	23	69	15	106
39813	Lower Talley Lake	7	17	15	39
40067	Llangorse Lake	527	806	309	1641
40571	Llyn Llech Owen	4	9	0	13
40608	Marsworth Reservoir	572	180	4576	5328
41210	Llyn Fach	4	8	0	13
43218	Bolder Mere	48	20	98	166
43651	Albury Mill	357	138	7012	7507
44635	Cinder Hill	212	110	3603	3925
50001	Slapton Ley	1264	1545	514	3323

Table 6.13 Modelled lake parameters using Tier 1 approach

NAME	WBID	Modelled TP ($\mu g\ l^{-1}$)	Modelled Chl.*a* (mean) ($\mu g\ l^{-1}$)	Modelled Chl.*a* (max.) ($\mu g\ l^{-1}$)	Modelled Secchi depth (m)
Eela Water	704	8	2	4	6.89
Loch of Girlsta	1271	6	1	3	7.48
Loch of Boardhouse	1694	24	5	15	3.97
Loch Maree	14057	3	1	2	10.11
Loch Ness	18767	4	1	2	9.93
Loch Davan	21123	40	9	25	3.12
Loch Kinord	21189	14	3	8	5.16
Loch Earn	24132	5	1	2	8.80
Loch Lomond	24447	7	2	4	7.24
Loch Leven	24843	52	11	34	2.76
Bassenthwaite Lake	28847	17	4	10	4.67
Ullswater	28955	8	2	5	6.61
Derwent Water	28965	8	2	4	6.87
Loweswater	28986	19	4	11	4.51
Wast Water	29183	5	1	3	8.05
Grasmere	29184	14	3	8	5.24
Windermere	29233	15	3	8	5.06
Blelham Tarn	29270	24	5	15	3.97
Coniston Water	29321	10	2	5	6.15
Esthwaite Water	29328	21	5	13	4.23
Malham Tarn	29844	18	4	11	4.53
Rostherne Mere	32650	80	16	53	2.26
The Mere	32744	84	17	56	2.20
Llyn yr Wyth-Eidion	32761	66	14	44	2.47
Tatton Mere	32804	379	74	290	1.07
Llyn Dinam	32948	131	27	91	1.78
Llyn Coron	33337	114	23	79	1.90
Oak Mere	33474	113	23	78	1.91
Llyn Tegid (Bala Lake)	34987	18	4	11	4.58
White Mere	35091	44	9	28	3.00
Hickling Broad	35640	121	25	84	1.85
Barton Broad	35655	274	54	203	1.25
Upton Broad	36202	154	31	109	1.65
Llyn Glanmerin	37080	33	7	20	3.43
Bugeilyn	37437	21	5	12	4.28
Llyn Hîr	38394	14	3	8	5.10
Llyn Eiddwen	38422	28	6	17	3.72
Llyn Gynon	38525	16	3	9	4.91
Maes-Llyn	38623	32	7	19	3.51
Llan Bwch-llyn Lake	39267	56	12	36	2.68
Upper Talley Lake	39796	33	7	21	3.41
Lower Talley Lake	39813	42	9	26	3.07
Llangorse Lake	40067	64	13	42	2.50
Llyn Llech Owen	40571	24	5	15	3.98
Marsworth Reservoir	40608	406	79	312	1.04
Llyn Fach	41210	11	3	6	5.71
Bolder Mere	43218	188	38	135	1.50
Albury Mill	43651	584	112	462	0.88
Cinder Hill	44635	506	98	396	0.94
Slapton Ley	50001	60	13	39	2.58

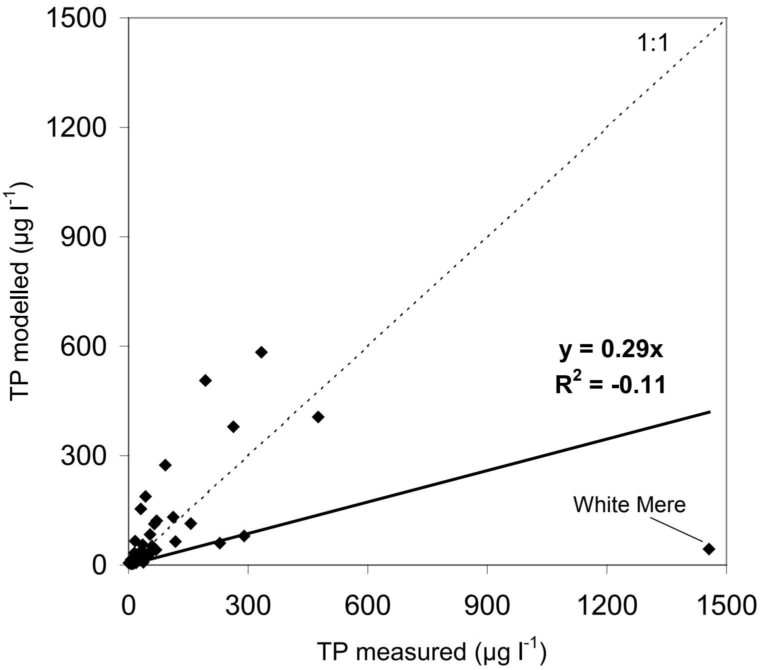


Figure 6.4 Comparison of modelled and measured lake TP concentrations in 50 test lakes

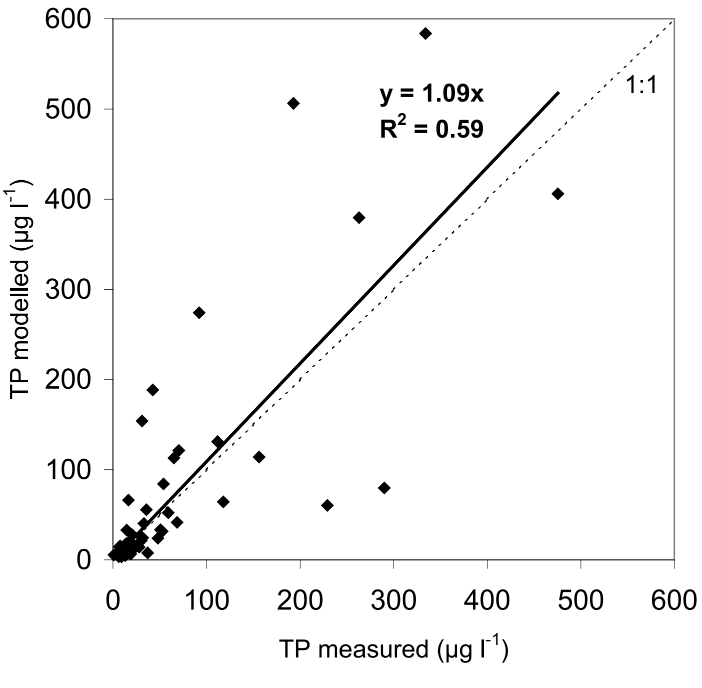


Figure 6.5 Comparison of modelled and measured lake TP concentrations in 49 test lakes (White Mere excluded)

Given the uncertainty in the per capita TP export coefficient for humans (See Chapter 5), a further comparison of modelled and measured data was carried out using a value of 1.0 kg a^{-1}, rather than 0.4 kg a^{-1}. This value is at the upper end of the range of per capita export values estimated for specific STWs, and so represents more a worse-case scenario. The comparison illustrates this clearly, although there is a similar level of correlation. The relationship observed is not 1:1, modelled TP concentrations generally being double measured values (y = 2.03x) (Table 6.14, Figure 6.6). Despite this, at 39 % of sites the model still underestimated measured TP concentrations, compared with 47% of sites using a per capita TP export coefficient for humans of 0.4 kg a^{-1}. No clear pattern is observed as to which sites modelled TP under-estimates measured TP, although it is observed for all Scottish sites except Loch Davan.

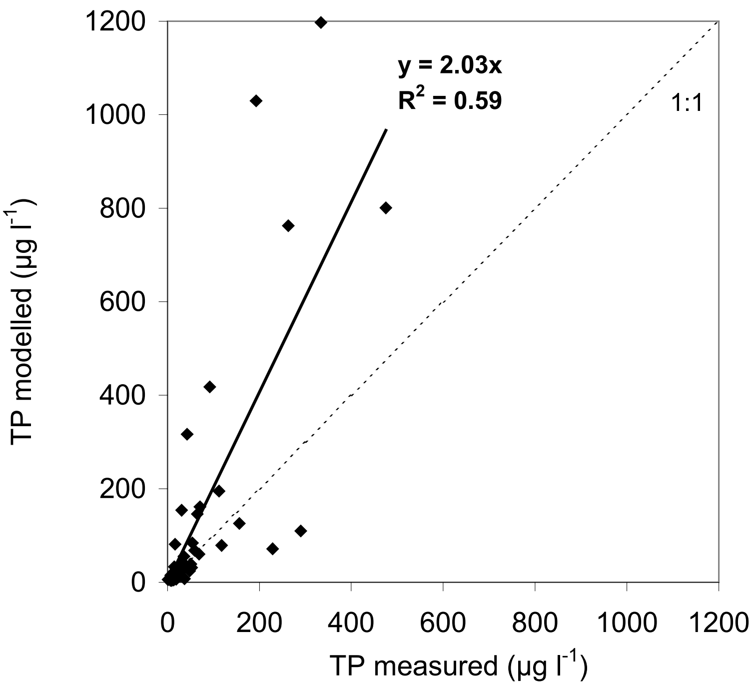


Figure 6.6 Comparison of modelled and measured TP concentrations in 49 test lakes using per capita TP export coefficient of 1.0 kg a^{-1} for human population

6.5.2 Tier 2

Figures 6.7 to 6.9 illustrate the range of P pressure values from both surface and subsurface sources for three catchments using the tier 2 approach. Also shown is the average P pressure value for the whole catchment, which enables catchments to be ranked in order of P pressure (highest to lowest) as shown in Table 6.15 for all eight catchments investigated. Output for all the sites tested using the Tier 2 approach is given in Appendix 5. Smaller catchments, such as Fenemere (Figure 6.8) do not show great variation within catchment due to the homogeneity of input data used to drive the approach. However, larger catchments, such as Slapton Ley (Figure 6.9) show considerable spatial variation in surface P pressure with numerous ‘hotpsots’ or areas where P pressures on surface waters are ranked highly.

Table 6.14 Comparison of measured TP with modelled TP using two per capita export coefficients for humans (0.4 and 1.0 kg a^{-1})

Name	WBID	Measured TP ($\mu g l^{-1}$)	Modelled TP (0.4)	Modelled TP (1.0)
Eela Water	704	37	8	8
Loch of Girlsta	1271	19	6	6
Loch of Boardhouse	1694	31	24	25
Loch Maree	14057	9	3	3
Loch Ness	18767	6	4	4
Loch Davan	21123	33	40	43
Loch Kinord	21189	28	14	14
Loch Earn	24132	13	5	5
Loch Lomond	24447	9	7	9
Loch Leven	24843	59	52	68
Bassenthwaite Lake	28847	21	17	22
Ullswater	28955	10	8	9
Derwent Water	28965	7	8	9
Loweswater	28986	16	19	19
Wast Water	29183	1	5	6
Grasmere	29184	24	14	18
Windermere	29233	14	15	23
Blelham Tarn	29270	32	24	34
Coniston Water	29321	7	10	12
Esthwaite Water	29328	30	21	27
Malham Tarn	29844	17	18	23
Rostherne Mere	32650	290	80	110
The Mere	32744	54	84	84
Llyn yr Wyth-Eidion	32761	17	66	81
Tatton Mere	32804	263	379	763
Llyn Dinam	32948	112	131	195
Llyn Coron	33337	156	114	126
Oak Mere	33474	65	113	146
Llyn Tegid (Bala Lake)	34987	14	18	20
White Mere	35091	1456	44	44
Hickling Broad	35640	70	121	161
Barton Broad	35655	92	274	418
Upton Broad	36202	31	154	154
Llyn Glanmerin	37080	15	33	33
Bugeilyn	37437	18	21	21
Llyn Hîr	38394	7	14	14
Llyn Eiddwen	38422	21	28	28
Llyn Gynon	38525	8	16	16
Maes-Llyn	38623	53	32	32
Llan Bwch-llyn Lake	39267	36	56	56
Upper Talley Lake	39796	51	33	39
Lower Talley Lake	39813	69	42	60
Llangorse Lake	40067	118	64	79
Llyn Llech Owen	40571	48	24	24
Marsworth Reservoir	40608	476	406	801
Llyn Fach	41210	10	11	11
Bolder Mere	43218	43	188	316
Albury Mill	43651	334	584	1197
Cinder Hill	44635	193	506	1029
Slapton Ley	50001	229	60	72

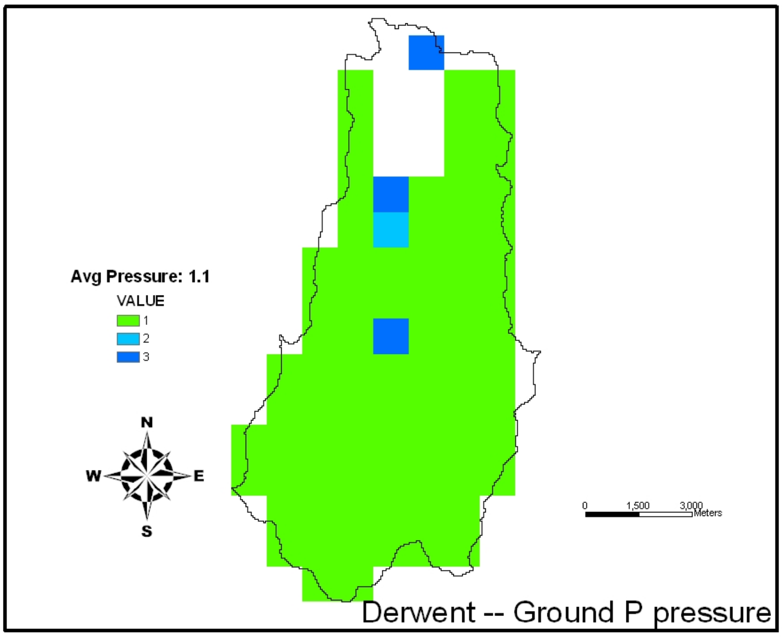
Differences in intra-catchment variation are notable between subsurface and surface sources. Low levels of variation are associated with P pressure from subsurface areas throughout all the catchments modelled. Furthermore, subsurface P pressure rarely exceeds the low pressure category, with the exception of the Derwent Water and Fenemere catchments (Figures 6.7 and 6.8). From this initial analysis of a small group of catchments it would seem that surface sources of P pressure are more important (and more spatially heterogeneous) than subsurface sources, though it is recognised that without full validation against observed data this may simply be a relict of the input data available (i.e. lower resolution data for subsurface areas).

In general, the catchments modelled here represent low to medium risk catchments on a national scale. Values rarely exceed 3-4 (medium risk) and often the majority of each catchment is classified as 0-2 (i.e. low risk). The approach does, however, highlight areas that are high risk and therefore should be targeted by catchment management. Finally, no general trend can be established between upland and lowland catchments, or catchments that might be treated as being in similar regions of the UK. It, therefore, seems fair to assume that within catchment variability captured by the spatially variable datasets used in this approach is an important driver of P pressure to surface waters which must be incorporated in order to model P pressure risk at the national scale.

Table 6.15 Catchments used for the tier 2 approach, ranked on averaged surface P pressure across each catchment

Catchment	Averaged Ground P Pressure	Averaged Surface P Pressure	Rank
Slapton Ley	1.7	3.02	1
Comber Mere	1.89	3	2
Fenemere	2.5	2.7	3
Malham Tarn	1.7	2	4
Ullswater	1.01	1.9	5
Oak Mere	2.7	1.7	6
Derwent Water	1.1	1.6	7
Wast Water	1.05	1.34	8

Producing Figures 6.7–6.9 has highlighted the limitation of extracting individual catchment data from the results produced on a national scale (Pope and Hughes, pers. comm.). Certain grid cells that are not identified on a national scale map, contain no data values when viewed at the catchment scale (e.g. Figure 6.8) which impacts not only upon the visual comparisons between techniques, but also on the catchment average surface and subsurface P pressure.



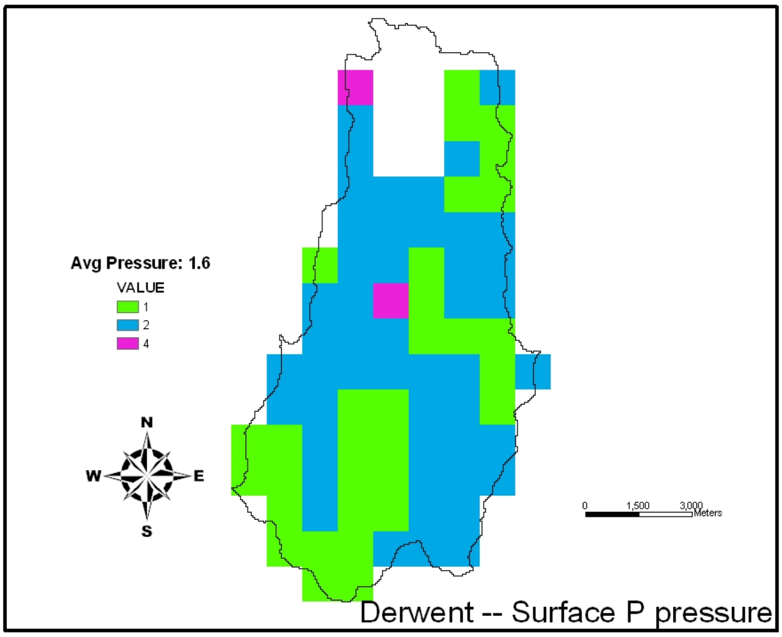
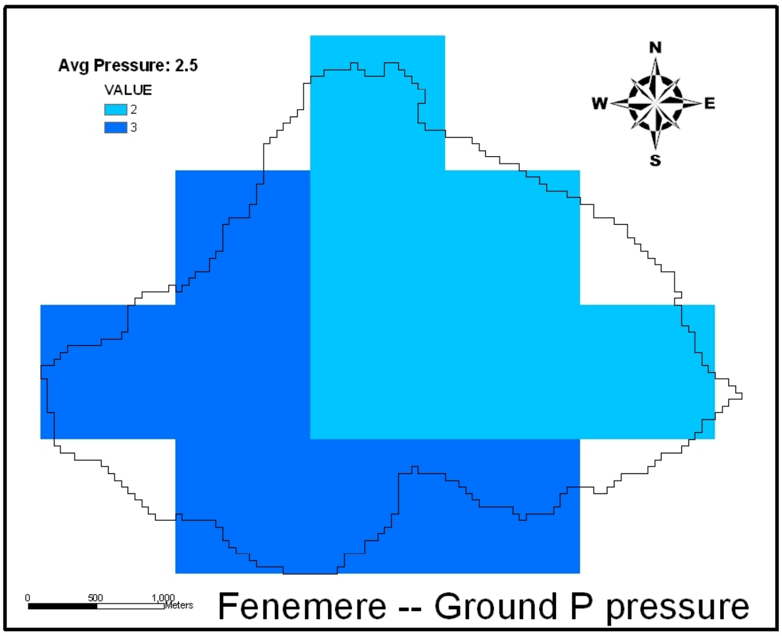


Figure 6.7 Subsurface and surface P pressure maps for the Derwent catchment using the tier 2 approach



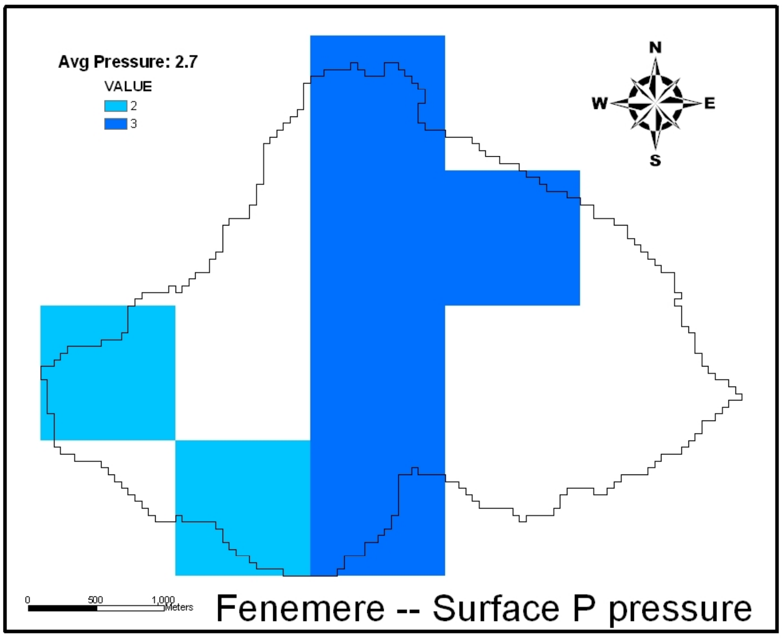
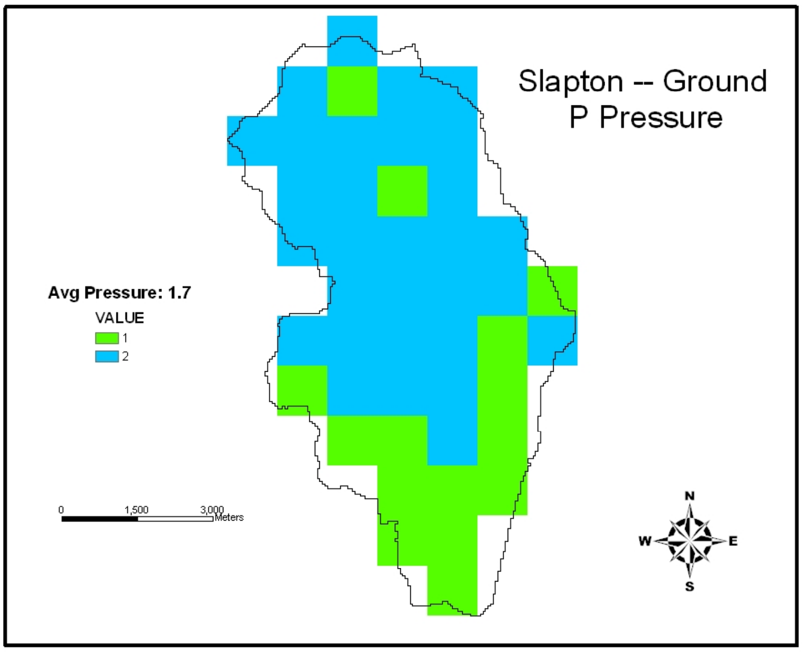


Figure 6.8 Subsurface and surface P pressure maps for the Fenemere catchment using the tier 2 approach



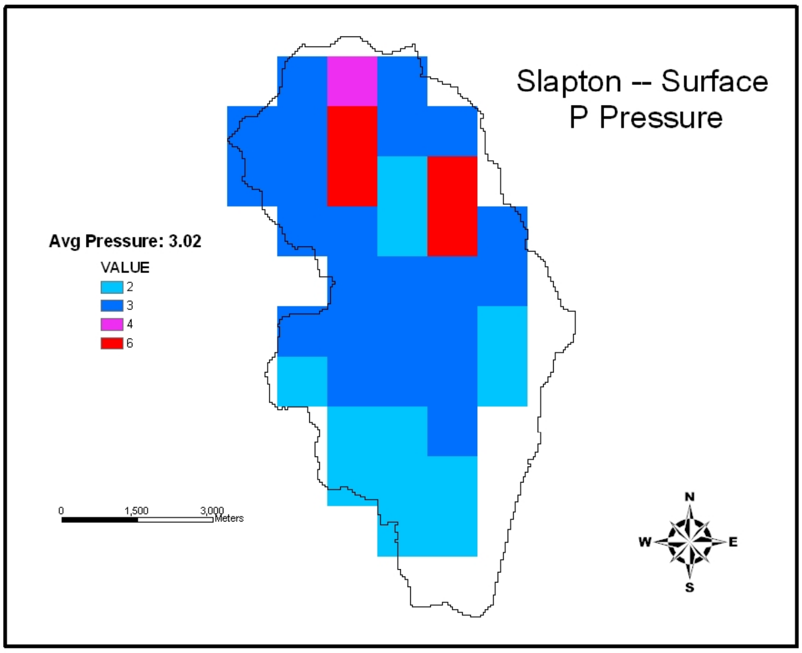


Figure 6.9 Subsurface and surface P pressure maps for the Slapton Ley catchment using the tier 2 approach

6.5.3 Tier 3a

Table 6.16 describes modelled response using the tier 3a (PIT uncalibrated) approach for 21 of the 50 test lakes outlined in section 6.3.1. The test catchments are ranked in terms of predicted output from layer 3 of the model, which represents phosphorus delivery from the landscape. Perhaps not surprisingly, those catchments demonstrating higher availability of phosphorus within the landscape (shown by Layer 1 values) also tend to produce higher predictions of phosphorus transfer and delivery, thus ranking based on phosphorus availability would be largely similar to those shown below.

Table 6.16 Modelled catchment average response using the Tier 3a – uncalibrated PIT approach for layers 1, 2 and 3

Catchment	Area (km^2)	Layer1 (kgP/ha/yr)	Layer2 (kgP/ha/yr)	Layer3 (kgP/ha/yr)
Slapton Ley	46	59.25	4.38	2.14
Blelham Tarn	4	24.37	0.97	1.16
Esthwaite Water	18	21.52	0.83	0.99
Loch Leven	160	96.81	0.16	0.92
Windermere	253	12.31	0.35	0.43
Llyn Coron	22	8.21	0.47	0.2
Comber Mere	6	6.03	0.31	0.16
Fenemere	12	5.88	0.31	0.13
Llangorse Lake	23	9.81	0.51	0.13
Rostherne Mere	10	5.55	0.26	0.06
Bassenthwaite	358	3.42	0.08	0.05
Betley Mere	4	4.41	0.15	0.04
Malham Tarn	4	4.35	0.09	0.03
Marsworth Res.	14	7.31	0.23	0.03
Llyn Tegid	147	3.2	0.06	0.03
Ullswater	150	2.65	0.04	0.03
Wast Water	45	2.78	0.01	0.03
Coniston Water	64	2.78	0.03	0.02
Barton Broad	109	2.57	0.06	0.01
Derwent Water	90	2.42	0.02	0.01
Oak Mere	5	2.68	0.05	0.01

Figures 6.10 to 6.12 illustrate for three sites the spatial dimension to the results shown in Table 6.16. Thus, it is possible to identify hotspots or critical source areas within the catchments that dominate the average output shown above. Outputs for all the sites tested using the Tier 3a approach are given in Appendix 5. Such spatially distributed results identify areas within catchments that may require changes in land use or remediation measures to reduce overland flow, soil erosion and consequent removal of phosphorus from hillslopes into channels. By focussing on maps of phosphorus distribution at the three levels output from the PIT approach, it is possible to determine whether it is necessary to reduce available phosphorus, or perhaps introduce measures (such as buffer strips or cover cropping) that reduce the proportion of phosphorus transported from the land to surface water bodies.

Concentrating on Figure 6.10, which describes results from the Barton Broad catchment, it is evident that certain areas within the catchment contain high values of available phosphorus (up to 20 kg/ha/yr) as seen in the output from Layer 1. However, when results from Layer 2 are considered, it is clear that much of this available phosphorus will not be transferred within the landscape. This is highlighted for similar areas, in the south-east of the catchment, that are predicted to transfer less than 10% of the phosphorus available from layer 1. Furthermore, when results from layer 3 are considered, a maximum of 0.13 kg/ha/yr of phosphorus are predicted to be delivered from the land to surface waters, hence the relatively low ranking of the Barton Broad catchment.

In contrast, results from the Slapton Ley catchment (Figure 6.12) demonstrate that high available phosphorus loads (a catchment average of 59.25 kg/ha/yr) can propagate through the system to produce high phosphorus delivery loads as well – on average 2.14 kg/ha/yr. In such catchments, it is clear that remediation must target not only the availability of phosphorus in the landscape, but also the transfer <u>and</u> delivery of phosphorus if successful attempts are to be made to reduce loads to surface waters. The results from Loch Leven (Figure 6.11) are presented to illustrate that the approach is potentially applicable to Scotland if soils and hydrology data are made available.

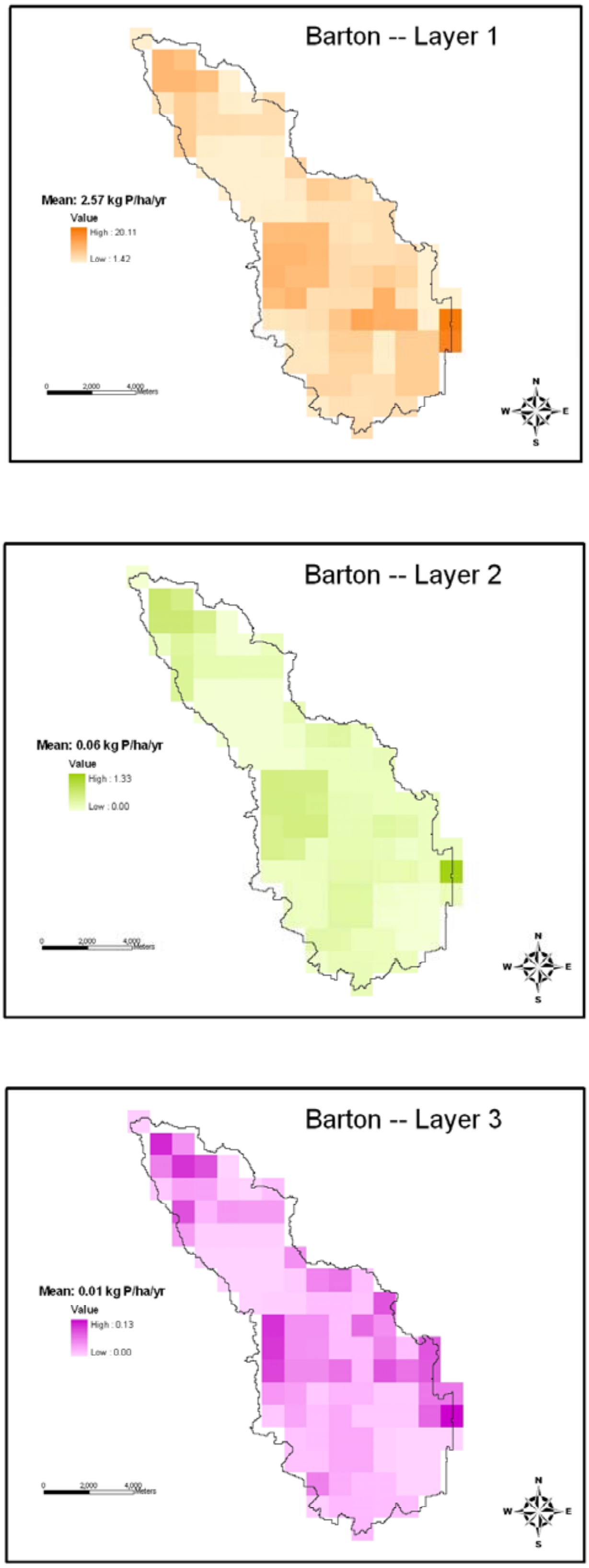


Figure 6.10 Tier 3a results (PIT Silver standard) for Barton Broad

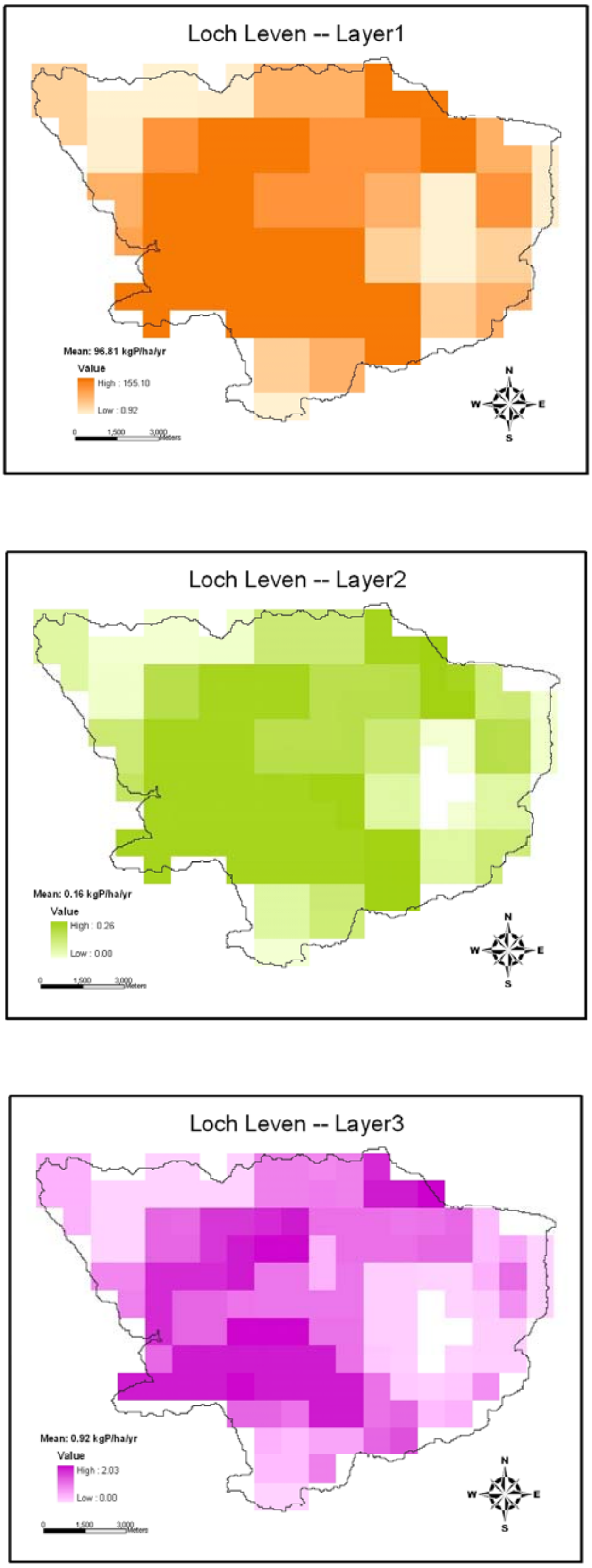


Figure 6.11 Tier 3a results (PIT Silver standard) for Loch Leven

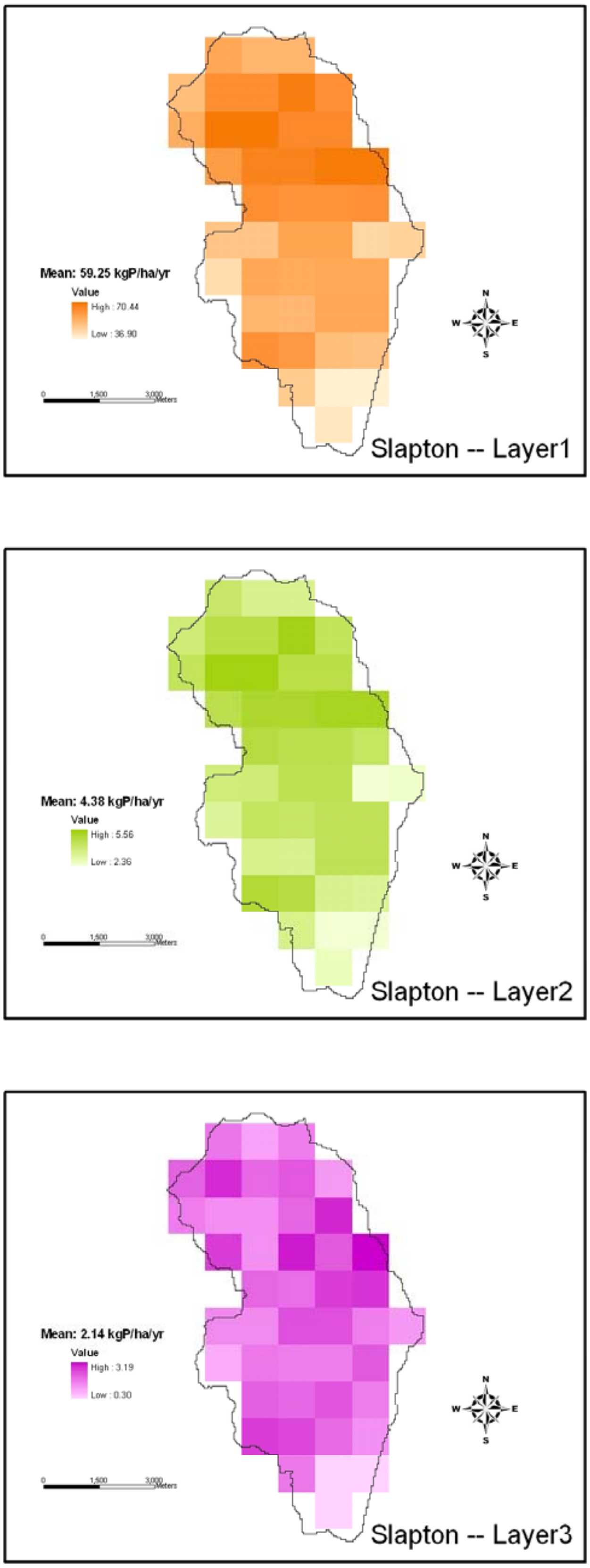


Figure 6.12 Tier 3a results (PIT Silver standard) for Slapton Ley

6.5.4 Tier 3b

Figures 6.14–6.16 illustrate the spatially distributed results generated using the calibrated version of the PIT model for three catchments. Outputs for all the sites tested using the Tier 3b approach are given in Appendix 5. Though patterns look similar to the tier 3a (uncalibrated) results, crucially, these results are calibrated, with excellent fits to observed data (as shown in Figure 6.13). Consequently, there is more confidence in the absolute values of spatial predictions as they are based on very good predictions of catchment average phosphorus data (as is shown by the 0.99 r^2 value).

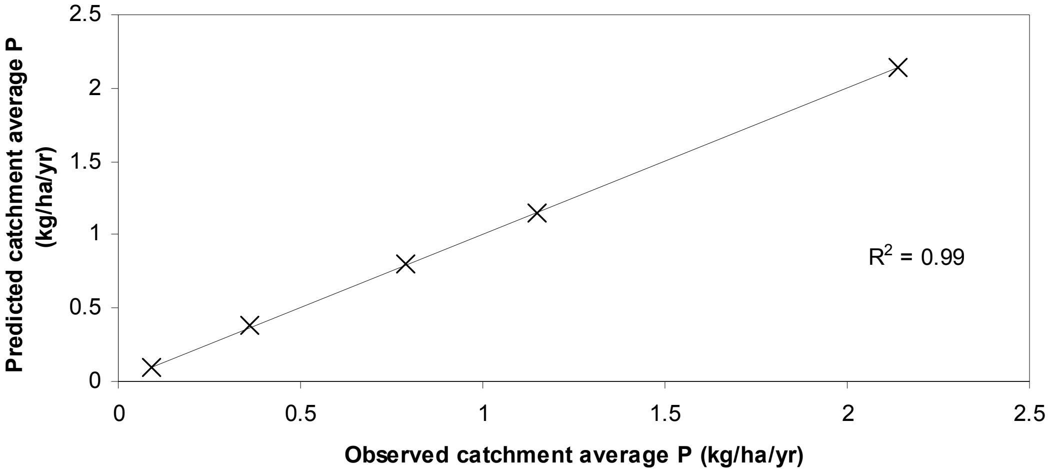


Figure 6.13 Observed against predicted catchment average P from the Tier 3b (PIT calibrated) model runs

Though detailed observed data are not available to perform such calibration on the majority of UK catchments, the Tier 3b approach demonstrates how well the model can perform when applied to a data rich catchment. Furthermore, by calibrating key, sensitive coefficients, it is demonstrated that the PIT model can predict a wide range of catchment responses in terms of observed P, from in excess of 2 kg/ha/yr, as observed at Slapton Ley, to as low as <0.1 kg/ha/yr as observed at Barton Broad.

As more observed data become available for a range of UK catchments it will be possible to evaluate the response of both the Tier 3a and Tier 3b version of PIT on a greater number of catchment types, than those within this analysis. This will permit the application of the PIT model, with an explicit consideration of model performance against observed data for more than just the 5 catchments examined here.

In summary, despite the lack of observed data for many lakes and their catchments, the performance of the model against those datasets that do exist is encouraging as a basis for further development of the PIT model to improve predictions where datasets do not exist (i.e. data poor catchments) as a first step to addressing the problem of prediction in ungauged basins.

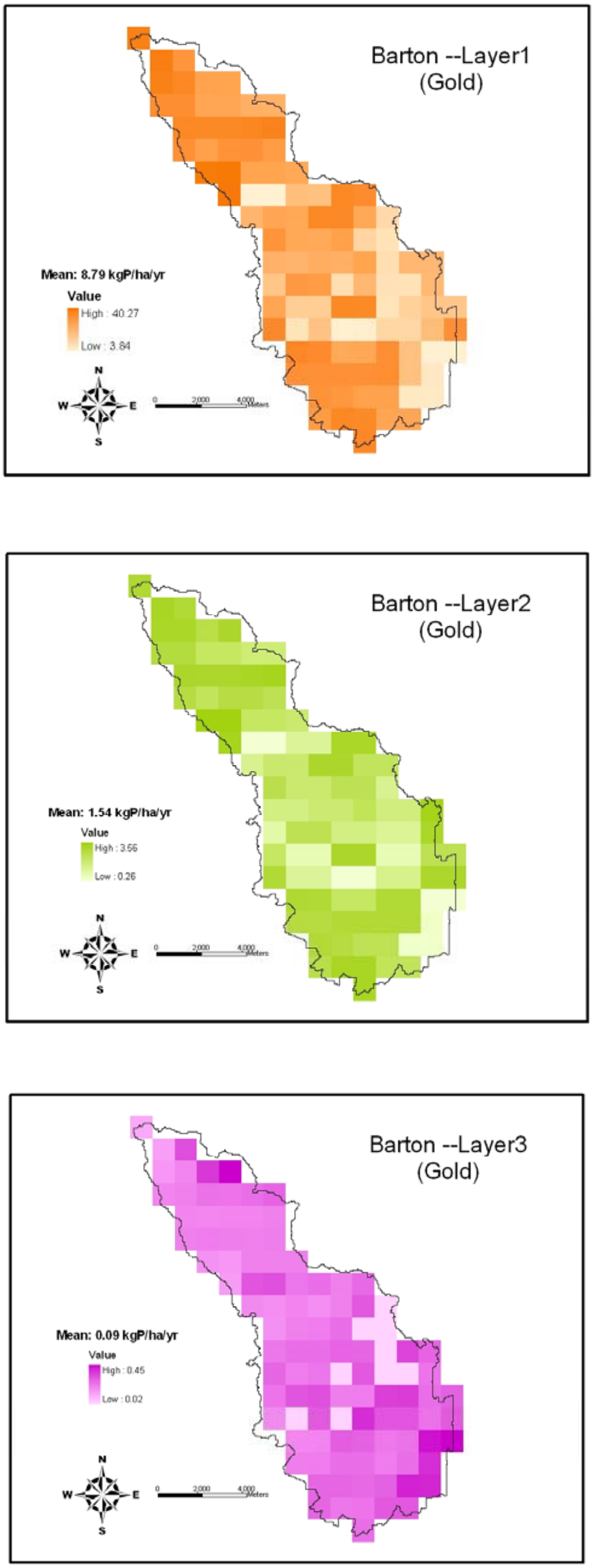


Figure 6.14 Tier 3b results (PIT calibrated) for Barton Broad

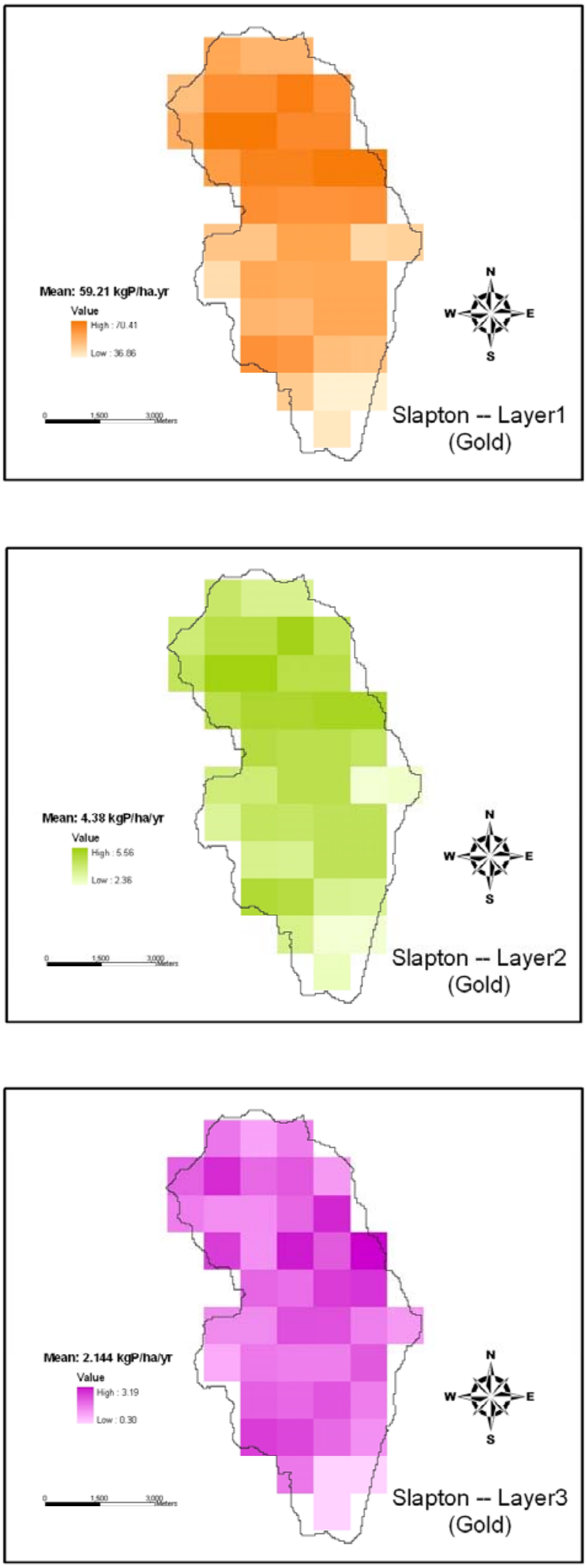


Figure 6.15 Tier 3b results (PIT calibrated) for Slapton Ley

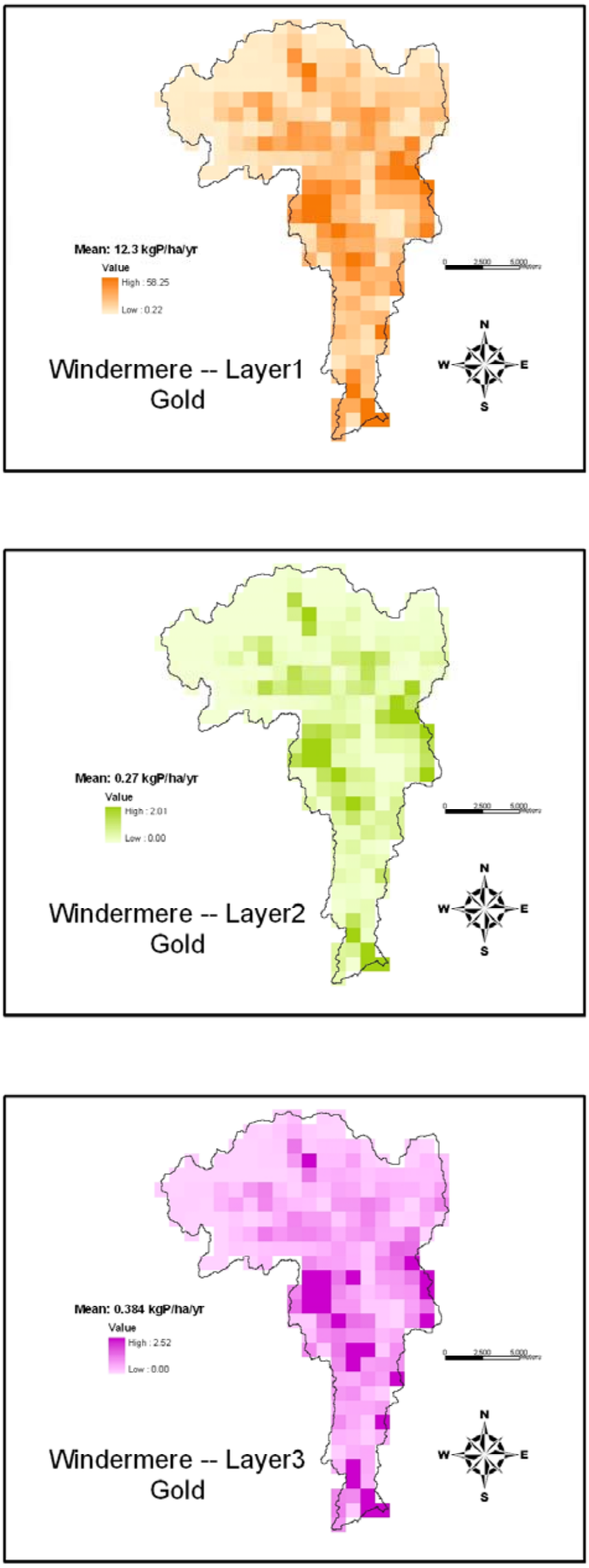


Figure 6.16 Tier 3b results (PIT calibrated) for Windermere

Table 6.17 Catchment average results from the PIT (calibrated) model runs against observed data

Catchment	Layer1(kgP/ha/yr)	Layer2 (kgP/ha/yr)	Layer3 (kgP/ha/yr)	Observed (kgP/ha/yr)
Slapton	59.21	4.38	2.144	2.14
Blelham	24.36	0.93	1.146	1.15
Esthwaite	21.5	0.79	0.8	0.79
Windermere	12.3	0.27	0.384	0.36
Barton	8.79	1.54	0.09	0.092

6.6 Comparison of GBI (Tier 1), DPSR (Tier 2), and PIT (Tier 3a and 3b) results

Table 6.18 below, outlines the key differences between the three approaches used in this project. In order to compare the outputs from the three modelling approaches a ranking approach was used. The test lakes were ranked in terms of the outputs from the three modelling approaches based on the diffuse agricultural component of the model only i.e. for the Tier 1 approach, only TP loads from land cover and animals were included. The diffuse TP loads from the Tier 1 output were then weighted relative to the catchment area to provide a TP export per catchment land area (kg ha^{-1} a^{-1}), which is comparable to the output from the PIT model (both Tier 3a and 3b). The output from the Tier 2 approach ranks sites according to P pressure, but not in quantified units.

Table 6.18 Key differences between tiers 1-3b

Criteria	Tier 1	Tier 2	Tier 3a	Tier 3b
Method	GBI	PDRS	PIT (uncalibrated)	PIT (calibrated)
Spatial output	Lumped	Semi-distributed	Distributed	Distributed
Scale	Spatially lumped Great Britain	$1km^2$ England	$1km^2$ Great Britain	$1km^2$ Data rich sites
Timestep	Temporally lumped	Temporally lumped	Annual	Annual
Input data requirements	Low	Medium	High	High
Calibrated?	No	(not possible)	No	Yes
Validated?	No	(not possible)	No	Yes
Output	Total in lake P	P pressure risk	Total P per $1km^2$ or catchment ave	Total P per $1km^2$ or catchment ave

Table 6.19 compares the ranks for the 50 test lakes. All sites are ranked for the Tier 1 approach, 6 for the tier 2 approach, 18 for the tier 3a approach and 5 for the tier 3b approach.

Table 6.19 Test lakes ranked in terms of diffuse phosphorus pressures in the catchment using Tiers 1, 2, 3a and 3b approaches

	Risk Rank			
Lake	**Tier 1**	**Tier 2**	**Tier 3a**	**Tier 3b**
Llyn Coron	1		6	
Llyn Dinam	2			
Upton Broad	3			
Barton Broad	4		16	5
The Mere, Mere	5			
Llan Bwch-llyn lake	6			
Slapton Ley	7	1	1	1
Loch Leven	8		4	
Llyn Glanmerin	9			
Llyn yr Wyth-Eidion	10			
Llangorse lake	11		7	
Maes-Llyn	12			
Rostherne Mere	13		8	
Llyn Eiddwen	14			
Lower Talley Lake	15			
Marsworh Reservoir	16		11	
Oak Mere	17	4	18	
Upper Tally Lake	18			
Loweswater	19			
Tatton Mere	20			
Bugeilyn	21			
Hickling Broad	22			
Llyn Tegid (Bala lake)	23		12	
Llyn Llech Owen	24			
Blelham Tarn	25		2	2
Esthwaite Water	26		3	3
White Mere	27			
Albury Mill	28			
Cinder Hill	29			
Bassenthwaite lake	30		9	
Llyn Hir	31			
Loch Davan	32			
Malham Tarn	33	2	10	
Llyn Gynon	34			
Grasmere	35			
Loch of Boardhouse	36			
Llyn Fach	37			
Windermere	38		5	4
Coniston Water	39		15	
Ullswater	40	3	13	
Bolder Mere	41			
Wast Water	42	6	14	
Derwent Water	43	5	17	
Loch Lomond	44			
Loch Earn	45			
Loch of Girlsta	46			
Eela Water	47			
Loch Maree	48			
Loch Kinord	49			
Loch Ness	50			

Output from the three modelling approaches show some consistency in terms of predicting which lakes are most ‘at risk’ from diffuse agricultural sources of P. (Table 6.19). For example, Slapton Ley is a well-studied catchment with one of the best historical datasets describing P fluxes (at the catchment outlet) in the UK. Within this analysis it is the only site ranked by all four approaches and is ranked highest (i.e. most sensitive in terms of P pressure and in terms of catchment average P) in Tiers 2, 3a and 3b. The tier 1 approach also ranks Slapton Ley high in terms of P pressure (7/50) (Table 6.19).

There are, however, some clear differences between the model predictions. For example, Barton Broad is ranked 4th out of 50 using the Tier 1 approach, whereas it is ranked 16th out of 18 using the uncalibrated PIT (Tier 3a) approach and 5th out of 5 using the calibrated PIT (Tier 3b) approach. The latter low risk rankings have previously been suggested to be a consequence of the low relief of this catchment, the predominance of groundwater flows and the importance of in-stream P processing within the water body and its tributaries (Johnes *et al*., 2003). These criteria are not taken into account in the simplified approach used in Tier 1; none of the approaches currently include provision for in-stream nutrient processing. The measured in-lake TP concentrations for Barton Broad are, however, relatively high, more in agreement with its high ranking for P pressure using the Tier 1 approach.

The three modelling approaches show some consistency in terms of predicting which lakes are most ‘at risk’ from diffuse agricultural sources of P. The fact that they are not completely consistent is unsurprising, as the models have been designed using different methodologies, for different purposes and with different objectives, as highlighted in Table 6.18. Rather they should be viewed as providing a suite of different tools for different tasks that may be applied in a step-wise fashion to hone-in on high risk areas of vulnerable catchments in terms of mitigation to reduce diffuse P delivery from land to water.

The GB Lakes Inventory approach (Tier 1) is suitable for general risk assessment purposes at a national scale, and can be applied across the whole of GB using currently available data at the 5 km^2 scale. It is useful for identifying those lakes that may be most at risk to increases in total P from point and diffuse sources. However, the lumped approach means that it is not possible to identify the spatial distribution of high or low risk areas within a catchment, or quantify losses from specific areas. Further validation and sensitivity analysis is needed, as levels of uncertainty associated with the export coefficients are not known.

The Pressure Delivery Risk Screening matrix approach (Tier 2) also provides a nationally applicable method for modelling diffuse agricultural P pressure from surface and subsurface sources, and it provides an initial screening of the spatial distribution of P pressure within a catchment at the 1 km^2 scale. However, it remains a qualitative approach and so it cannot provide a quantitative assessment of the spatial distribution of the pressure. Further work is required to justify the position of the pressure boundaries using water quality monitoring data, and to examine the uncertainty of the approach when it is applied to small catchments.

The uncalibrated and calibrated PIT approach (Tier 3a and b, respectively) identifies the spatial distribution of the risk of P delivery from agricultural land to the receiving water, and quantifies this delivery. Export coefficients used in the uncalibrated PIT model have been developed (Heathwaite et al., 2003), so in the absence of observed data to calibrate model output these are the recommended coefficient values to be used in the PIT model. These coefficients can be altered on a site-specific basis (i.e. through calibration) where quality-assured observed data exist with which to evaluate model performance. The PIT model is aimed more at describing catchment wide diffuse P behaviour and for identifying where to target remediation management action. Currently, the PIT model does not permit prediction of in-lake TP concentrations; being aimed more at describing catchment wide behaviour.

6.7 Recommended guidance for the risk assessment

6.7.1 Current recommended guidance (including coefficients) for use in basic risk assessment (Tier 1)

- Comparison of measured and modelled data suggests Tier 1 approach is a robust method for **general** risk assessment purposes at a national scale
- It provides a consistent approach across the whole of GB, although the spatial resolution of agricultural data differs slightly in Scotland compared with England and Wales, and deer numbers have only been included for Scotland.
- Need further work to explore variability in measured/modelled comparison
- Validation and sensitivity analysis of export coefficients is essential as levels of uncertainty associated with coefficients are currently unquantified
- Comparison differences could equally be due to unrepresentative measured water quality data, so improved quality water databases are needed.

6.7.2 Current recommended guidance for use in basic risk assessment (Tier 2)

- Tier 2 results permit quick and nationally-based approach to modelling phosphorus pressure from surface and subsurface sources
- Coefficients need further calibration/validation to constrain uncertainty
- The position of the pressure boundaries must be reviewed using monitoring data from catchments to justify the position of the boundaries
- The DPSR method for groundwater catchments requires further work when the data are available to decide the proportions of low-medium-high pressure within a groundwater catchment that should be used to give the appropriate pressure band.
- Furthermore, the uncertainty of the approach increases for small catchments; so this must be borne in mind as a limitation

6.7.3 Recommended guidance for use in detailed risk assessment (PIT Tier 3a)

Tables 6.20-6.22 list the current export coefficients used in the uncalibrated (Tier 3a) PIT model. In the absence of observed data to calibrate model output these are the recommended coefficient values to be used in the PIT model.

Table 6.20 Layer 1 coefficients for livestock and fertilizer inputs for PIT model

Livestock type	P excretion kg head^{-1} a^{-1}	Crop	P applied kg ha^{-1} a^{-1}
Dairy adult	29.0	winter wheat	22
Dairy young stock	17.5	winter barley	22
Beef > 2 years	17.5	spring barley	16
Beef 1-2 years	14.2	oats/rye	21
Cattle < 1 year	5.5	potatoes	81
Sheep	1.2	field beans and peas	15
Lambs	0.3	oilseed rape	21
Breeding sows	8.0	linseed	11
Small fattening pigs	3.1	sugar beet	22
Large fattening pigs	5.1	maize	29
Laying hens	0.233	kale/cabbage etc stock feed	18
Broiler hens	0.187	turnips/swedes etc stock feed	18
		horticultural/hops	17
		set aside, fallow	0
		grass under 5 years	13
		grass 5 years and over	9

Table 6.21 Layer 2 Coefficients for P transfer (a) soluble P transfer and (b) particulate P transfer

(a) Soluble P	Surface flow/ drain flow	Subsurface matrix flow	(b) Particulate P Class	Risk	Surface flow/ Drain flow	Subsurface matrix flow
Olsen P	0.05	0.01	1	Extreme	0.1	0.008
Manure	0.10	0.08	2	Very high	0.05	0.005
Fertiliser	0.10	0.08	3	High	0.025	0.001
			4	Moderate	0.01	0.0005
			5	Slight	0.005	0.0001
			6	Negligible	0.001	0

Table 6.22 Layer 3 Coefficients for P delivery for the PIT model

Subsurface flow coefficients

Land Use	Delivery through drains as bypass flow	Delivery through deep percolation to groundwater as matrix flow
Arable/ Grassland	1.0	0.05

Surface flow coefficients

Land Use		Within-field delivery: plot to field edge	Edge-of-field delivery: field edge to stream via gateway	Edge-of-field delivery: field edge to stream via breakthrough
Arable/ Grassland	Particulate P	0.9	0.5	0.6
	Olsen P	0.9	0.5	0.8
	Manure	0.9	0.5	0.9
	Fertiliser	0.9	0.5	0.8

Coefficients for proportion moving to stream based on HOST

Class	HOST classes	Arable	Grassland	Manure
1	1,2,4,11,13	0.2	0.2	0.2
2	3,5	0.3	0.3	0.3
3	9,10,14,16,17	0.4	0.4	0.4
4	6,24	0.6	0.6	0.6
5	7,8,15,18,21,25	0.8	0.8	0.8
6	12,19,20,22,23,26,27,28,29	1.0	1.0	1.0

Modifier coefficients for slope angle

Class	Slope range (^{o})	Arable	Grassland	Manure
1	< 1	0.05	0.05	0.05
2	1 - 3	0.3	0.3	0.3
3	3 - 7	0.6	0.6	0.6
4	> 7	0.9	0.9	0.9

Coefficients for the direct delivery of P to receiving waters from farm hard-standings

Coefficient	Proportion excreta voided	Proportion excreta not collected
Dairy cattle - road	0.05	1.0
Dairy cattle - farmyard	0.18	0.1
Beef cattle - farmyard	0.25	0.5

These coefficients can be altered on a site-specific basis (i.e. calibration), as carried out for the Tier 3b (calibrated) version of the PIT model where quality observed data exist with which to evaluate model performance. An example of this is provided for Windermere (Tables 6.23).

6.7.4 Recommended guidance for use in detailed risk assessment (PIT Tier 3b)

Where observed data are available to calibrate the model it is recommended that they are employed to calibrate coefficients as below as a means of best producing catchment average P, and therefore simulating spatial patterns of P.

It is recognised that the data to calibrate these coefficients are rarely available outside intensively monitored research catchments, so this is seen as a limitation of the Tier 3b approach. However, as more data become available it is anticipated that calibration of the PIT model can be furthered in this site-specific manner. Furthermore it is a recommendation of this project that continued data collection of P fluxes at a range of nested catchment scales is made to aid future model development in the form of calibration and validation.

Table 6.23 Layer 2 Coefficients for P transfer (a) soluble P transfer and (b) particulate P transfer - Windermere

(a) Soluble P	Surface flow/ drain flow	Subsurface matrix flow	(b) Particulate P Class	Risk	Surface flow/ Drain flow	Subsurface matrix flow
Olsen P	0.04	0.01	1	Extreme	0.1	0.008
Manure	0.04	0.08	2	Very high	0.05	0.005
Fertiliser	0.04	0.08	3	High	0.025	0.001
			4	Moderate	0.01	0.0005
			5	Slight	0.005	0.0001
			6	Negligible	0.001	0

Table 6.24 Layer 3 Coefficients for P delivery for the PIT model - Windermere

Subsurface flow coefficients

Land Use	Delivery through drains as bypass flow	Delivery through deep percolation to groundwater as matrix flow
Arable/ Grassland	1.0	0.05

Surface flow coefficients

Land Use		Within-field delivery: plot to field edge	Edge-of-field delivery: field edge to stream via gateway	Edge-of-field delivery: field edge to stream via breakthrough
Arable/ Grassland	Particulate P	0.9	0.5	0.6
	Olsen P	0.9	0.5	0.8
	Manure	0.33	0.5	0.9
	Fertiliser	0.9	0.5	0.8

Coefficients for proportion moving to stream based on HOST

Class	HOST classes	Arable	Grassland	Manure
1	1,2,4,11,13	0.2	0.2	0.27
2	3,5	0.3	0.3	0.27
3	9,10,14,16,17	0.4	0.4	0.27
4	6,24	0.6	0.6	0.27
5	7,8,15,18,21,25	0.8	0.8	0.27
6	12,19,20,22,23,26,27,28,29	1.0	1.0	0.27

Modifier coefficients for slope angle

Class	Slope range (^{o})	Arable	Grassland	Manure
1	< 1	0.05	0.05	0.39
2	1 - 3	0.3	0.3	0.39
3	3 - 7	0.6	0.6	0.39
4	> 7	0.9	0.9	0.39

Coefficients for the direct delivery of P to receiving waters from farm hard-standings

Coefficient	Proportion excreta voided	Proportion excreta not collected
Dairy cattle - road	0.2	0.54
Dairy cattle - farmyard	0.2	0.54
Beef cattle - farmyard	0.2	0.54

6.8 Recommendations for future development

This project has developed a three tier risk assessment methodology for determining nutrient impacts in surface freshwater bodies. The Tier 1 approach has been tested on 50 lakes, the tier 2 approach on 8 lakes, the tier 3a approach on 21 lakes and the tier 3b approach on 5 lakes.

To date the PIT model has been tested using data from the Windermere, Slapton Ley, Esthwaite Water, Barton Broad and Blelham Tarn catchments (Tier 3b). The results for the tier 1 and tier 3b model runs have been compared with observed water quality data (see Table 6.14 and Figure 6.7). The results show that the tier 1 model is relatively robust as a national screening tool and the PIT model works particularly well when calibrated. Some of the coefficients for the models are highly dependent on the quality of empirical knowledge, so further (and ongoing) work must focus on this area to improve predictions where site-specific calibration is not readily possible (i.e. tiers 1-3).

Ongoing work under the DEFRA funded PEDAL project (PEO113) seeks to quantify the delivery and transfer of phosphorus to water bodies which in essence is the final element of the PIT approach that will allow predictions to be compared with in-lake water quality data (as with tier 1). Thus, a layer 4 is being developed to transfer phosphorus in stream and provide not only a spatial distribution of phosphorus predictions, but also an assessment of the absolute quantity of phosphorus distributed in a catchment that reaches the catchment outlet.

7 RISK ASSESSMENT

Laurence Carvalho and Mike Hughes

7.1 Introduction

As part of the review of the impact of human activity on the status of waters, Article 5 of the WFD requires Member States to carry out an assessment of the risk that water bodies will fail to meet the Directive's environmental objectives. This task involves both an analysis of "pressures" and "impacts" (IMPRESS, 2002; UK TAG, 2003). In terms of diffuse and point-source nutrient pressures, the risk of failing the WFD objective of good status should be based primarily on predicted impacts on the most sensitive quality elements to nutrient pressures. In this study, phytoplankton were selected as the most sensitive biological quality element, with in-lake nutrient concentrations as the most appropriate supporting physico-chemical quality element. Final guidance from REFCOND and ECOSTAT CIS working groups state that the supporting physico-chemical quality elements should also be of "good status" and overall status should be determined from the lowest of the status classes predicted, i.e. in this study either in-lake TP concentrations or phytoplankton composition and abundance (chlorophyll$_a$).

It was not, however, possible to develop ecological classification schemes sufficiently for either phytoplankton composition or abundance to be used in the risk assessment process. Furthermore, nationally-applicable models are only currently available for predicting impacts of catchment nutrient pressures on in-lake TP concentrations. For this reason it is recommended that the 2004 risk assessment round, predicted impacts of nutrient pressures be based solely on in-lake TP concentrations.

Following application of the risk assessment guidance outlined in this report, this chapter details the outcome of the Tier 1 risk assessment process for more than 14,000 GB lakes, according to ecotype and country. To assess the validity of these results a more detailed comparison is made for 50 test lakes between the Tier 1 modelled results and risk assessments based on observed water quality data. Issues in relation to boundary thresholds are also illustrated using particular case-studies.

7.2 Methods

Initially a Tier 1 risk assessment procedure was carried out on all GB lakes. This follows the guidance outlined in earlier chapters of this report and summarised in Appendix 1. In brief it included the following steps:

- Estimation of point-source nutrient pressures – STWs, septic tanks and package treatment works were considered together in the form of the catchment population (1991 census data) with a per capita export of 0.4 kg a^{-1} assumed. Fish farm data were not included at this stage as data

- were not available in the GB lakes database (and are currently only available for Scotland).
- Estimation of diffuse-source nutrient pressures – land-use (CEH land cover) and animal stocking data (including wild deer in Scotland)
- Estimated TP load summed from above and converted to in-lake TP concentration using OECD regression equations (OECD ,1982)
- Comparison of modelled in-lake TP concentration with ecotype-specific reference TP concentrations, providing a modelled EQR and risk/status class prediction

For the 50 test lakes, three further steps were carried out:
- Estimates of point-source nutrient pollution included fish farms (Loch Earn and Esthwaite Water only) and level of sewage treatment (Bassenthwaite lake, Windermere and Barton Broad)
- Modelled in-lake TP concentrations were compared with both ecotype-specific and site-specific reference TP concentrations
- Measured (observed) in-lake TP concentrations were compared with site-specific reference TP concentrations, providing a measured EQR and predicted risk/status class

A site was identified as “at risk” of failing good status (and if applicable Habitats Directive favourable condition) if it achieved an EQR below 0.5 (predicted moderate, poor or bad status).

Recent guidance (UK TAG, 2003) suggested three reporting categories for the risk assessment, although this has recently been modified to four reporting categories (Ingrid Baber, pers. comm.):
- 1a Water bodies at significant risk
- 1b Water bodies probably at significant risk
- 2a Water bodies not at significant risk for which confidence in the available information being comprehensive and reliable is low
- 2b Water bodies not at significant risk for which confidence in the available information being comprehensive and reliable is high

In terms of the tier 1 approach adopted here, the results are reported in terms of predicted status classes. Although a number of factors affect confidence in the risk assessment, the status classes could be provisionally interpreted in terms of the above UK TAG reporting classes in the following way:
- High status = 2b
- Good status = 2a
- Moderate status = 1b
- Poor or Bad status = 1a

7.3 Results of tier 1 approach on all GB lakes

Results of the Tier 1 risk assessment for all GB lakes are presented in a number of formats. Figure 7.1a-f illustrates the cumulative frequency and predicted status class of lakes along the modelled TP concentration gradient for six lake ecotypes. These plots can be used to examine how changing the TP status class boundaries will affect the number of GB sites considered at risk (moderate, poor or bad status). For example, using the current recommended threshold of 22 µg l^{-1} for the good/moderate boundary for very shallow low alkalinity lakes, about 75 % of sites are predicted as at risk (moderate, poor or bad status). Even if the good/moderate threshold was raised to a state representing major (not slight) enrichment at about 100 µg l^{-1}, about 40 % of sites would still be predicted as at risk.

Staying with the recommended nutrient classification outlined in Chapter 2, the predicted number of GB lakes of different status classes for these six lake ecotypes are given in Table 7.1 and illustrated in Figure 7.2. It is very evident from Figures 7.1 and 7.2 that "Very Shallow" lakes of all alkalinity classes appear to be at much greater risk than "Deeper" lakes, and that in general risk appears to increase with increasing alkalinity.

The distinctly different proportion of status classes (risk categories) between "Very Shallow" and "Deeper" lakes in GB is illustrated further in Figure 7.3; type-specific reference conditions are not currently available for marl, peat and brackish lake ecotypes, so all these sites are included as "unknown" status and considered to be at risk. A summary of percentage of known sites at risk is illustrated in Figure 7.4, which shows that overall 51 % of all GB lakes are predicted to not meet the TP criteria for high or good status, and must, therefore, be considered at risk, an additional number of sites of unknown status must also be considered at risk.

From the above analysis, it is clear that of all the six lake ecotypes examined, very shallow, medium alkalinity lakes are at greatest risk; not only do they have the highest percentage of sites at risk (92%), they are a relatively rare lake type (<1000 lakes) (Table 7.1). The results suggest that only 79 sites of this lake ecotype in the whole of GB are predicted to be not at risk (Table 7.1). More detailed investigation of these 79 sites should be considered a priority to confirm their status. This lake ecotype is likely to incorporate most lakes classified as "Mesotrophic" under the UK Biodiversity Action Plan, providing further support to the argument that this is a particularly threatened habitat, which requires targeted management and protection.

The results do raise a number of further issues:

- Are the results similar across GB countries/regions?
- Are they similar for 'important' lakes? (Importance being defined in Bennion *et al*. (2002) based on lake size, conservation status and legislative designations)
- Are the reference conditions for Very Shallow lakes particularly demanding?

The first of these questions was examined further using the data from the Tier 1 analysis. The latter two questions are discussed later in relation to the 50 test lakes, although importance could be examined further in terms of all GB lakes identified as "important" in Bennion *et al*. (2002) or relatively simply all large lakes (e.g. WFD threshold of >0.5 km^2 in area).

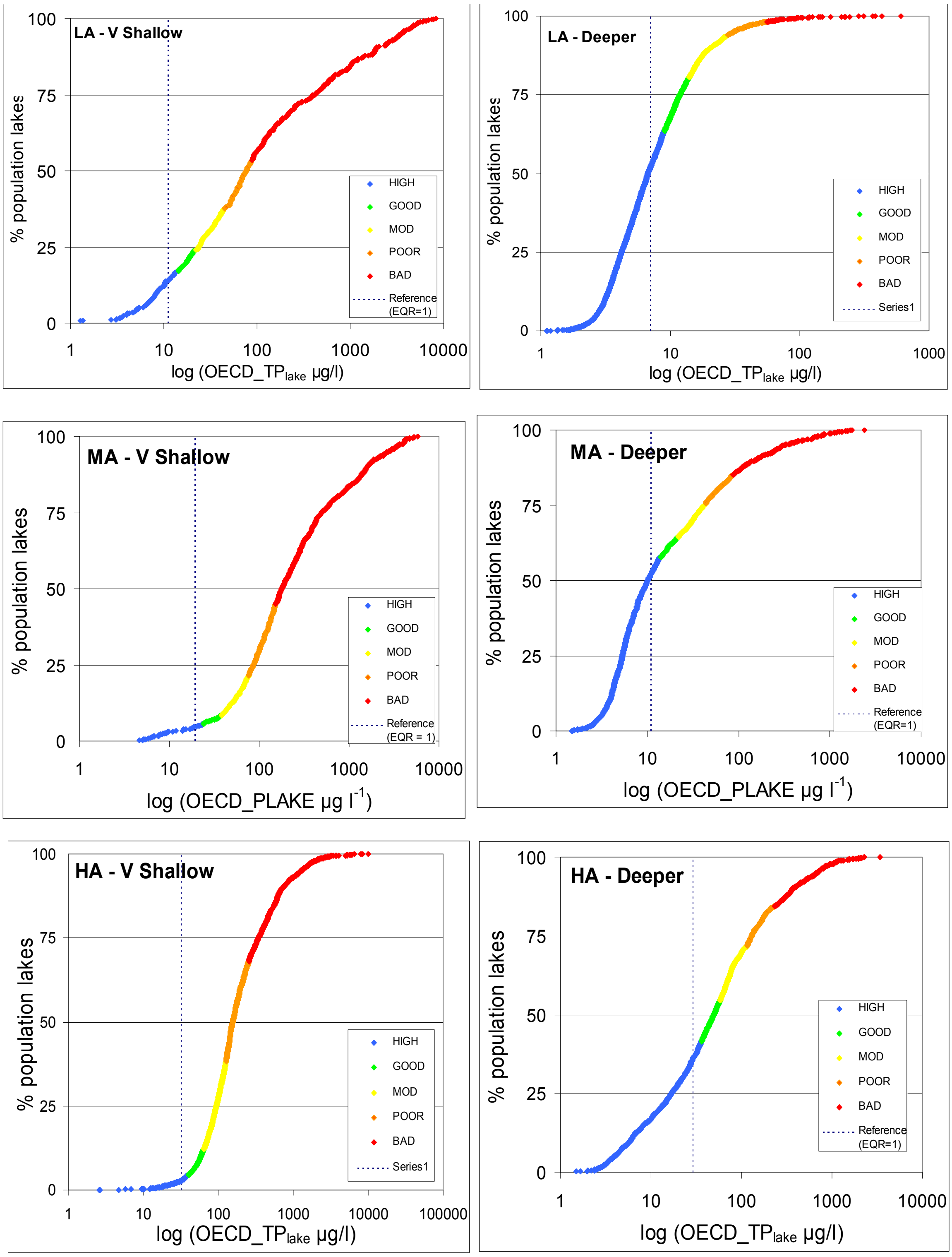


Figure 7.1 **Cumulative frequency and status class of lakes along a modelled TP concentration gradient for the following ecotypes: a) low alkalinity, very shallow, b) low alkalinity, deeper, c) medium alkalinity, very shallow, d) medium alkalinity, deeper, e) high alkalinity, very shallow and f) high alkalinity, deeper**

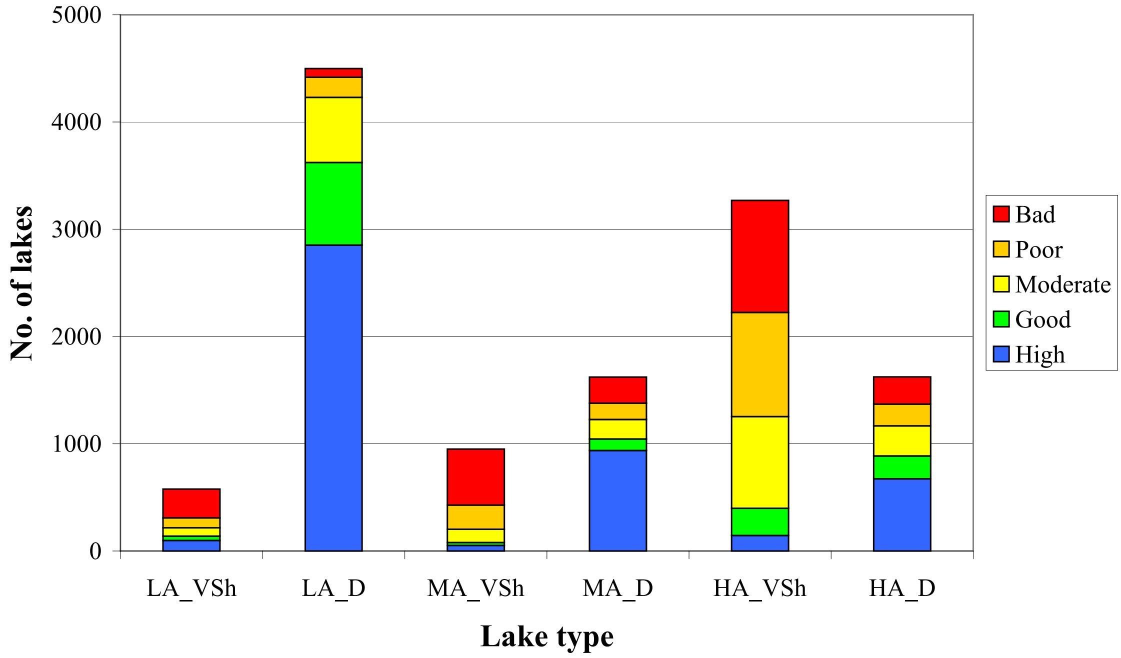


Figure 7.2 Predicted number of GB lakes of different status classes for six lake ecotypes

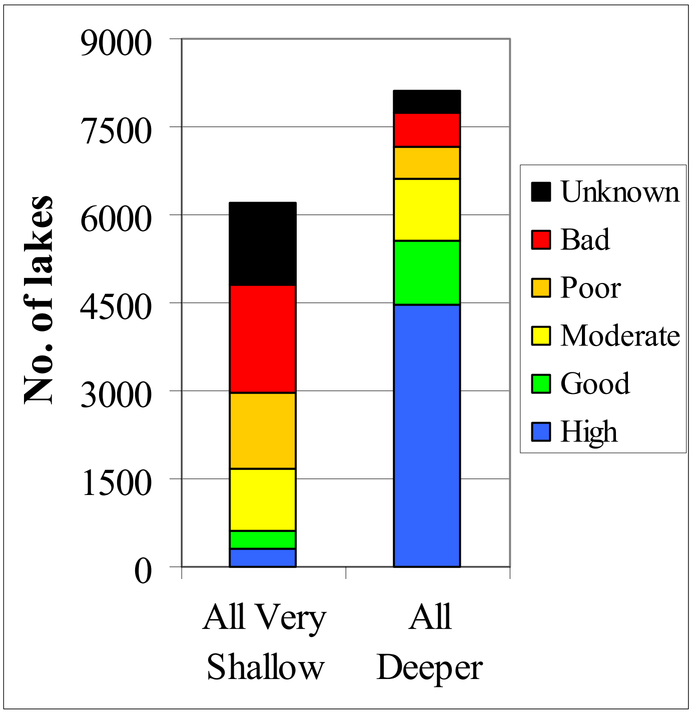


Figure 7.3 Predicted number of GB lakes of different status classes for all "Very Shallow" and all "Deeper" lake types

Note: All marl, peaty and brackish lakes are considered as unknown status

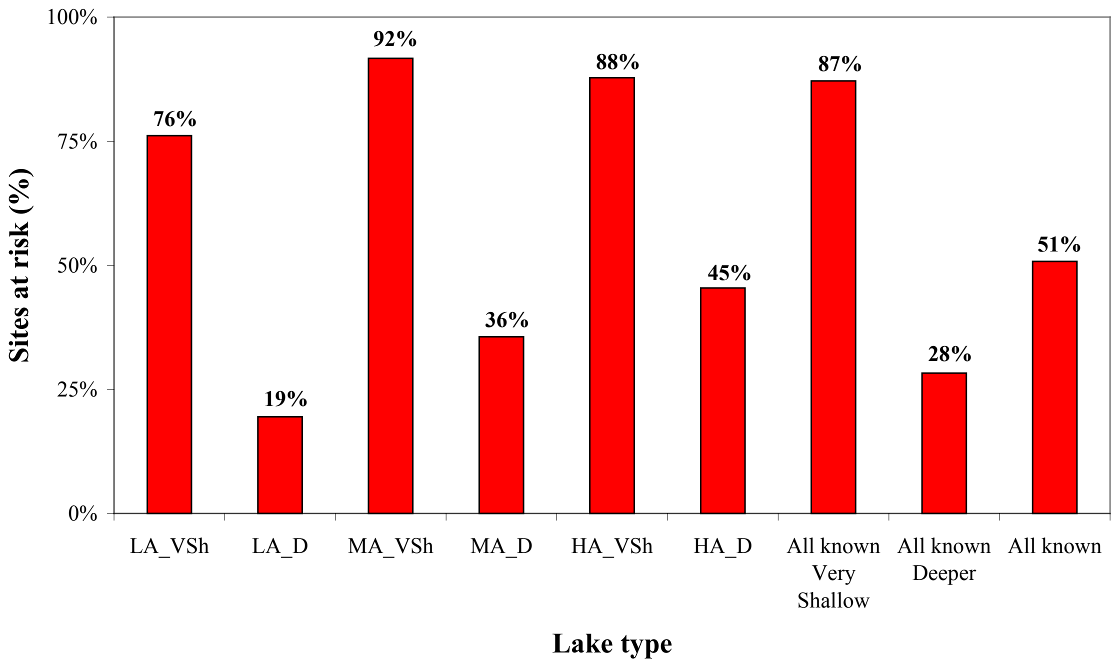


Figure 7.4 Percentage of sites considered "at risk" for six lake ecotypes, all very shallow, all deeper and all GB lakes

Note: All known very shallow, all known deeper and all known lakes exclude all marl, peaty and brackish sites of unknown status

Table 7.1 Predicted numbers of GB lakes by status class and consequent numbers at risk

ALL GB LAKES Ecotype	No. Lakes	High	Good	Moderate	Poor	Bad	Unknown	No. at risk	% at risk
LA_VSh	578	98	40	78	93	269	0	440	76%
LA_D	4498	2852	770	607	189	80	0	876	19%
MA_VSh	951	53	26	124	225	523	0	872	92%
MA_D	1621	936	108	182	153	242	0	577	36%
HA_VSh	3268	145	254	853	972	1044	0	2869	88%
HA_D	1623	674	212	281	203	253	0	737	45%
Marl_VSh	6						6	6	100%
Marl_D	78						78	78	100%
Peat_VSh	1383						1383	1383	100%
Peat_D	50						50	50	100%
Unclassified_VSh	35						35	35	100%
Unclassified_D	251						251	251	100%
Brackish_VSh	0						0	0	
Brackish_D	0						0	0	
All known Very Sh	4797	296	320	1055	1290	1836	0	4181	87%
All known Deeper	7742	4462	1090	1070	545	575	0	2190	28%
All known	12539	4758	1410	2125	1835	2411	0	6371	51%
All Very Shallow	6186	296	320	1055	1290	1836	1389	5570	90%
All Deeper	7870	4462	1090	1070	545	575	128	2318	29%
All GB Lakes	14056	4758	1410	2125	1835	2411	1517	7888	56%

Regional differences in risk

Figure 7.5 shows the geographical distribution of GB lakes by ecotype and predicted status class. From this figure it is very clear that there are major regional differences in numbers of sites at risk. Figures 7.6a-c and Tables 7.2-7.4 display this in quantitative terms and reveal that Scotland has by far the fewest sites at risk (18%), England by far the most (88%) with Wales having an intermediate percentage (56%). If sites of unknown status are also considered, Scotland shows a large increase in numbers of sites at risk (35%), whilst England (89%) and Wales (57%) increase only slightly. This increase in Scotland is due to the large number of peaty lochs present. There is a high probability that most of these sites are not at risk from nutrient pressures, being largely present in the more undisturbed parts of northern Scotland. Additionally, peaty waters may be less sensitive to eutrophication than non-peaty waters as light availability is likely to become more important in limiting phytoplankton populations. Submerged macrophytes that are already light-stressed will, however, be very sensitive to any increased phytoplankton or phytobenthos production. Further work is required not only to establish reference nutrient conditions for this lake type, so a nutrient classification can be developed, but also the ecological impact of nutrient pressures in peaty waters requires much further study. The same is true for marl lakes, which potentially are also less sensitive to nutrient pressures due to the phosphorus-binding capacity of calcium carbonate.

Figures 7.5 and 7.6 also highlight the large difference in numbers of sites between countries and the different proportions of lake ecotypes. Scotland contains the largest lake resource dominated by deeper, low alkalinity lakes with few very shallow lakes (particularly high alkalinity), whilst England, and to a lesser extent Wales, have higher numbers and the highest proportion of very shallow, high alkalinity lakes.

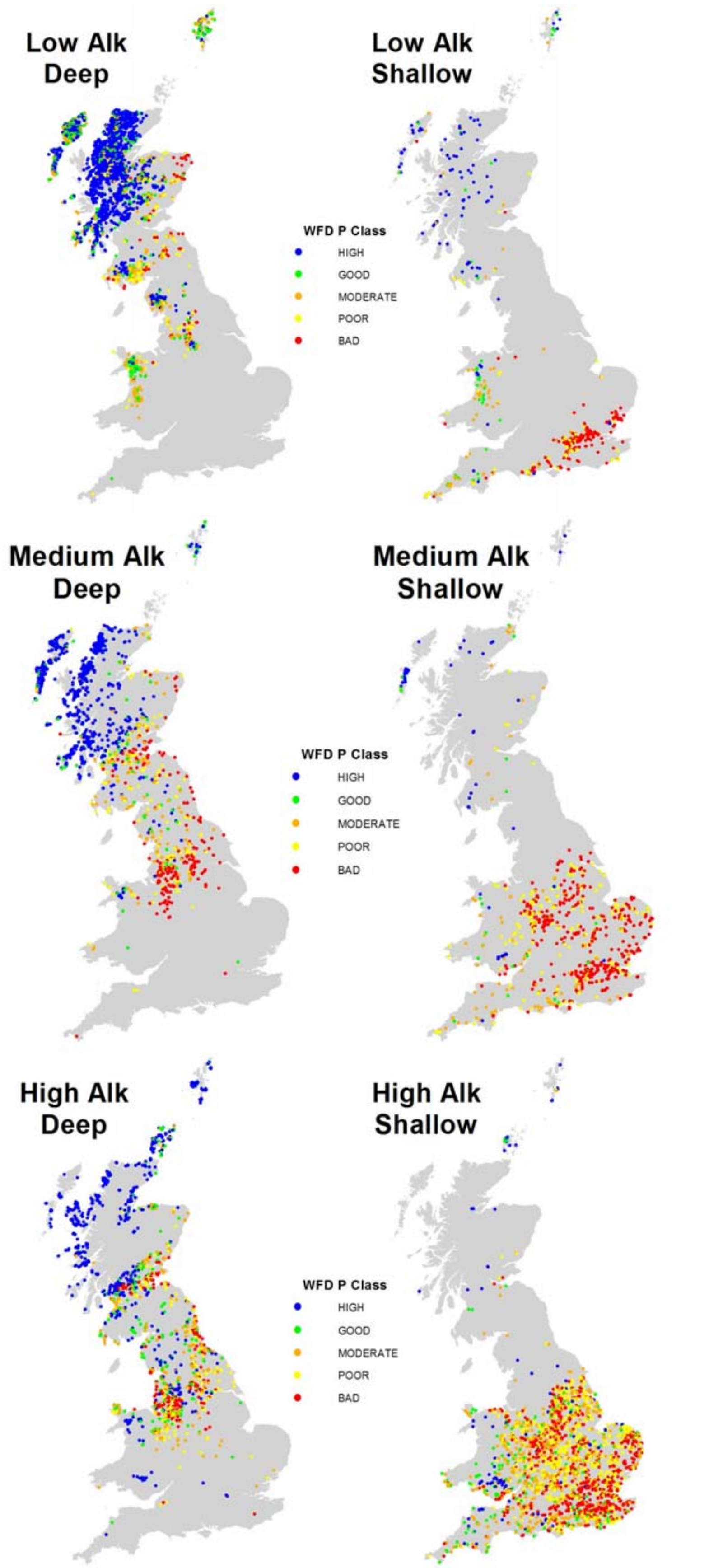


Figure 7.5 Distribution of GB lakes by ecotype and predicted status class

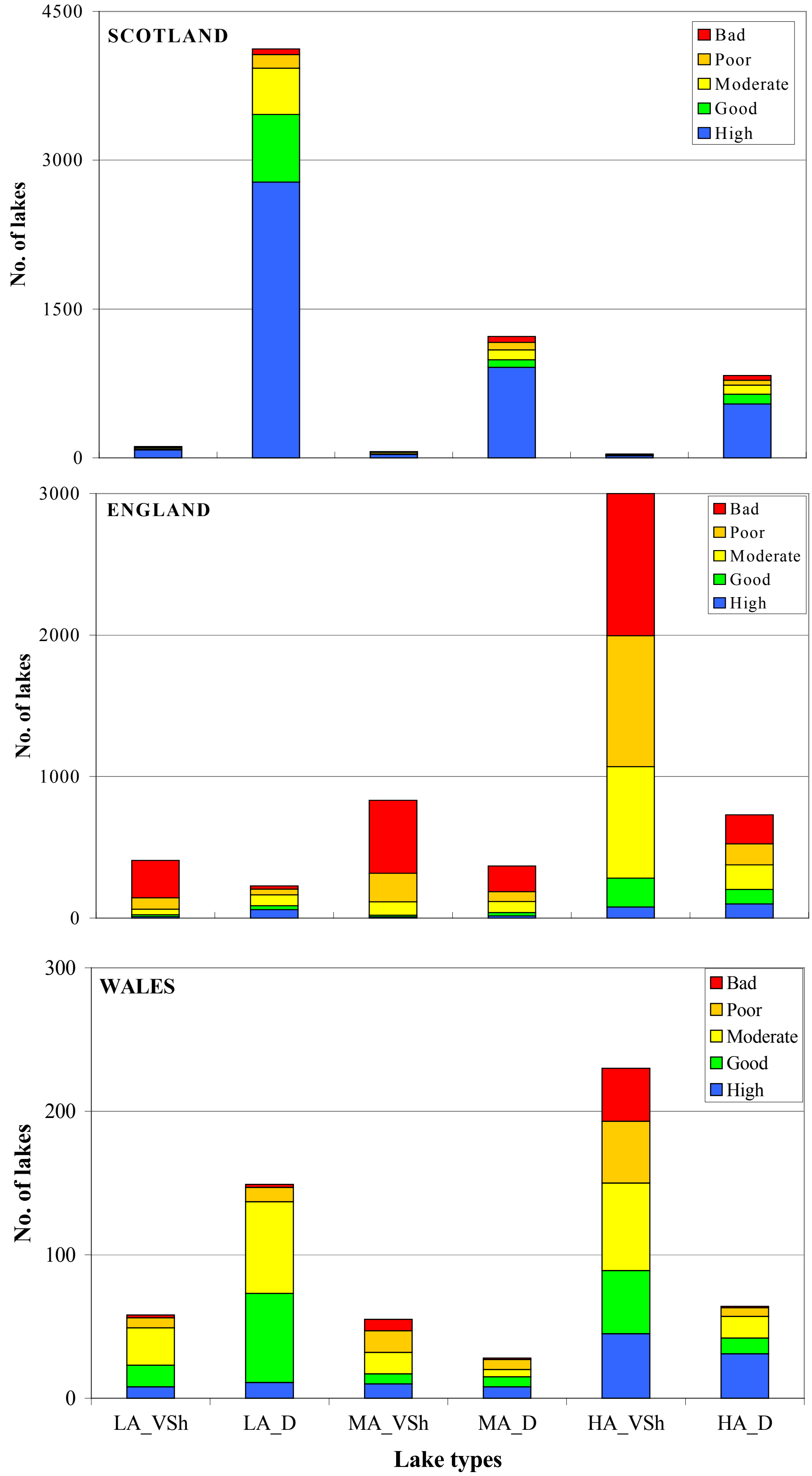


Figure 7.6 Predicted number of lakes of different status classes for six lake ecotypes for a) Scotland, b) England and c) Wales

Table 7.2 Predicted numbers of Scottish lochs by status class and consequent numbers at risk

SCOTLAND Ecotype	No. Lakes	High	Good	Moderate	Poor	Bad	Unknown	No. at risk	% at risk
LA_VSh	113	80	12	12	6	3	0	21	19%
LA_D	4122	2781	680	466	139	56	0	661	16%
MA_VSh	64	34	7	15	8	0	0	23	36%
MA_D	1224	913	77	99	76	59	0	234	19%
HA_VSh	38	21	6	5	4	2	0	11	29%
HA_D	830	543	99	92	49	47	0	188	23%
Marl_VSh	2						2	2	100%
Marl_D	41						41	41	100%
Peat_VSh	31						31	31	100%
Peat_D	1318						1318	1318	100%
Unclassified_VSh	8						8	8	100%
Unclassified_D	242						242	242	100%
Brackish_VSh	0						0	0	
Brackish_D	0						0	0	
Known_VSh	215	135	25	32	18	5	0	55	26%
Known_D	6176	4237	856	657	264	162	0	1083	18%
All Known	6391	4372	881	689	282	167	0	1138	18%
All Very Shallow	256	135	25	32	18	5	41	96	38%
All Deeper	7777	4237	856	657	264	162	1601	2684	35%
All Lakes	8033	4372	881	689	282	167	1642	2780	35%

Table 7.3 Predicted numbers of English lakes by status class and consequent numbers at risk

ENGLAND Ecotype	No. Lakes	High	Good	Moderate	Poor	Bad	Unknown	No. at risk	% at risk
LA_VSh	407	10	13	40	80	264	0	384	94%
LA_D	227	60	28	77	40	22	0	139	61%
MA_VSh	832	9	12	94	202	515	0	811	97%
MA_D	369	15	24	78	70	182	0	330	89%
HA_VSh	3000	79	204	787	925	1005	0	2717	91%
HA_D	729	100	102	174	148	205	0	527	72%
Marl_VSh	2						2	2	100%
Marl_D	36						36	36	100%
Peat_VSh	15						15	15	100%
Peat_D	63						63	63	100%
Unclassified_VSh	22						22	22	100%
Unclassified_D	8						8	8	100%
Brackish_VSh	0						0	0	
Brackish_D	0						0	0	
Known_VSh	4239	98	229	921	1207	1784	0	3912	92%
Known_D	1325	175	154	329	258	409	0	996	75%
All Known	5564	273	383	1250	1465	2193	0	4908	88%
All Very Shallow	4278	98	229	921	1207	1784	39	3951	92%
All Deeper	1432	175	154	329	258	409	107	1103	77%
All Lakes	5710	273	383	1250	1465	2193	146	5054	89%

Table 7.4 Predicted numbers of Welsh lakes by status class and consequent numbers at risk

WALES Ecotype	No. Lakes	High	Good	Moderate	Poor	Bad	Unknown	No. at risk	% at risk
LA_VSh	58	8	15	26	7	2	0	35	60%
LA_D	149	11	62	64	10	2	0	76	51%
MA_VSh	55	10	7	15	15	8	0	38	69%
MA_D	28	8	7	5	7	1	0	13	46%
HA_VSh	230	45	44	61	43	37	0	141	61%
HA_D	64	31	11	15	6	1	0	22	34%
Marl_VSh	1						1	1	100%
Marl_D	1						1	1	100%
Peat_VSh	4						4	4	100%
Peat_D	2						2	2	100%
Unclassified_VSh	5						5	5	100%
Unclassified_D	1						1	1	100%
Brackish_VSh	0						0	0	
Brackish_D	0						0	0	
Known_VSh	343	63	66	102	65	47	0	214	62%
Known_D	241	50	80	84	23	4	0	111	46%
All Known	584	113	146	186	88	51	0	325	56%
All Very Shallow	353	63	66	102	65	47	10	224	63%
All Deeper	245	50	80	84	23	4	4	115	47%
All Lakes	598	113	146	186	88	51	14	339	57%

7.4 Results of enhanced tier 1 approach on 50 test lakes

In order to assess the validity of the GB-wide results, a number of well-studied test lakes were selected to examine whether results were in agreement with observed data and expert opinion on the sites.

7.4.1 Selection of Test Lakes

Lakes were initially selected from all available sources if they had measured depth, alkalinity, TP and chlorophyll data. Data were primarily sourced from the GB lakes database, SEPAs Scottish loch monitoring programme and CEH datasets. These were reduced to a selection of 134 lakes (Appendix 2), which had a minimum of four regularly-spaced observations of TP and chlorophyll$_a$ concentrations, which passed all quality assurance checks. These were then sorted into lake types according to the latest guidance from the Lakes Task Team (Phillips, 2003a).

Summarising the mean chemistry data for these 134 lakes illustrates the differences between lake types (Figures 7.7 and 7.8). In terms of lake geology types, alkalinity is highest in brackish (1 site), calcareous (HA) and calcareous-marl (1 site) sites and, as expected, medium alkalinity siliceous sites (Si_MA) have higher alkalinities than low alkalinity siliceous (Si_LA) sites (Figure 7.7a). Sites with peat-dominated catchments (4 sites) appear to have a surprisingly high alkalinity.

Calcareous sites have on average much higher, although variable, concentrations of TP, whereas the single marl site (Malham Tarn) has very low TP concentrations (Figure 7.7b). Presumably this reflects the importance of P-precipitation processes in marl lakes, supporting the decision to separate marl sites out as a distinct lake type in terms of sensitivity to nutrient pressures. Data from other marl sites are, however, required to examine whether this observation at a single site is typical.

Phytoplankton chlorophyll$_a$ is high in the one brackish site, and higher in the calcareous sites than the remaining lake types, although the single marl site is again relatively low compared with the other calcareous sites (Figure 7.7c). Chlorophyll$_a$ concentrations are generally higher in medium alkalinity siliceous sites than low alkalinity siliceous sites.

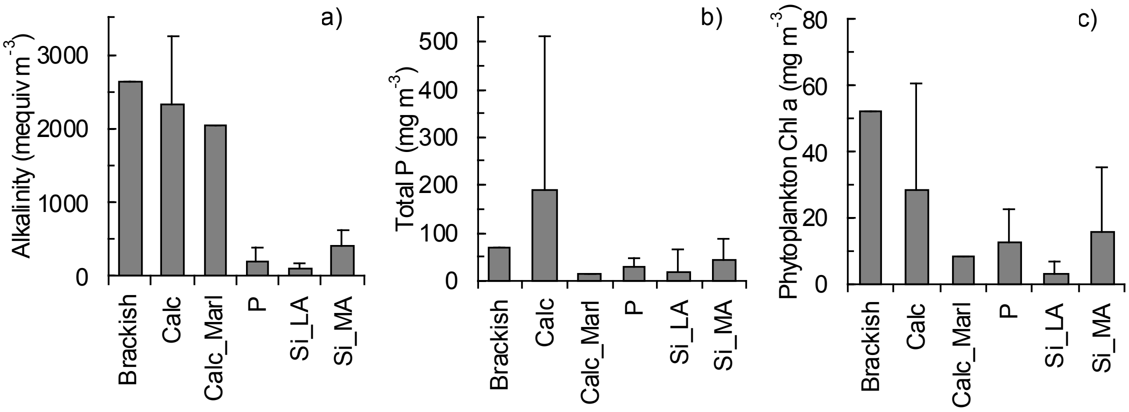


Figure 7.7 Mean water chemistry for the different geological lake classes: a) alkalinity, b) total phosphorus and c) phytoplankton chlorophyll$_a$

Note: The error bars represent one standard deviation

Differences in water chemistry are even more distinct across the different depth classes (Figure 7.8). In general, deep lakes have lower alkalinity, TP and phytoplankton chlorophyll$_a$ than very shallow lakes; shallow lakes are intermediate.

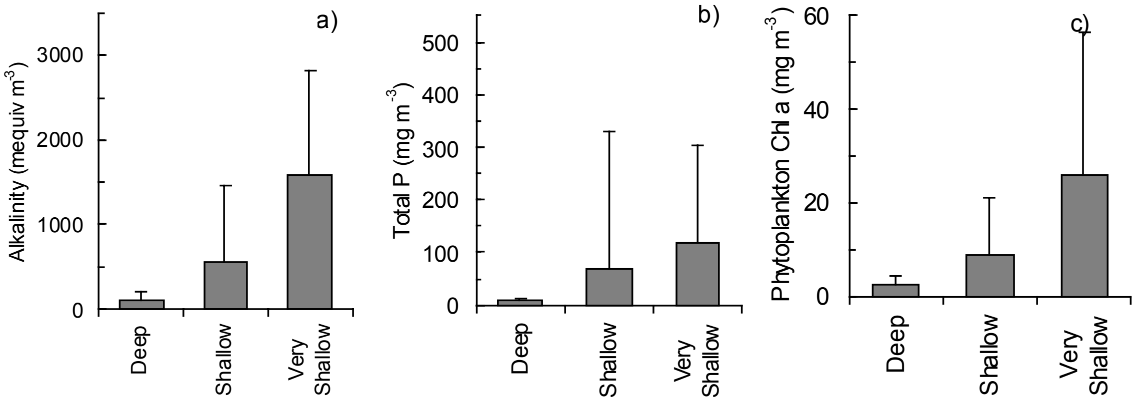


Figure 7.8 Mean water chemistry for the different depth categories: a) alkalinity, b) total phosphorus and c) phytoplankton chlorophyll$_a$

Note: the error bars represent one standard deviation.

A selection of 50 test lakes was then chosen, from the list of 134 lakes, to initially trial the risk assessment process (Table 7.5). These 50 lakes were chosen as equally as possible across all lake types with the most well studied sites qualifying first as there is greater confidence in their water chemistry data (more frequent sampling) and a better understanding exists for discussion of the risk assessment results.

Table 7.5 Selected 50 lakes for testing risk assessment

WBID	OSNAME	Mean Depth (m)	Depth Type	Geology Type	Alkalinity (μequiv.l^{-1})
Peat - Very shallow					
38623	Maes-Llyn	2.7	VSh	P	527
41210	Llyn Fach	1.7	VSh	P	99
Peat - Deeper					
704	Eela Water	5.1	Sh	P	106
1271	Loch of Girlsta	9.6	Sh	P	106
Low Alkalinity - Very Shallow					
33474	Oak Mere	1.6	VSh	Si_LA	30
37080	Llyn Glanmerin	1.6	VSh	Si_LA	97
37437	Bugeilyn	1.9	VSh	Si_LA	7
38394	Llyn Hîr	2.8	VSh	Si_LA	14
38422	Llyn Eiddwen	2.6	VSh	Si_LA	89
38525	Llyn Gynon	2.2	VSh	Si_LA	13
43651	Albury Mill	0.5	VSh	Si_LA	119
Low Alkalinity - Deeper					
14057	Loch Maree	38.2	D	Si_LA	50
18767	Loch Ness*	132.0	D	Si_MA	61
28965	Derwent Water	5.5	Sh	Si_LA	96
28986	Loweswater	8.4	Sh	Si_LA	195
29183	Wast Water	40.2	D	Si_LA	58
29184	Grasmere	7.7	Sh	Si_LA	157
34987	Llyn Tegid or Bala Lake	24.0	D	Si_LA	134
Medium Alkalinity - Very Shallow					
21123	Loch Davan	1.2	VSh	Si_MA	516
21189	Loch Kinord	1.5	VSh	Si_MA	228
39796	Upper Talley Lake	1.9	VSh	Si_MA	448
39813	Lower Talley Lake	1.9	VSh	Si_MA	343
40571	Llyn Llech Owen	1.2	VSh	Si_MA	-8
43218	Bolder Mere	0.7	VSh	Si_MA	860
44635	Cinder Hill	1.2	VSh	Si_MA	690
Medium Alkalinity - Deeper					
24132	Loch Earn	42.0	D	Si_MA	342
24447	Loch Lomond (S. Basin)*	37.0	D	Si_LA	230
28847	Bassenthwaite Lake	5.3	Sh	Si_MA	180
28955	Ullswater	25.3	D	Si_MA	236
29233	Windermere (S. Basin)	16.8	D	Si_MA	269
29270	Blelham Tarn	6.8	Sh	Si_MA	470
29321	Coniston Water	24.1	D	Si_MA	200
29328	Esthwaite Water	6.4	Sh	Si_MA	425
High Alkalinity - Very Shallow					
1694	Loch of Boardhouse	1.8	VSh	Calc	1814
32744	The Mere (Mere Mere)	2.8	VSh	Calc	1510
32948	Llyn Dinam	1.4	VSh	Calc	1533
33337	Llyn Coron	1.8	VSh	Calc	1869
35655	Barton Broad	1.5	VSh	Calc	3385
36202	Upton Broad	0.8	VSh	Calc	2701
40067	Llangorse Lake	2.0	VSh	Calc	2446
50001	Slapton Ley	1.6	VSh	Calc	2018
High Alkalinity - Deeper					
24843	Loch Leven	4.5	Sh	Calc	1407
32650	Rostherne Mere	13.6	Sh	Calc	2208
32761	Llyn yr Wyth-Eidion	6.0	Sh	Calc	3997
32804	Tatton Mere	4.3	Sh	Calc	2600
35091	White Mere	5.4	Sh	Calc	1880
39267	Llan Bwch-llyn Lake	3.0	Sh	Calc	1391
40608	Marsworth Reservoir	3.0	Sh	Calc	4200
High Alkalinity (Marl) - Shallow					
29844	Malham Tarn	2.6	VSh	Calc_Marl	2040
Brackish - Shallow					
35640	Hickling Broad	1.3	VSh	Brackish	2640

*Measured alkalinity over-ridding geology type

7.4.2 Additional point-source considerations at test lakes

Currently the tier 1 risk assessment process based on the GB lakes inventory is not set-up to deal with site-specific issues in the estimation of point-source pressures to a water body. Two areas that it was felt were worth considering were sites where the population is served by a STW with tertiary treatment and sites with a resident cage fish farm.

Of the 50 test lakes, data were available on three sites (Barton Broad, Bassenthwaite and Windermere) which have had tertiary treatment installed at one of the STWs in their catchment and two sites (Esthwaite Water and Loch Earn) which have an established cage fish farm. The revised loadings for these five sites were estimated following the guidance outlined in Chapter 5 (Table 7.6). Comparisons of the revised modelled-TP concentrations were made with the original modelled-TP concentrations and measured data (Table 7.7).

The addition of the fish farm data appears to give a much closer estimate of measured TP. It is, therefore, strongly recommended that the current cage fish farm dataset available for Scotland is added to the GB lakes database and similar data is gathered for England and Wales.

In terms of tertiary treatment, the revised approach involved applying a further 80% P removal coefficient to the proportion of the catchment population served by the upgraded STW. This varied from 49% of the population served by Keswick STW in the Windermere catchment to 72% of the population served by Stalham STW in the Barton Broad catchment (Table 7.6). A large improvement is apparent for Barton Broad (Table 7.7), although the tier 1 model still greatly over-estimates, compared to measured TP, presumably because a sizeable proportion of the catchment population are served by South Walsham STW, which discharges to sea. The revised modelled data for Windermere and Bassenthwaite Lake appear to result in a slight under-estimate when compared with measured TP (Table 7.7). A plausible explanation for this is the non-resident tourist population, which increase TP loads greatly during certain times of the year, as recognised in the design capacity of the STWs in their catchments (see Table 5.4), but are generally not included in the load estimates below based on census data taken in early summer before the main holiday season.

Table 7.6 Revised loading figures (kg a^{-1}) for five test lakes

WBID	NAME	% tertiary treatment	Land Cover	Animals	Population	Fish Farm	Total TP Load
24132	Loch Earn		365	1358	153	3600	5476
29328	Esthwaite Water		363	364	209	800	1735
28847	Bassenthwaite Lake	59%	3797	8013	1751		13561
29233	Windermere	49%	2795	3872	3507		10173
35655	Barton Broad	72%	5826	1663	2587		10076

Table 7.7 Comparison of modelled TP (original) with Modelled TP (revised) and measured TP concentrations (µg l^{-1})

WBID	NAME	Modelled TP_original	Modelled TP_revised	Measured TP
24132	Loch Earn	5	11	13
29328	Esthwaite Water	21	35	30
28847	Bassenthwaite Lake	17	16	21
29233	Windermere	15	12	14
35655	Barton Broad	274	214	92

7.4.3 Results of 50 test lakes

Results of the risk assessment for the 50 test lakes (using revised load estimates) are presented in Table 7.8 and Figure 7.9. Three approaches were considered:

- EQR based on measured TP concentrations and site-specific reference conditions
- EQR based on modelled TP concentrations and site-specific reference conditions
- EQR based on modelled TP concentrations and type-specific reference conditions

The modelled data using site-specific reference conditions appears to give a very good fit with the measured TP data – with the same number of sites (but not exactly the same sites) in high, good and moderate status classes and 54% of sites predicted as at risk of failing good status (Table 7.8 and Figure 7.9). The EQR based on modelled TP concentrations and type-specific reference conditions predicted only 46% of sites at risk, suggesting that if anything this approach will misclassify many more sites as not at risk, when in fact they are. The same approach applied to all GB lakes with known type-specific reference conditions predicted 51% of sites at risk (Table 7.1), which is a slightly higher incidence than in the 50 test lakes – this could, however, reflect the predominance of well-studied ‘important’ (Bennion *et al.*, 2002) and mainly rural sites amongst the test lakes.

The biggest discrepancies between the approaches were seen at the two sites with very low reference TP concentrations (Wast Water and Loch Maree), where only slight changes in TP concentration (within precision of many analytical laboratories) will lead to large changes in EQR (Table 7.8). The classification of Loch Maree as poor status based on observed/measured data, could be considered precautionary, as small nutrient changes at these sites could have significant ecological impacts, although equally it could be misclassification of its true status, simply due to a lack of precision in analysis of current chemistry. This may support the argument of having a threshold TP concentration, such as 5 or 10 µg l^{-1}, below which sites are considered at least as good status and not at risk, irrespective of their very low reference TP concentration. Alternatively, it may be more precautionary to simply highlight these sites for particular attention. Another large discrepancy was Slapton Ley; the reason for this is unclear.

Most other discrepancies were of the order of only one status class, although it must be noted that there are several sites which are predicted as having good status using the modelled approaches, but were of moderate status based on measured TP data (e.g. Loch Leven, Grasmere, Upper Talley lake, Blelham Tarn, Llangorse Lake). This highlights the need for these sites to be categorised as “not at risk – low confidence” requiring further detailed investigation (risk assessment or operational monitoring). The opposite where modelled status is lower than measured status is of less concern for a precautionary risk assessment process.

Table 7.8 Results of risk assessment based on three approaches (i) measured TP concentrations and site-specific reference conditions, (ii) modelled TP concentrations and site-specific reference conditions and (iii) modelled TP concentrations and type-specific reference conditions

		EQR / Status		
WBID	**OSNAME**	**Measured - site-specific reference**	**Modelled - site-specific reference**	**Modelled - type-specific reference**
Peat - Very Shallow				
38623	Maes-Llyn	0.342	0.563	0.469
41210	Llyn Fach	1.263	1.091	1.364
Peat - Deeper				
704	Eela Water	0.216	1.000	1.000
1271	Loch of Girlsta	0.368	1.167	1.333
Low Alkalinity - Very Shallow				
33474	Oak Mere	0.108	0.062	0.097
37080	Llyn Glanmerin	0.816	0.364	0.333
37437	Bugeilyn	0.278	0.238	0.524
38394	Llyn Hîr	0.735	0.357	0.786
38422	Llyn Eiddwen	0.732	0.536	0.393
38525	Llyn Gynon	0.779	0.375	0.688
43651	Albury Mill	0.057	0.033	0.019
Low Alkalinity - Deeper				
14057	Loch Maree	0.333	1.000	2.333
18767	Loch Ness	1.000	1.500	1.750
28965	Derwent Water	0.801	0.750	0.875
28986	Loweswater	0.547	0.474	0.368
29183	Wast Water	0.926	0.200	1.400
29184	Grasmere	0.340	0.571	0.500
34987	Llyn Tegid or Bala Lake	0.571	0.444	0.389
Medium Alkalinity - Very Shallow				
21123	Loch Davan	0.485	0.400	0.475
21189	Loch Kinord	0.643	1.286	1.357
39796	Upper Talley Lake	0.374	0.576	0.576
39813	Lower Talley Lake	0.248	0.405	0.452
40571	Llyn Llech Owen	0.396	0.792	0.792
43218	Bolder Mere	0.753	0.170	0.101
44635	Cinder Hill	0.130	0.049	0.038
Medium Alkalinity - Deeper				
24132	Loch Earn	0.385	0.455	1.000
24447	Loch Lomond (S. Basin)	0.556	0.714	1.571
28847	Bassenthwaite Lake	0.719	0.938	0.688
28955	Ullswater	0.601	0.750	1.375
29233	Windermere (S. Basin)	0.648	0.750	0.917
29270	Blelham Tarn	0.410	0.542	0.458
29321	Coniston Water	0.801	0.600	1.100
29328	Esthwaite Water	0.463	0.400	0.314
High Alkalinity - Very Shallow				
1694	Loch of Boardhouse	0.677	0.875	1.333
32744	The Mere	0.463	0.298	0.381
32948	Llyn Dinam	0.429	0.366	0.244
33337	Llyn Coron	0.320	0.439	0.281
35655	Barton Broad	0.434	0.187	0.150
36202	Upton Broad	1.454	0.292	0.208
40067	Llangorse Lake	0.272	0.500	0.500
50001	Slapton Ley	0.144	0.550	0.533
High Alkalinity - Deeper				
24843	Loch Leven	0.492	0.558	0.558
32650	Rostherne Mere	0.059	0.213	0.363
32761	Llyn yr Wyth-Eidion	1.024	0.258	0.439
32804	Tatton Mere	0.099	0.069	0.077
35091	White Mere	0.014	0.477	0.659
39267	Llan Bwch-llyn Lake	0.646	0.411	0.518
40608	Marsworth Reservoir	0.072	0.084	0.071
High Alkalinity (Marl) - Very Shallow				
29844	Malham Tarn	0.544	0.500	0.500
Brackish - Very Shallow				
35640	Hickling Broad	0.540	0.314	0.314

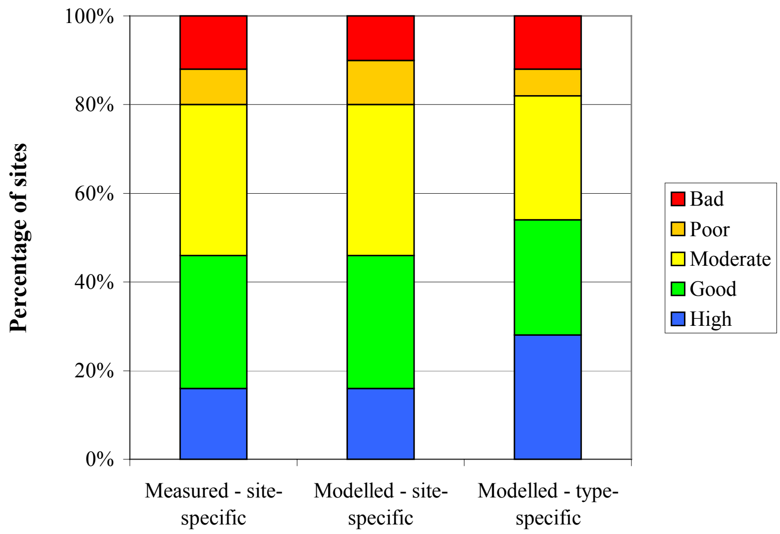


Figure 7.9 Summary results of risk assessment based on three approaches (i) measured TP concentrations and site-specific reference conditions, (ii) modelled TP concentrations and site-specific reference conditions and (iii) modelled TP concentrations and type-specific reference conditions

Regional differences were again clear amongst the 50 test lakes, with many of the high status sites being Scottish or from the Cumbrian lake district and many of the poor or bad status sites being sites located in south-east England or West Midland meres (Table 7.8). These were generally in agreement with expert opinion on these sites.

7.5 Summary and Recommendations

The results from the tier 1 risk assessment suggest a relatively high proportion of sites will be at risk of failing to achieve good status in terms of nutrient pressures (about 50% in GB as a whole but up to almost 90% in England), potentially leading to a very costly programme of measures. It could be argued that the nutrient classification is too stringent, either with too low reference conditions, or with a good/moderate boundary reflecting less than a "slight" deviation from undisturbed.

In terms of reference conditions it is difficult to justify different levels than those set in Chapter 2. These were based on all available data using three independent models (MEI, DI-TP, PLUS). It could be argued that the MEI approach requires calibration for GB lakes before being applied, although this is unlikely to change reference conditions dramatically. It may also be true that for some lake ecotypes too little data are available to estimate a representative reference condition. Similarly looking at the range of data it is unlikely to alter reference conditions dramatically.

In terms of the good/moderate boundary, it is difficult to consider a doubling of reference TP concentrations as being less than “minor” or “slight” change, for the boundary to be set at even higher levels than this. The cumulative frequency plots (Figure 7.1) highlight the fact that the good/moderate boundary would need to be shifted significantly to deliver a large decrease in the proportion of sites at risk.

Further justification for the current nutrient classification is provided through the analysis of 50 well-studied test lakes. Expert opinion on the project team (including Agency representatives), was generally in agreement with the predicted status classes for the 50 test lakes. It appeared to identify sites generally considered of high (Loch Ness) or good (Loch Lomond) status, sites considered to be around the good/moderate boundary (Loch Leven, Loweswater, Malham Tarn), sites at risk for which, there exists some concern (e.g. Esthwaite Water and Loch Earn) and sites clearly of poor or bad status (Rostherne Mere and Marsworth Reservoir).

There were clearly specific cases where other factors may be important in terms of susceptibility to nutrient pressures (e.g. White Mere), but no nationally applicable screening tool is likely to be able to pick these up. The procedure outlined is a basic “tier 1” risk assessment, although the slightly more refined risk assessment using site-specific reference conditions, rather than type-specific is highly recommended. The site-specific approach does, however, require data (preferably measured) on lake depth and alkalinity to apply the MEI approach to identifying reference conditions. The fact that some sites may slip through and be predicted as not at risk, when in fact they are, supports the use of the criteria outlined in Bennion *et al*. (2002), which recommended all “important” sites are automatically selected for further investigation.

What is important to stress is that the reference conditions developed and current conditions used for comparison are based on very limited data. Further data collection is necessary to produce more confidence in the results, particularly for peaty, marl and brackish lakes. Further work is also required to assess the confidence and precision in the classification scheme with increasing monitoring effort, as outlined in recent ECOSTAT guidance (ECOSTAT, 2003). The risk of misclassification is lower if the EQR is nearer the middle of the class than the class boundary. For this reason, sites near the good/moderate class boundary should, in particular, be selected for enhanced monitoring (ECOSTAT, 2003). This is also in agreement with recent UK guidance (UK TAG, 2003)

In summary it does, therefore, appear that a large number of sites in GB are at risk from nutrient pressures, particularly in England where on average about 90 % of sites are at risk. The risk assessment process outlined above is, however, just the first step in a tiered process in identifying sites at risk. Those identified require further investigation, both monitoring and modelling using the more sophisticated approaches outlined in earlier chapters. It may be that these more detailed studies do not alter this picture dramatically and that the results reflect a true picture of the impacts of nutrient pressures in the generally more populated and intensively farmed landscape of England compared with much of Scotland and North Wales.

The 2004 risk assessment is, however, just the first stage in delivering improved management of water resources through the WFD. Risk assessment is just a component of risk management (Adams, 1997). Later stages in the WFD deal with how we manage these risks, estimating the benefits associated with 'derogating' against these risks or, if we seriously aim to achieve good status in all waters, how we balance our 'behaviour' to minimise the risks. More sophisticated site-specific approaches to both the assessment of pressures (e.g. PIT, septic tanks, fish farms etc.) and assessment of impact (metabolic models, intensive monitoring) will ultimately be necessary.

8 BIBLIOGRAPHY

Adams J, 1997 *Cars, cholera and cows: virtual risk and the management of uncertainty*. Science Progress, **80**, 253-272.

Bailey-Watts A E, Gunn I D M, Kirika A and Lyle A A, 1993 *Threats to the oligotrophic status of Loch Shiel*. Report to Scottish Natural Heritage, 35 pp., 11 Figures

Bennion H, Fluin J, Appleby P and Ferrier B, 2001 *Palaeolimnological investigation of Scottish Freshwater Lochs*. SNIFFER Report No. SR(00)02F

Bennion H, Hilton J, Hughes M, Clark J, Hornby D, Kernan M and Simpson G, 2002 *Development of a risk based prioritisation protocol for standing waters in Great Britain based on a georeferenced inventory – Phase 2*. Environment Agency R&D Technical Report P2-260/2/TR1, Bristol.

Beven K and Binley A, 1992 '*The future of distributed models: Model calibration and uncertainty prediction',* Hydrological Processes, **6**, 279-298

Beyruth, Z, 2000 *Periodic disturbances, trophic gradient and phytoplankton characteristics related to Cyanobacterial growth in Guarapiranga Reservoir, Sao Paulo State, Brazil.* Hydrobiologia, **424**, 51–65

Borics G, Grigorszky I, Szabó S and Padisák J, 2000 *Phytoplankton associations in a small hypertrophic fishpond in east Hungary during a change from bottom-up to top-down control.* Hydrobiologia, **424**, 79–90

Braun-Blanquet, J, 1964 *Pflanzensociologie.* Springer, Wien.

Brett S J, Guy G K, Morse and Lester J N, 1997 *Phosphorus removal and recovery technologies*. Selper Publications on behalf of CEEP.

Carvalho L, Bennion H, Dawson H, Furse M, Gunn I, Hughes R, Johnston A, Luckes S, Maitland P, May L, Monteith D, Taylor R, Trimmer M and Winder J, 2002 *Nutrient conditions for different levels of ecological status and biological quality in surface waters (Phase 1)*. Environment Agency R&D Technical Report P2-260/4/TR, Bristol.

Carvalho L, Beklioglu M and Moss B, 1995 *Changes in a deep lake following sewage diversion – a challenge to the orthodoxy of external phosphorus control as a restoration strategy?* Freshwater Biology, **34**, 399-410

Cassell E A, Kort R L, Meals D W, Aschmann S G, Dorioz J M and Anderson D P, 2001 *Dynamic phosphorus mass balance modelling of large watersheds: long-term implications of management strategies.* Water Science and Technology, **4,** 153-162

Chambers P A and Kalff J, 1985 *Depth distribution and biomass of submersed aquatic macrophyte communities in relation to Secchi depth*. Canadian Journal of Fisheries and Aquatic Sciences **42**: 701-709

Cooper P, Upton J E, Smith M and Churchley J, 1995 *Biological nutrient removal: design snags, operational problems and costs.* J.IWEM, **9**, 7-18

Council of the European Communities, 2000 *Council Directive of 23 October 2000 establishing a framework for Community action in the field of water policy (2000/60/EC).* Official Journal of the European Communities, **L 327/1**

Deevey D G and Harkness N, 1973 *The significance of man-made sources of phosphorus: detergents and sewage.* Water Research, **7**, 35-54

Department of Environment Food and Rural Affairs (DEFRA), 2002 *The Government's Strategic Review of diffuse water pollution from agriculture in England. Agriculture and Water: A Diffuse Pollution Review*. Department for Environment, Food and Rural Affairs, June 2002, London

Duarte C M and Kalff J, 1987 *Latitudinal influences on the depth of maximum colonization and maximum biomass of submerged angiosperms in lakes.* Canadian Journal of Fisheries and Aquatic Sciences **44**: 1759-1764

Ebers T and Bischofsberger W, 1987 *Contamination of groundwater by septic tank percolation systems*. Water Science and Technology, **19**, 1275-1279

ECOSTAT, 2003 *Technical guidance on achieving adequate confidence and precision in classification*, CIS ECOSTAT Working Group Paper 03 Technical Annex, Version 3, 12 September 2003.

Ellis B and Childs K E, 1973 *Nutrient movements from septic tanks and lawn fertilization*. Technical Bulletin 73-75, Michigan Department of Natural Lansing, M1

Environment Agency, 2002 *The Water Framework Directive – Guiding principles on the technical requirements*. Environment Agency, Bristol.

Environment Agency, 2003. *Erosion risk of catchments in England and Wales*. Environmental Policy Risk and Forecasting Report Number 45

Fabbro L D and Duivenvoorden L J, 2000 *A two-part model liking multidimensional environmental gradients and seasonal succession of phytoplankton assemblages*. Hydrobiologia, **438**, 13–24

Ferrier R C, Malcolm A, McAlister E, Edwards T, Morrice J and Owen R, 1996 *Hindcasting of in-loch phosphorus concentrations based on land cover classification*. Report for the Scotland and Northern Ireland Forum for Environmental Research

Foy R H and Lennox S D, 2000 *Contributions of diffuse and point sources to the phosphorus loads in the River Main over a 22-year period.* Boreal Environment Research, **5**, 27-37

Foy R H, Smith R V, Jordan C and Lennox S D, 1995 *Upward trend in soluble phosphorus loadings to Lough Neagh despite phosphorus reduction at sewage treatment works.* Water Research, **29**, 1051-1063

Foy R H, Smith R V, Stevens R J and Stewart D A, 1982 *Identification of factors affecting nitrogen and phosphorus loadings to Lough Neagh*. Journal of Environmental Management, **15**, 109-129

Fozzard I R, Doughty C R, Ferrier R C, Leatherland T M and Owen R, 1999 *A quality classification for management of Scottish standing waters.* Hydrobiologia **395/396**: 433-453

Fraser A I and Harrod T R, 1998 *A systematic approach to predict phosphorus transfer from agriculture using GIS, Proceedings of the Practical and Innovative Measures for the Control of Agricultural Phosphorus Losses to Water*, OECD-sponsored workshop, Greenmount College of Agriculture and Horticulture, Northern Ireland, June 1998, pp. 182-183

Frost A F, 1996 *Loch Leven and diffuse pollution*. In Diffuse pollution and Agriculture. Proceedings of a conference held in Edinburgh (12-14 April 1995), edited by A M Petchey, B J D'Arcy and C A Frost, 174-182. Scottish Agricultural College, Aberdeen.

Gburek W J, Sharpley A S N, Heathwaite A L and Folmar G, 2000 *Phosphorus management at the watershed scale.* Journal of Environmental Quality, **29**, 130-144

Gosselain V, 1998 *Phytoplancton de la Mouse et de la Moselle et l'impact du broutage par le zooplancton*. Presses Universitaires de Namur, Namur.

Hall G H, Corry J E, and Lishman J P, 1993 *The surface water quality of Esthwaite Water in response to a reduction of the phosphate load from Hawkshead sewage treatment works. The potential of external and internal phosphorus loads to prevent lake restoration*. Report to National rivers Authority (North West region), 36 pp

Harper D, 1992 *Eutrophication of freshwaters*. Chapman and Hall, London.

Harris G P, 1998 *Predictive models in spatially and temporally variable freshwater systems.* Australian Journal of Ecology, **23**, 80-94

Haslam S M, Sinker C A and Wolseley P A, 1975 *British water plants*. Field Studies, **4**, 243–351

Heathwaite A L and Sharpley A N, 1999 *Evaluating measures to control the impact of agricultural phosphorus on water quality.* Water Science and Technology, **39**, 149-155

Heathwaite A L, Sharpley A and Gburek W, 2000b *A conceptual approach for integrating phosphorus and nitrogen management at watershed scales*. Journal of Environmental Quality, **29**, 158-166

Heathwaite A L, Fraser A I, Johnes P J, Hutchins M, Lord E and Butterfield D, 2003 *The phosphorus indicators tool: a simple model of diffuse P loss from agriculture land to water*. Soil Use and Management, **19**, 1-11

Hennesey M M, Wilson L, Struthers W and Kelly L A, 1996 *Waste loadings from two freshwater Atlantic salmon juvenile farms in Scotland.* Water, Air and Soil Pollution, **86**, 235-245

Hilton J, Irons G P and McNally S (1999) *Pilot Catchment Study of Nutrient Sources – Control Options and Costs*. pp.107. Environment Agency. Technical Report P345.

Hooda P S, Edwards A C, Anderson H A and Miller A, 2000 *A review of water quality concerns in livestock farming areas.* The Science of the Total Environment, **250**,143-167

Huszar V L M, Silva L H S, Marinho M, Domingos P and Sant'Anna C L, 2000 *Cyanoprokaryote assemblages in eight productive tropical Brazilian waters*. Hydrobiologia, **424**, 67–77

IMPRESS, 2002 *Guidance for the analysis of Pressures and Impacts in accordance with the Water Framework Directive*. IMPRESS CIS Working Group Internal Report, Final version 5.3: 04 December 2002

Johnes P J, 1999 *Understanding catchment history as a tool for integrated lake and catchment management*. Hydrobiologia, **395/396**, 41-60

Johnes P J, Bennion H B, Curtis C, Moss B, Whitehead P and Patrick S, 1998 *Trial Classification of Lake Water Quality in England and Wales: a proposed approach*. Environment Agency R&D Project Record E2-i721\5, Bristol.

Johnes P J, Moss B and Phillips G L, 1996 *The determination of water quality by land use, livestock numbers and population data - testing of a model for use in conservation and water quality management.* Freshwater Biology, **36**, 451-473

Johnes P J, 1996 *Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach*. Journal of Hydrology, **183**, 323-349

Johnes P J and Heathwaite A L, 1997 *Modelling the impact on water quality of land use change in agricultural catchments.* Hydrological Processes, **11**, 269-286

Kadiri M O and Reynolds C S, 1993 *Long-term monitoring of the conditions of lakes: the example of the English Lake District*. Archiv für Hydrobiologie, **129**, 157–178

Kelly L A, Stellwagen J and Bergheim A, 1996 *Waste loadings from a freshwater Atlantic salmon farm in Scotland.* Water Resources Bulletin, **32**, 1017-1025

Kilinc S, 1995 *Limnological studies on the North West Midland meres, with special reference to White Mere*. PhD Thesis, University of Liverpool, 264 pp.

Kirk J T O, 1994 *Light and photosynthesis in aquatic ecosystems*. Cambridge University Press.

Kruk C, Mazzeo N, Lacerot G and Reynolds C S, 2002 *Classification schemes for phytoplankton: a local validation of a functional approach to the analysis of species temporal replacement.* Journal of plankton Research, **24**, 901–912

LLCMMS Lough Leane Catchment Monitoring and Management System, 2000 *A catchment based approach for reducing nutrient inputs from all sources to the lakes of Killarney.* Second Interim Report, December 2000, 91pp

Lord E I and Anthony S G, 2000 MAGPIE: *A modelling framework for evaluating nitrate losses at national and catchment scales.* Soil Use Mgt., **16**, 167-174

May L and Gunn I D M G, 2000 *An assessment of the possible impact of gulls from a proposed landfill site on the nutrient load to Loch Ussie, Easter Ross, Scotland*. Report to S.I.T.A. Northeast (NEM), 20pp., 5 Figures, 2 Plates

May L, Gunn I D M and Kirika A, 2001 *Phosphorus study at Loch Flemington*. Report to Scottish Natural Heritage, 37pp., 10 tables, 9 Figures

May L, Place C J and George D G, 1996 *The effects of soil type on nutrient losses and runoff in the catchment of Bassenthwaite Lake*. Report to National Rivers Authority, NW Region, 46 pp.

May L, Place C J and George D G, 1997a *An assessment of the phosphorus load to Esthwaite Water and the relative importance of diffuse and point sources within the catchment*. Report to Environment Agency (NW Region) 44 pp., 14 Figures, 2 Appendices

May L, Place C J, George D G and McEvoy J, 1997b *Assessment of the nutrient loadings from the catchment to Bassenthwaite Lake*. Report to Environment Agency (NW Region) 49pp

May L, Place C J and George D G, 1998 *Estimation of the phosphorus load to surface waters from septic tanks in rural catchments*. In: Petchey A M, D'Arcy B J, and Frost C A (Eds) Diffuse pollution and Agriculture II. Proceedings of a conference held in Edinburgh, 9-11 April 1997. Scottish Agricultural College, Aberdeen, pp. 212-216

Melo S de and Huszar V L M, 2000 *Phytoplankton in an Amazonian flood-plain lake (Lago Batata, Brasil): diel variation and species strategies*. Journal of Plankton Research, **22**, 63–76

Middleboe A L and Markager S, 1997 *Depth limits and minimum light requirements of freshwater macrophytes.* Freshwater Biology **37**, 553-568.

Moss B, Beklioglu M, Carvalho L, Kilinc S, McGowan S and Stephen D, 1997 *Vertically-challenged limnology: contrasts between deep and shallow lakes*. Hydrobiologia, **342**, 257-267

Muscutt A D and Withers P J A, 1996 *The phosphorus content of rivers in England and Wales.* Water Research, **30**, 1258-1268

OECD, 1982 *Eutrophication of waters: monitoring, assessment and control*. OECD, Paris, 154 pp

Padisák J and Reynolds C S, 1998 *Selection of phytoplankton associations in Lake Balaton, Hungary, in response to eutrophication and restoration measures, with special reference to the cyanoprokaryotes.* Hydrobiologia, **384**, 41 – 53

Phillips G, 2003a *Reporting Typology for Ecoregion 18, Great Britain* – UKTAG Lakes Task Team Internal Report TAG/LTT 43, March 2003

Phillips G, 2003b *Approach to risk assessment for eutrophication in lakes – contribution to UKTAG Task 7f*. Lakes Task Team Internal Report

Phillips G, 2003c *Derivation of Lake P targets for WFD*. Lakes Task Team Internal Report

Quinn P, 2002 *Models and monitoring: scaling up cause and effect relationships in nutrient pollution to the catchment scale*. In Agricultural Effects on Ground and Surface Water: Research at the Edge of Science and Society. IAHS Publication **273**, 397-404

Reynolds C S and Maberly S C, 2002 *A simple method for approximating the supportive capacities and metabolic constraints in lakes and reservoirs*. Freshwater Biology, **47**, 1183-1188

Reynolds C S, Huszar V, Kruk C, Naselli-Flores L and Melo S, 2002 *Towards a functional classification of the freshwater phytoplankton.* Journal of Plankton Research, **24**, 417-428

Reynolds C S and Descy J –P, 1996 *The production, biomasa and structure of phytoplankton in large rivers.* Archiv für Hydrobiologie, Supplementband, **113**, 161–187

Reynolds C S, 1980 *Phytoplankton assemblages and their periodicity in stratifying lake systems.* Holarctic Ecology, **3**, 141–159

Reynolds C S, 1984 *Phytoplankton periodicity: the interactions of form, function and environmental variability*. Freshwater Biology, **14**, 111-142

Reynolds C S, 2000 *Phytoplankton designer – or how to predict compositional responses to trophic-state change*. Hydrobiologia, **424**, 67–77

Reynolds C S, 2001 *Hydroecology of river plankton: the role of variability in channel flow.* Hydrological Processes, **14,** 3119 – 3132

Reynolds C S, (submitted) *The development of perceptions of aquatic eutrophication and its control.* Ecohydrology and Hydrobiology.

Sand-Jensen K and Madsen T V, 1991 *Minimum light requirements of submerged freshwater macrophytes in laboratory growth experiments*. Journal of Ecology **79**, 749-764

Sayer C D, 2001 *Problems with the application of diatom-total phosphorus transfer functions: examples from a shallow English lake.* Freshwater Biology, **46**, 743-757

SCOPE (1999) *Implementation of the 1991 EU Urban Waste Water Treatment Directive, and its role in reducing phosphate discharges.* Summary Report 34, Scientific Committee of Phosphates in Europe, Belgium

Sharpley A N, Chapra S C, Wedepohl R, Sims J T Daniel T C and Reddy K R, 1994 *Managing agricultural phosphorus for the protection of surface waters: issues and opinions.* Journal of Environmental Quality, **23**, 437-451

Shimwell D W, 1971 *Description and classification of vegetation*. Sidgwick and Jackson: London.

Smith V H, Tilman G D and Nekola J C, 1999 *Eutrophication: impacts of nutrient inputs to freshwater, marine and terrestrial ecosystems*. Environmental Pollution, **100**, 179-196

SNIFFER, 2003 *Assessment for the Development of a Screening Tool to Identify and Characterise Diffuse Pollution Pressures*, Edinburgh.

Soil Science Society of America, 2000 *Agricultural Nutrient Management and Environmental Quality*, Position of the Soil Science Society of America; prepared by Sims T., Glasner K. and Hall, T. Department of Plant and Soil Sciences, University of Delaware, Newark, DE 19717-1303, ASA/SSSA/CSSA, 2pp.

Spence D H N, 1982 *The zonation of plants in freshwater lakes*. Advances in Ecological Research **12**: 37-125

Tansley A G, 1935 *The British Isles and their vegetation*. Cambridge University Press: Cambridge

Tüxen R, 1955 *Das Systeme der nordwestdeutschen Pflanzengesellschaft. Mitteilungen der Florist-soziologische*. Arbeitsgemeinschaft, **5**, 1–119

UK TAG, 2003 *Guidance on general principles for pressures and impacts analysis (Final)*. UK TAG Internal Report for Work Plan Task 7.a. (01), 4 September 2003.

Vighi M and Chiaudani G, 1985 A simple method to estimate lake phosphorus concentrations resulting from natural loadings. *Water Research*, **19**, 987-991

Viney N R, Sivapalan M and Deeley D, 2000 *A conceptual model of nutrient mobilisation and transport applicable at large catchment scales*. Journal of Hydrology, **240**, 23-44

Weld J L and Beegle D B, 2001 *A summary of phosphorus indices*. SERA17 Meeting, July 18-21, State College, PA, USA

Weller N, 2000 *The development of a phosphorus budget for Loch Earn, Tayside*. M.Sc. Thesis, University of Edinburgh

Wu J and Loucks O L, 1995 *From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology*. Quarterly Review of Biology, **70**, 439-466

APPENDIX 1: SUMMARY RISK ASSESSMENT GUIDANCE

A1.1 Introduction

The interim risk assessment guidance follows the procedure illustrated schematically in Figure A1.1. The first stage involves estimation of all diffuse and point-source nutrient pressures, which are summed to give a total nutrient load. The impact of these pressures is then considered in terms of modelled in-lake nutrient concentrations, derived from the nutrient load. Modelled concentrations are then compared with reference concentrations (preferably site-specific) to assess the risk of failing good status. The sensitivity of the lake to the nutrient pressures is incorporated in the conversion of nutrient load to in-lake nutrient concentrations (using site-specific retention times) and indirectly in the reference conditions derived for each lake ecotype.

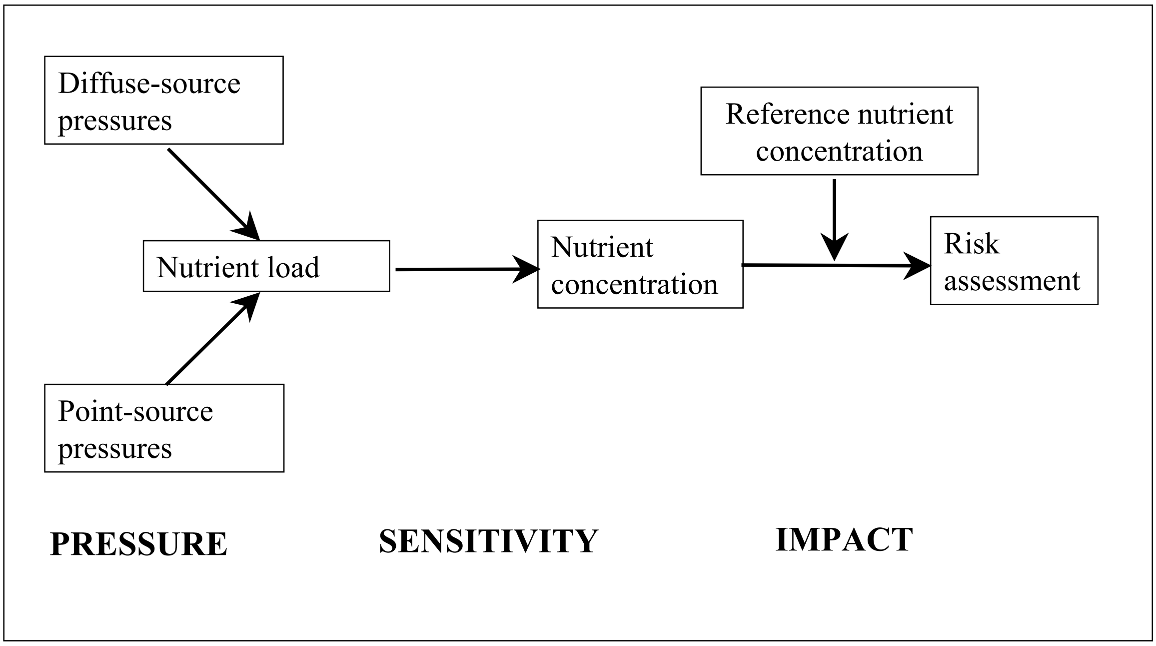


Figure A1.1 Scheme of risk assessment procedure

A1.2 Assessment of Diffuse Source Nutrient Pressures

The Phosphorus Indicators Tool (PIT) is currently being calibrated and validated on a number of test catchments across the UK. In the interim the calculation steps for diffuse nutrient loads should be estimated by the "export coefficient" approach using GIS-derived catchment land use as outlined in Bennion *et al*. (2002). The calculation should, however, use the revised export coefficients outlined in this report (Table 6.X).

A1.3 Assessment of Point-Source Nutrient Pressures

A1.3.1 Sewage works and septic tanks

It is difficult to differentiate between the P load to surface waters from STW effluent and that from septic tanks at the national scale due to limitations in the available data. Until this situation improves, and per capita P export coefficients can be determined more accurately, the following approach is recommended for estimating the P load to lakes from these sources:

Step 1. Estimate the resident population (N_{1991}) in each catchment from the 1991 population census data
Step 2. Multiply N_{1991} by a *per capita* P export value of 0.4 kg y^{-1}

If it is known that tertiary treatment exists for a particular site, then the value of 0.4 kg y^{-1} should be reduced by 80% to 0.08 kg y^{-1}.

A1.3.2 Fish farms

Most information on the way that fish farms are managed, such as the amount and type of feed used and the mortality and production rates of fish, is confidential and only available from individual fish farm operators. The only widely available data on fish farms is the consented annual rate of fish production or biomass, which is held by the regulatory agencies. A methodology based on these data is probably the best approach to assessing the P load from cage fish farms, at present:

Step 1. Obtain information on the location and annual biomass production of each cage fish farm; enter these into a GIS
Step 2. Determine the annual tonnage of fish produced by cage fish farms in each lake (B_{CAGE})
Step 3. Multiply this value by a P export coefficient of 10 kg P per tonne of fish per year for salmon and 26 kg P per tonne of fish per year for trout.

The data required in Steps 1 and 2 are incomplete at present and require further compilation for application on a national basis.

A1.4 Assessment of Total Nutrient Pressures

TP load expressed as kg yr^{-1}, is calculated for each lake catchment by summing the total contribution from diffuse- and point-sources.

A1.5 Assessment of Impact

It is recommended that for risk assessment purposes, the ecological response is only considered in terms of supporting nutrient conditions within the lake. The response in terms of phytoplankton composition and abundance requires further development of classification schemes (See Chapters 3 and 4). Even when phytoplankton classification schemes are established, tier 1 risk assessment is still likely to focus on modelled nutrient concentrations. More sophisticated (and data hungry) phytoplankton models (such as PROTECH) are more applicable to higher-tier site-specific risk assessments and in defining a suitable programme of measures for a site

A1.5.1 Nutrient concentrations

The risk assessment for nutrient concentrations is structured into a number of steps

Step 1. Determine water body type based on LTT typology (Phillips, 2003a)
Step 2. Ideally identify site-specific reference conditions, using approaches outlined in Section 2.4. If not available, use type-specific reference conditions outlined in Table 2.6.
Step 3. Establish current nutrient conditions, preferably from measured total phosphorus concentrations (minimum four, regularly-spaced sampling occasions per year).
Step 4. If measured data are unavailable, estimate current conditions from nutrient load determined by GIS-derived catchment land-use and point-source pressures as outlined in Sections A1.2 and A1.3. TP load can then be converted to an in-lake concentration using the equations outlined in OECD (1982).
Step 5. Calculate an Ecological Quality Ratio (EQR) by dividing reference TP concentration with current (measured or modelled) TP concentration.

“Not at risk” status is achieved with an EQR of 0.5 or above. If below 0.5, list water-body as “at risk” of failing to achieve good status.

APPENDIX 2: LAKE DATA USED IN THE PROJECT

All data is taken from the GB lakes database (http://ecrc.geog.ucl.ac.uk/gblakes/), where the source of the data is available. Access is currently password restricted, please contact Geoff Phillips at the Environment Agency or Ian Fozzard at SEPA to discuss access.

Values presented for mean depth are all based on measurements taken from bathymetric surveys

Values provided for water chemistry are annual means based on at least four-regularly spaced sampling occasions throughout the year.

WBID	OSNAME	Geotype	Mean Depth (m)	Alkalinity (μequiv. l^{-1})	TP ($\mu g l^{-1}$)	Chl_a ($\mu g l^{-1}$)
Peat - All						
41210	Llyn Fach	P	1.7	99	10	1
38623	Maes-Llyn	P	2.7	527	53	23
704	Eela Water	P	5.1	106	37	4
1271	Loch of Girlsta	P	9.6	106	19	21
Low Alkalinity - Deep						
35233	Gloyw Lyn	Si_LA	3.0	25	5	2
20754	Loch Loyne	Si_LA	3.2	39	10	1
33836	Llyn Idwal	Si_LA	3.4	70	5	1
18825	Lochindorb	Si_LA	3.8	167	13	1
38544	Llyn Fanod	Si_LA	3.8	108	18	3
37834	Llynnoedd Ieuan	Si_LA	3.9	-9	5	1
20657	Loch Morlich	Si_LA	4.5	72	18	3
28965	Derwent Water	Si_LA	5.5	96	7	5
34319	Llyn Llagi	Si_LA	5.8	6	4	2
40297	Llyn y Fan Fawr	Si_LA	6.0	86	11	4
23559	Loch of Lowes	Si_LA	6.2	445	19	12
19935	Loch Mhor	Si_LA	7.3	63	14	1
24754	Loch Chon	Si_LA	7.6	30	7	1
34400	Llyn Conwy	Si_LA	7.7	33	15	2
29184	Grasmere	Si_LA	7.7	157	24	12
20647	Loch Alvie	Si_LA	8.2	165	19	7
28986	Loweswater	Si_LA	8.4	195	16	10
14585	Loch Bad an Sgalaig	Si_LA	9.7	71	13	1
13463	Loch a' Bhaid-luachraich	Si_LA	10.4	83	12	1
24744	Loch Achray	Si_LA	11.0	58	6	2
20860	Loch Insh	Si_LA	11.4	156	29	6
24295	Loch Voil	Si_LA	12.5	97	5	1
24758	Loch Venachar	Si_LA	12.9	152	6	1
24459	Loch Lubnaig	Si_LA	13.0	177	8	2
24892	Loch Ard	Si_LA	13.4	65	8	1
22725	Loch Tummel	Si_LA	14.6	144	13	2
29021	Thirlmere	Si_LA	16.1	64	4	1
29052	Buttermere	Si_LA	16.6	51	1	2
29062	Ennerdale Water	Si_LA	17.8	42	2	1
19572	Loch Beinn a' Mheadhoin	Si_LA	19.9	30	8	1
21576	Loch Laggan	Si_LA	20.6	55	14	0
36267	Llyn Cau	Si_LA	21.0	17	4	1
34002	Llyn Cwellyn	Si_LA	22.6	37	7	2
29073	Haweswater Reservoir	Si_LA	23.4	196	8	3
34987	Llyn Tegid or Bala Lake	Si_LA	24.0	134	14	6
29000	Crummock Water	Si_LA	26.7	53	3	3
18216	Loch Monar	Si_LA	30.0	37	11	1
21790	Loch Muick	Si_LA	35.4	30	21	4
24447	Loch Lomond	Si_LA	37.0	230	9	3
14057	Loch Maree	Si_LA	38.2	50	9	1
29183	Wast Water	Si_LA	40.2	58	1	1
22010	Loch Treig	Si_LA	63.2	60	11	0
Low Alkalinity - Shallow						
43651	Albury Mill	Si_LA	0.5	119	334	3
33474	Oak Mere	Si_LA	1.6	30	65	7
37080	Llyn Glanmerin	Si_LA	1.6	97	15	3
37437	Bugeilyn	Si_LA	1.9	7	18	3
38525	Llyn Gynon	Si_LA	2.2	13	8	2
38422	Llyn Eiddwen	Si_LA	2.6	89	21	8
38394	Llyn Hîr	Si_LA	2.8	14	7	2

WBID	OSNAME	Geotype	Mean Depth (m)	Alkalinity (μequiv. l^{-1})	TP ($\mu g l^{-1}$)	Chl_a ($\mu g l^{-1}$)
Medium Alkalinity - Deep						
7	Loch of Cliff	Si_MA	3.2	527	27	5
19540	Loch Ruthven	Si_MA	3.4	218	16	6
23531	Loch of Butterstone	Si_MA	3.4	518	26	14
26581	Gladhouse Reservoir	Si_MA	5.0	525	19	8
28847	Bassenthwaite Lake	Si_MA	5.3	180	21	15
24919	Lake of Menteith	Si_MA	6.0	430	18	9
29328	Esthwaite Water	Si_MA	6.4	425	30	21
18982	Loch Ashie	Si_MA	6.5	479	12	3
29270	Blelham Tarn	Si_MA	6.8	470	32	15
29233	Windermere	Si_MA	16.8	269	14	6
29321	Coniston Water	Si_MA	24.1	200	7	6
29233	Windermere	Si_MA	25.1	226	12	4
28955	Ullswater	Si_MA	25.3	236	10	5
19214	Loch Duntelchaig	Si_MA	25.6	259	10	1
24132	Loch Earn	Si_MA	42.0	342	13	3
18767	Loch Ness	Si_MA	132.0	61	6	1
Medium Alkalinity - Shallow						
34622	Llyn Glasfryn	Si_MA	0.7	439	147	101
43218	Bolder Mere	Si_MA	0.7	860	43	5
45422	Abbotts Wood	Si_MA	0.9	810	98	16
40571	Llyn Llech Owen	Si_MA	1.2	-8	48	38
21123	Loch Davan	Si_MA	1.2	516	33	7
44635	Cinder Hill	Si_MA	1.2	690	193	27
20757	Loch of Skene	Si_MA	1.4	700	90	57
21189	Loch Kinord	Si_MA	1.5	228	28	6
44241	Shortheath Common	Si_MA	1.5	590	52	7
21187	Loch of Aboyne	Si_MA	1.8	590	22	12
39813	Abbey Lake or Lower Talley L	Si_MA	1.9	343	69	25
45221	Farthing Lake	Si_MA	1.9	430	93	15
39796	Upper Talley Lake	Si_MA	1.9	448	51	11

WBID	OSNAME	Geotype	Mean Depth (m)	Alkalinity (µequiv. l^{-1})	TP ($\mu g l^{-1}$)	Chl_a ($\mu g l^{-1}$)
High Alkalinity - Deep						
39267	Llan Bwch-llyn Lake	Calc	3.0	1391	36	15
34780	Hanmer Mere	Calc	3.0	1685	1806	4
40608	Marsworth Reservoir	Calc	3.0	4200	476	76
25038	Gartmorn Dam	Calc	3.3	1796	53	24
1570	Loch of Spiggie	Calc	3.5	1086	34	15
24843	Loch Leven	Calc	4.5	1407	59	40
1418	Loch of Tingwall	Calc	5.8	1396	22	6
32761	Llyn yr Wyth-Eidion	Calc	6.0	3997	17	4
38355	Mitcham Pit	Calc	8.0	1950	115	10
32650	Rostherne Mere	Calc	13.6	2208	290	24
High Alkalinity - Shallow						
33627	Llyn Rhos-ddu	Calc	0.6	2483	42	13
38968	Lodge Lake, Branches Park	Calc	0.8	2000	46	4
36202	Upton Broad	Calc	0.8	2701	31	12
43315	Fleet Pond	Calc	0.9	2240	461	27
39798	Debden Park	Calc	0.9	3330	140	48
35791	Alderfen Broad	Calc	1.0	1775	655	14
38758	Grendon Quarter Pond	Calc	1.0	3510	167	59
35977	Hoveton Great Broad	Calc	1.0		94	61
44196	Lower Pond, Beachborough L	Calc	1.1	3610	105	35
44038	Hammer Pond	Calc	1.2	2380	110	5
35923	Hoveton Little Broad	Calc	1.2	3477	70	56
1853	Loch of Kirbister	Calc	1.3	1654	35	2
1692	Loch of Hundland	Calc	1.3	1993	30	2
36143	South Walsham Broad	Calc	1.3	3506	193	133
35953	Wroxham Broad	Calc	1.3	4164	76	45
32948	Llyn Dinam	Calc	1.4	1533	112	8
43941	Wire Mill Lake	Calc	1.5	1470	646	45
39053	Decoy Lake, Arlington	Calc	1.5	1790	26	4
43943	Frensham Little Pond	Calc	1.5	1910	68	27
35645	Horsey Mere	Calc	1.5	2714	52	32
35655	Barton Broad	Calc	1.5	3385	92	69
36050	Ranworth Broad	Calc	1.5	3465	103	79
40634	Bonnington Lake	Calc	1.5	3540	319	143
1694	Loch of Boardhouse	Calc	1.8	1814	31	3
33337	Llyn Coron	Calc	1.8	1869	156	21
42170	Kenfig Pool	Calc	1.8	2034	22	15
40067	Llangorse Lake	Calc	2.0	2446	118	15
32968	Llyn Penrhyn	Calc	2.2	2153	1085	4
25128	Loch Fitty	Calc	2.3	1341	77	14
35981	Rollesby Broad	Calc	2.5	2787	116	21
1753	Loch of Harray	Calc	2.7	2539	29	6
1678	Loch of Swannay	Calc	2.8	1302	27	3
32744	The Mere	Calc	2.8	1510	54	16
High Alkalinity (Marl) - Shallow						
29844	Malham Tarn	Calc_Marl	2.6	2040	17	8
Brackish - Shallow						
35640	Hickling Broad	Brackish	1.3	2640	70	52

APPENDIX 3: TEMPLATES OF PHYTOPLANKTON FUNCTIONAL GROUPS

Lake Type 1S

U

Season 1						Season 2						Season 3						Season 4						Season 5					
Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

O

Season 1						Season 2						Season 3						Season 4						Season 5					
Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

M

Season 1						Season 2						Season 3						Season 4						Season 5					
Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

E

Season 1						Season 2						Season 3						Season 4						Season 5					
Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

H

Season 1						Season 2						Season 3						Season 4						Season 5					
Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

Lake Type 1D

U

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

O

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

M

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

E

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

H

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Lake Type 2S

	Season 1						Season 2						Season 3						Season 4						Season 5					
U	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
O	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
M	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
E	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
H	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

Lake Type 2D

U

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

O

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

M

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

E

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

H

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Lake Type 3S

U

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

O

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

M

Season 1

Z	A	E	F	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

E

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

H

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Lake Type 3D

U

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

O

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

M

Season 1

Z	A	E	F	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

E

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

H

Season 1

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 2

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 3

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 4

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Season 5

Z	A	E	S2	U	N
X3	B	F	H2	Lo	P
X2	C	G	H1	Lm	R/V
X1	D	Y	Sn	K	T
W2	W1	Q	J	M	S1

Lake Type 4S

	Season 1						Season 2						Season 3						Season 4						Season 5					
U	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
O	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
M	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
E	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
H	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

Lake Type 4D

	Season 1						Season 2						Season 3						Season 4						Season 5					
U	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
O	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
M	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
E	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1
H	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N	Z	A	E	S2	U	N
	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P	X3	B	F	H2	Lo	P
	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V	X2	C	G	H1	Lm	R/V
	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T	X1	D	Y	Sn	K	T
	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1	W2	W1	Q	J	M	S1

APPENDIX 4: BINARY-CODED PRESENCE/ABSENCE OF PHYTOPLANKTON FUNCTIONAL GROUPS IN EIGHT LAKE TYPES

Table A4.1 Lake Types 1S and 1D

	A	B	C	D	E	F	G	H1	H2	J	K	LO	LM	M	N	P	Q	R/V	S1	S2	SN	T	U	W1	W2	X1	X2	X3	Y	Z
1SU1	1				1																						1	1		1
1SU2	1				1	1																					1	1		1
1SU3					1	1																					1	1		1
1SU4															1													1		1
1SU5	1														1												1	1		1
1SO1	1	1			1																					1	1	1	1	
1SO2					1	1																				1	1	1	1	1
1SO3					1	1						1											1				1	1	1	1
1SO4					1	1						1			1								1				1	1	1	1
1SO5												1			1										1		1	1	1	
1SM1		1																							1	1	1	1		
1SM2					1	1	1			1													1			1	1	1	1	1
1SM3					1	1	1			1		1											1				1	1	1	1
1SM4					1	1				1		1	1		1	1						1	1				1	1	1	1
1SM5					1	1				1		1			1	1									1		1		1	
1SE1		1	1	1																					1	1	1	1	1	
1SE2					1	1	1			1							1								1	1	1	1	1	
1SE3					1	1	1					1											1			1	1	1	1	
1SM4					1	1				1		1	1		1	1						1	1			1	1		1	
1SM5			1							1		1			1	1						1			1	1	1		1	
1SH1			1	1																				1		1	1	1	1	
1SH2					1	1	1			1							1							1		1	1	1	1	
1SH3					1	1	1			1		1					1						1	1		1	1		1	
1SH4					1	1				1		1	1			1	1						1	1		1	1		1	
1SH5					1					1		1			1	1	1					1		1	1	1	1		1	
1DU1	1				1																						1	1		1
1DU2	1				1	1																					1	1		1
1DU3					1	1																					1	1		1
1DU4															1													1		1
1DU5	1														1												1	1		1
1DO1	1	1			1																					1	1	1	1	
1DO2					1	1																				1	1	1	1	1
1DO3					1	1						1											1				1	1	1	1
1DO4					1	1						1			1								1				1	1	1	1
1DO5												1			1							1			1		1	1	1	
1DM1		1																								1	1	1		
1DM2					1	1	1			1													1			1	1	1	1	1
1DM3					1	1						1											1				1	1	1	1
1DM4					1	1						1	1		1							1					1	1	1	1
1DM5					1	1						1			1	1									1		1		1	
1DE1		1	1	1																						1	1	1	1	
1DE2					1	1	1			1							1									1	1	1	1	
1DE3					1	1	1			1		1											1			1	1	1	1	
1DE4					1	1				1		1	1		1	1						1				1	1		1	
1DE5			1							1		1			1	1						1			1	1	1		1	
1DH1			1	1																				1		1	1	1	1	
1DH2					1	1	1			1							1							1		1	1	1	1	
1DH3					1	1	1			1		1					1							1		1	1		1	
1DH4					1	1				1		1	1			1	1							1		1	1		1	
1DH5				1	1					1		1			1	1						1		1	1	1	1		1	

Table A4.2 Lake Types 2S and 2D

	A	B	C	D	E	F	G	H1	H2	J	K	LO	LM	M	N	P	Q	R/V	S1	S2	SN	T	U	W1	W2	X1	X2	X3	Y	Z
2SU1	1				1																					1	1	1		1
2SU2	1				1	1																					1	1		1
2SU3					1	1																					1	1		1
2SU4															1													1		1
2SU5	1														1												1	1		1
2SO1	1	1			1																					1	1	1	1	
2SO2	1	1			1	1			1																	1	1	1	1	1
2SO3					1	1	1	1	1			1			1											1	1	1	1	1
2SO4					1	1			1			1	1		1								1			1	1	1	1	1
2SO5	1	1			1	1						1			1	1			1					1			1	1	1	1
2SM1	1	1	1		1																				1		1	1	1	
2SM2	1	1	1		1	1	1																				1	1	1	1
2SM3					1	1			1			1											1				1	1	1	1
2SM4					1	1		1	1			1	1		1												1	1	1	1
2SM5	1	1			1	1				1		1			1	1						1			1		1	1	1	
2SE1		1	1	1																					1	1	1		1	
2SE2			1	1	1	1	1	1	1																1	1	1	1	1	
2SE3					1	1	1	1	1	1	1	1	1												1	1	1	1	1	
2SE4					1	1	1	1	1		1	1	1			1						1			1	1	1		1	
2SE5			1							1			1			1						1			1	1	1		1	
2SH1		1	1	1																				1		1	1	1	1	
2SH2				1	1	1	1	1		1	1													1		1	1		1	
2SH3						1	1	1		1	1								1					1		1	1		1	
2SH4								1		1	1		1						1					1		1			1	
2SH5				1				1		1			1			1						1		1	1	1	1		1	
2DU1	1				1																					1	1	1		1
2DU2	1				1	1																					1	1		1
2DU3					1	1																					1	1		1
2DU4															1													1		1
2DU5	1														1												1	1		1
2DO1	1	1			1																					1	1	1	1	
2DO2	1	1			1	1			1																	1	1	1	1	1
2DO3					1	1		1	1			1														1	1	1	1	1
2DO4					1	1						1	1										1			1	1	1	1	1
2DO5	1	1			1	1						1			1										1		1	1	1	1
2DM1	1	1	1		1																						1	1	1	
2DM2	1	1	1		1	1	1																				1	1	1	1
2DM3					1	1			1			1															1	1	1	1
2DM4						1		1	1			1	1		1			1					1				1	1	1	1
2DM5	1	1			1							1			1	1		1				1			1	1	1		1	
2DE1		1	1	1																						1	1		1	
2DE2			1	1	1	1	1	1	1																	1	1	1	1	
2DE3						1	1	1	1			1	1					1								1	1		1	
2DE4							1	1	1		1	1	1					1									1		1	
2DE5				1									1			1		1				1			1	1	1		1	
2DH1		1	1	1																				1		1	1	1	1	
2DH2				1			1	1		1	1													1		1	1		1	
2DH3							1	1			1													1		1	1		1	
2DH4								1			1		1											1		1			1	
2DH5				1				1					1			1		1	1			1		1	1	1	1		1	

Table A4.3 Lake Types 3S and 3D

	A	B	C	D	E	F	G	H1	H2	J	K	LO	LM	M	N	P	Q	R/V	S1	S2	SN	T	U	W1	W2	X1	X2	X3	Y	Z
3SU1	1				1	1																			1	1	1	1		1
3SU2	1				1	1																					1	1	1	1
3SU3					1	1									1								1			1	1	1	1	1
3SU4					1							1			1								1			1	1	1		1
3SU5	1														1										1	1	1	1		1
3SO1	1	1													1												1	1	1	
3SO2	1	1			1	1																				1	1	1	1	1
3SO3	1				1	1			1			1											1		1		1	1	1	1
3SO4	1				1	1		1	1			1			1								1		1		1	1	1	1
3SO5	1				1	1						1			1							1			1		1	1	1	
3SM1	1	1	1		1	1																				1	1		1	
3SM2		1			1	1			1																	1	1	1	1	
3SM3					1	1	1	1	1			1				1										1	1	1	1	1
3SM4					1	1		1	1			1	1		1	1							1				1	1	1	
3SM5	1	1			1							1			1	1						1					1	1	1	
3SE1			1	1															1					1		1	1		1	
3SE2			1	1	1	1	1	1		1									1							1	1		1	
3SE3			1	1		1	1	1		1	1		1						1							1	1		1	
3SE4				1		1	1	1		1	1	1	1	1		1			1							1	1		1	
3SE5			1	1						1									1			1		1		1	1		1	
3SH1			1	1															1					1	1	1	1		1	
3SH2			1	1	1	1	1			1	1					1			1					1	1	1	1		1	
3SH3				1	1	1	1	1		1	1		1						1			1		1	1	1	1		1	
3SH4				1		1	1	1		1	1		1	1					1			1		1	1	1	1		1	
3SH5				1				1		1	1		1			1			1			1		1	1	1	1		1	
3DU1	1				1	1																			1	1	1	1		1
3DU2	1				1	1																					1	1	1	1
3DU3					1	1									1								1			1	1	1	1	1
3DU4					1							1			1								1			1	1	1		1
3DU5	1														1										1	1	1	1		1
3DO1	1	1													1												1	1	1	
3DO2	1	1			1	1																				1	1	1	1	1
3DO3	1				1	1			1			1													1		1	1	1	1
3DO4	1				1	1		1	1			1	1		1								1		1		1	1	1	1
3DO5	1				1	1						1			1										1		1	1		
3DM1	1	1	1																								1		1	
3DM2		1			1	1			1																	1	1	1	1	
3DM3		1			1	1		1	1			1				1		1								1	1	1	1	1
3DM4					1	1		1	1			1	1		1	1		1						1			1	1	1	
3DM5	1	1			1							1			1	1		1				1			1		1	1	1	
3DE1			1	1														1	1					1		1	1		1	
3DE2			1				1	1		1																1			1	
3DE3			1			1	1	1		1			1													1			1	
3DE4						1	1	1		1	1	1	1			1										1			1	
3DE5			1	1									1			1		1	1			1		1		1			1	
3DH1			1	1															1					1	1	1	1		1	
3DH2			1	1		1	1				1					1			1					1	1	1	1		1	
3DH3				1		1	1	1		1	1		1	1		1			1			1		1	1	1	1		1	
3DH4						1	1	1		1	1		1	1				1	1			1		1	1	1	1		1	
3DH5				1				1		1	1		1	1		1		1	1			1		1	1	1	1		1	

Table A4.4 Lake Types 4S and 4D

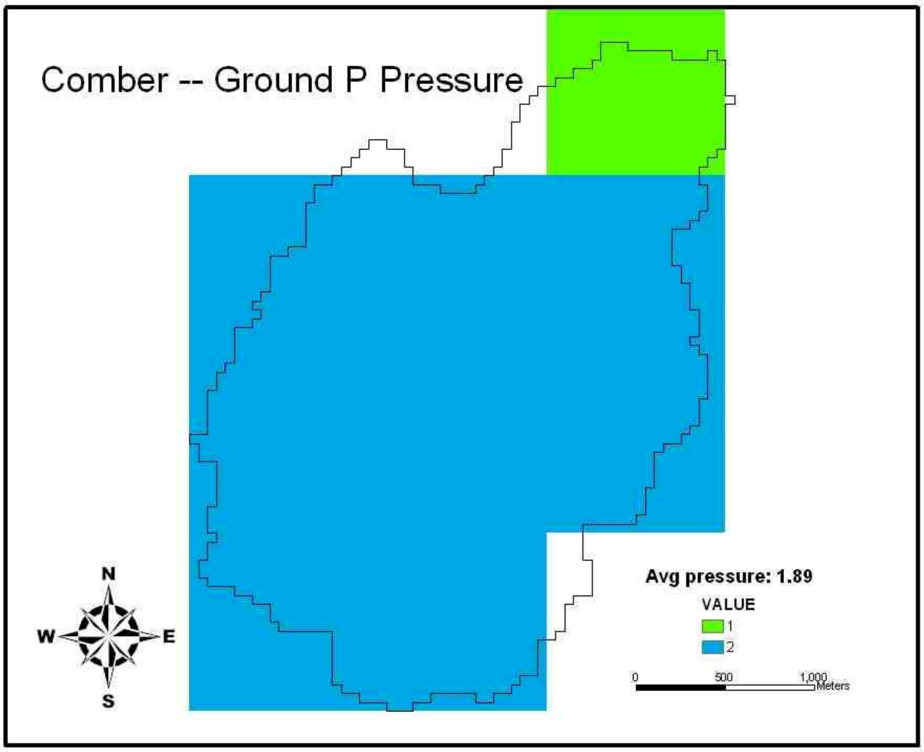
	A	B	C	D	E	F	G	H1	H2	J	K	LO	LM	M	N	P	Q	R/V	S1	S2	SN	T	U	W1	W2	X1	X2	X3	Y	Z
4SU1	1	1																									1			
4SU2						1																				1	1			1
4SU3																											1			1
4SU4													1			1											1			1
4SU5		1														1									1		1			1
4SO1	1	1	1																							1	1		1	
4SO2					1	1	1		1																	1	1		1	1
4SO3						1			1				1													1	1		1	1
4SO4									1		1		1			1										1	1			1
4SO5	1	1														1						1				1	1		1	
4SM1		1	1	1																				1		1	1		1	
4SM2					1	1	1	1	1																	1	1		1	
4SM3						1	1	1	1		1															1	1		1	1
4SM4						1		1	1		1	1	1			1										1	1		1	
4SM5													1			1						1		1	1				1	
4SE1			1	1												1			1					1		1	1		1	
4SE2			1	1		1	1	1		1	1															1	1		1	
4SE3				1			1	1		1	1		1													1	1		1	
4SE4				1			1	1		1	1		1	1								1				1	1		1	
4SE5				1									1			1			1			1		1		1	1		1	
4SH1			1	1												1			1					1		1			1	
4SH2			1	1	1	1	1	1		1	1								1					1		1	1		1	
4SH3				1		1	1	1		1	1		1	1					1			1		1		1	1		1	
4SH4						1	1	1		1	1		1	1					1			1		1		1	1		1	
4SH5				1									1	1		1			1			1		1		1			1	
4DU1	1	1																									1			
4DU2						1																				1	1			1
4DU3																											1			1
4DU4													1			1											1			1
4DU5		1													1	1									1		1			
4DO1	1	1	1																							1	1		1	
4DO2						1	1		1																	1	1		1	1
4DO3						1			1																	1	1			1
4DO4									1		1		1													1	1			1
4DO5	1	1														1						1				1	1		1	
4DM1		1	1	1																				1		1	1		1	
4DM2						1	1	1	1																	1	1		1	
4DM3						1		1	1		1		1													1	1		1	1
4DM4								1	1		1	1	1					1								1	1		1	
4DM5													1			1		1				1		1	1				1	
4DE1			1	1															1					1		1	1		1	
4DE2			1	1		1	1	1		1																1	1		1	
4DE3				1			1	1		1	1		1													1			1	
4DE4								1			1		1									1				1			1	
4DE5				1									1			1			1			1		1		1			1	
4DH1			1	1												1			1					1		1			1	
4DH2			1	1		1	1	1		1	1													1		1	1		1	
4DH3				1		1	1	1		1	1		1									1		1		1	1		1	
4DH4						1	1	1		1	1		1	1								1		1		1	1		1	
4DH5				1									1			1			1			1		1		1			1	

APPENDIX 5: GRAPHICAL OUTPUT FROM TIER 2, 3A AND 3B DIFFUSE MODELLING APPROACHES

Please refer to Environment Agency for output files

A5. Appendix 5, Tier 2

Figure A5.2. 1 Subsurface and surface P pressure maps for the Comber Mere catchment using the tier 2 approach



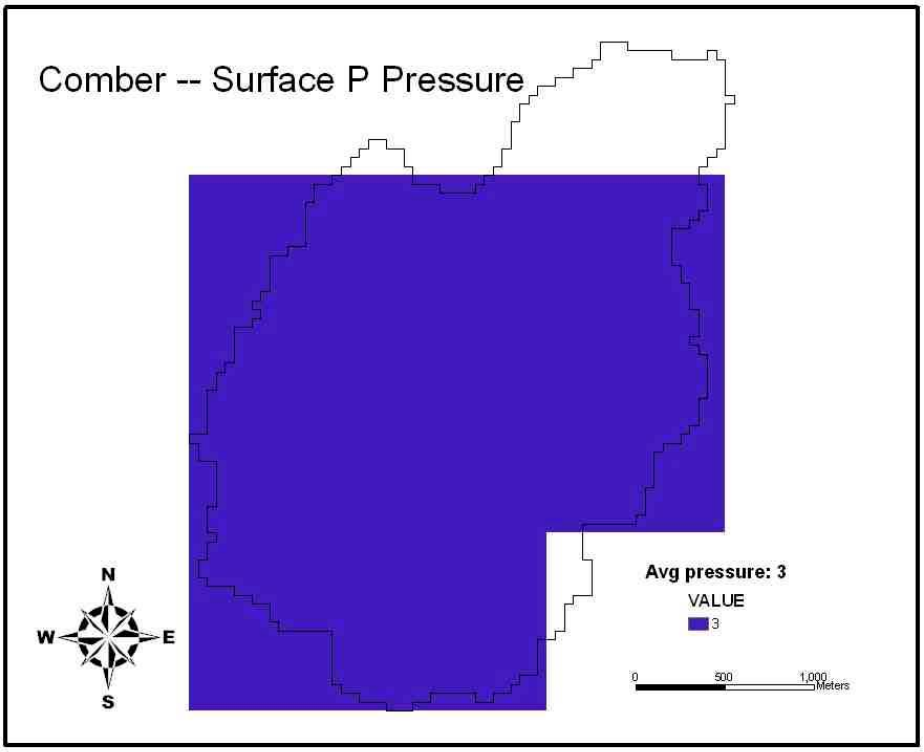
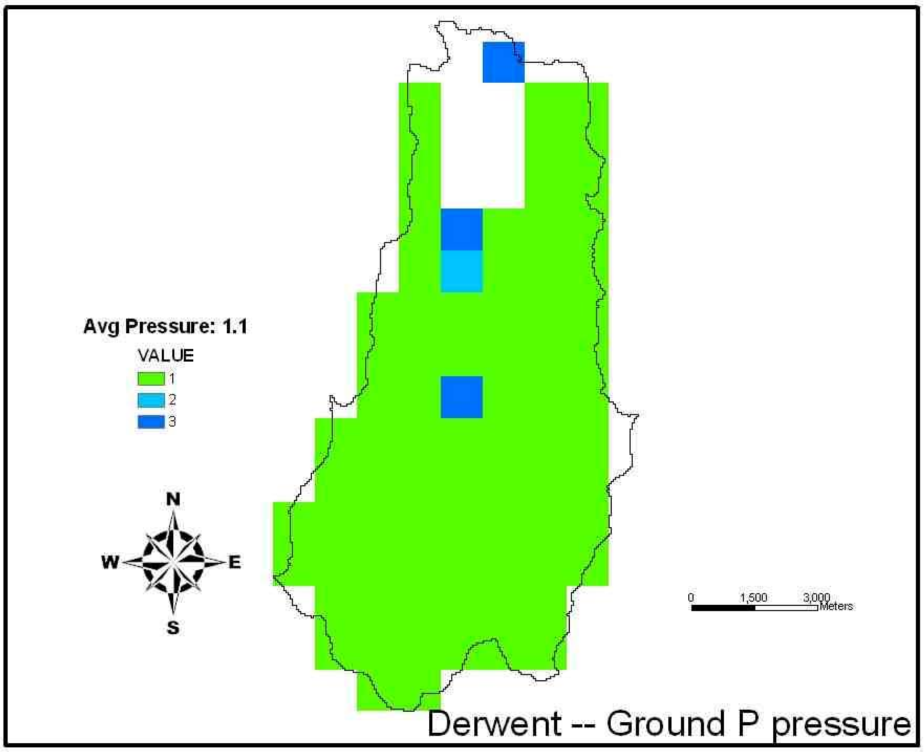


Figure A5.2. 2 Subsurface and surface P pressure maps for the Derwent Water catchment using the tier 2 approach



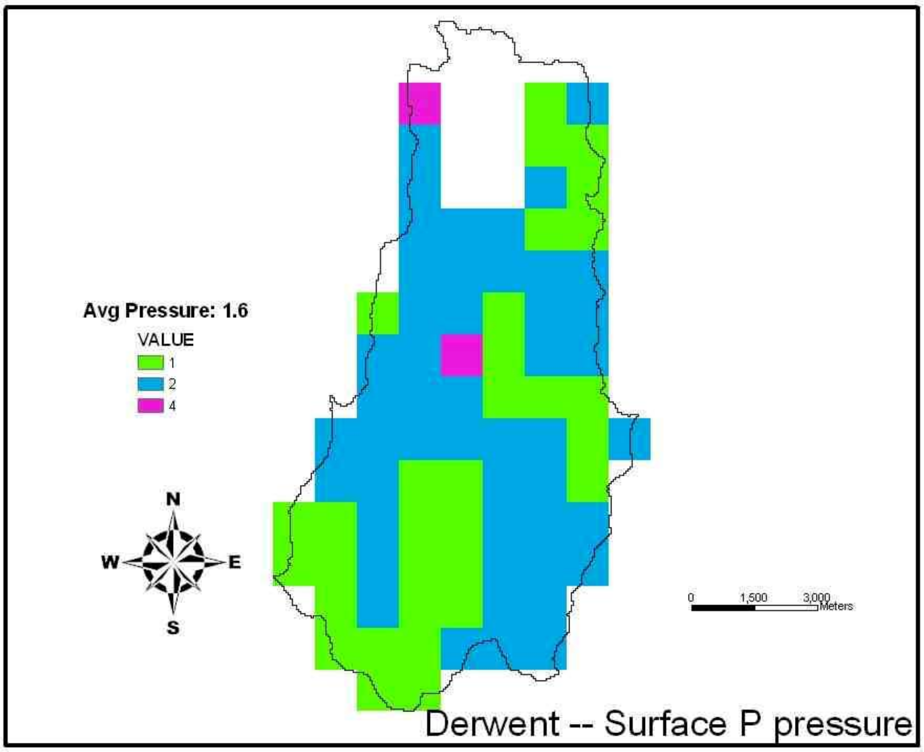
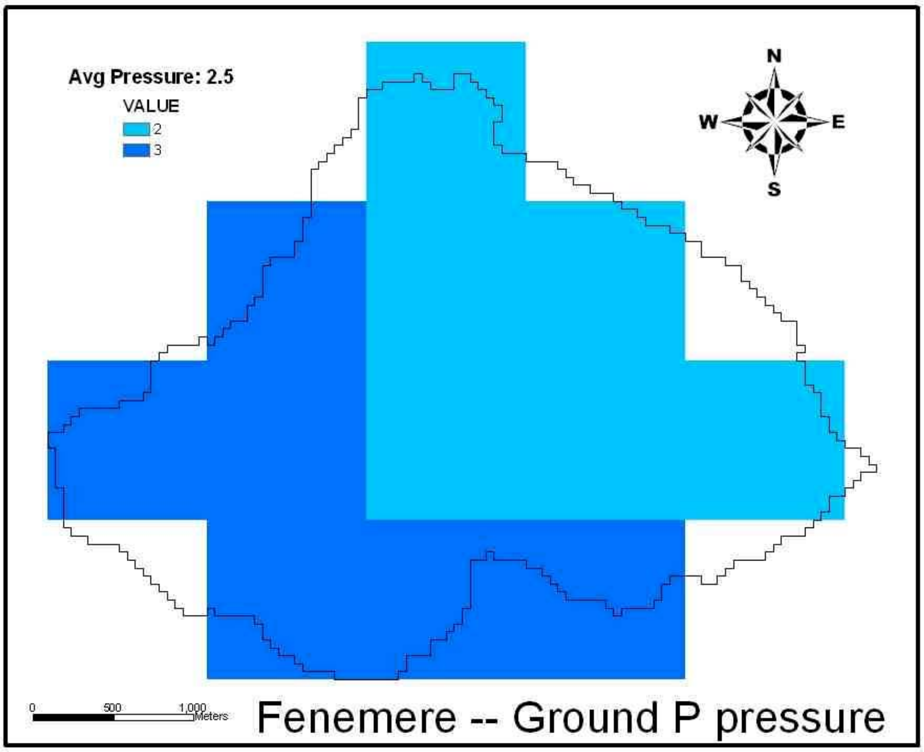


Figure A5.2. 3 Subsurface and surface P pressure maps for the Fenemere catchment using the tier 2 approach



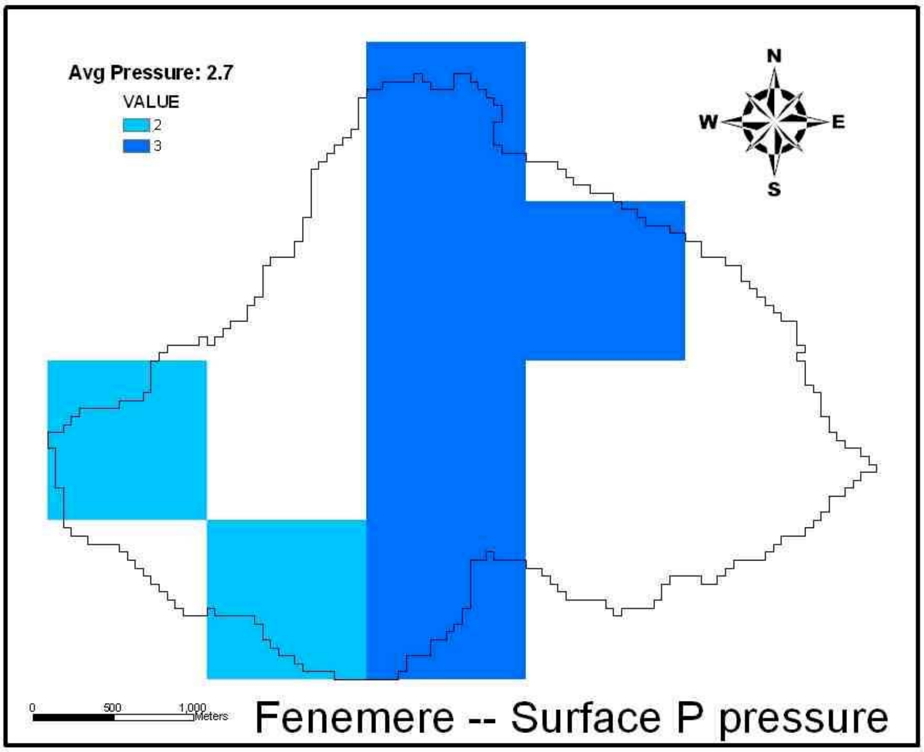
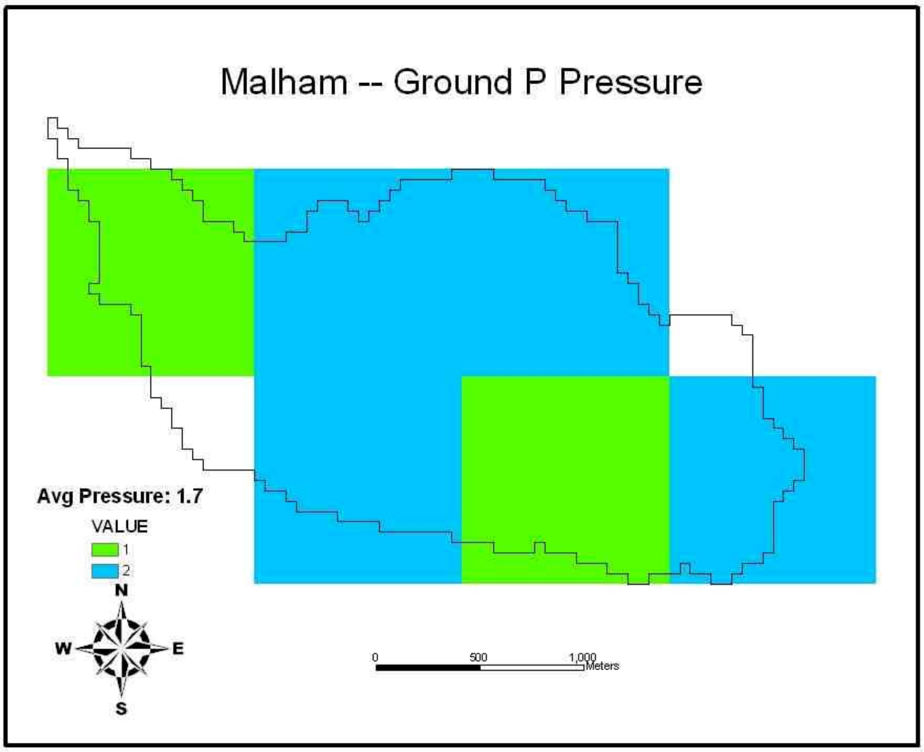


Figure A5.2. 4 Subsurface and surface P pressure maps for the Malham Tarn catchment using the tier 2 approach



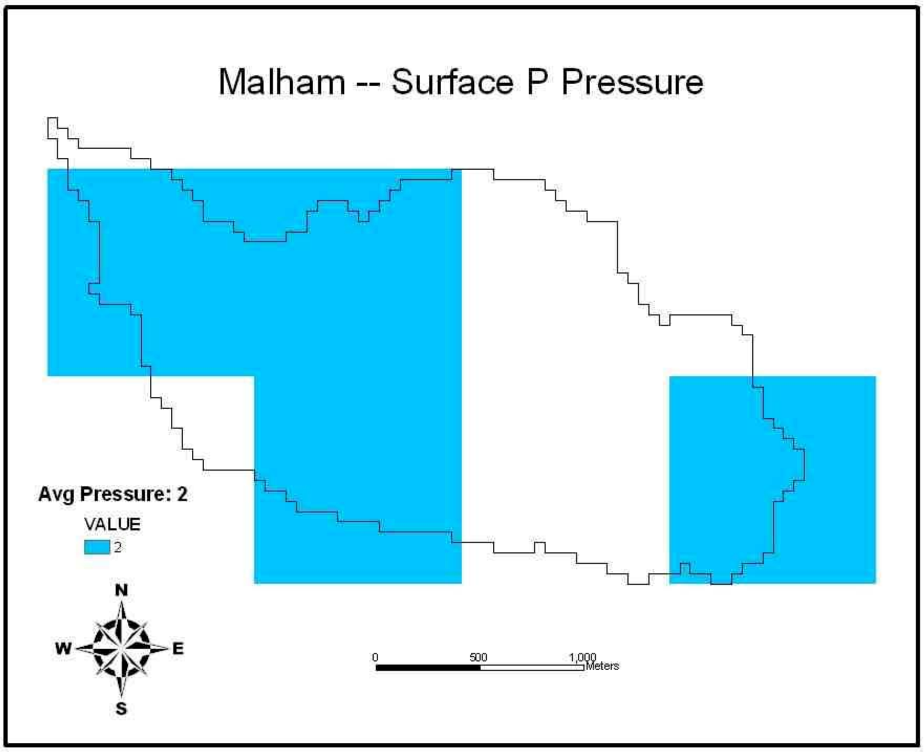
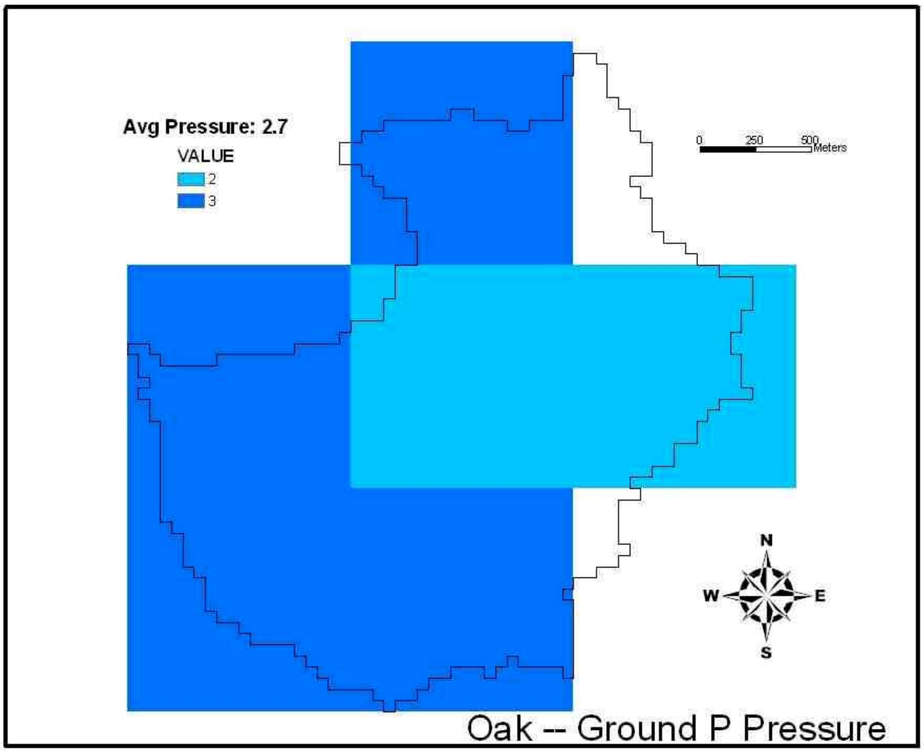


Figure A5.2. 5 Subsurface and surface P pressure maps for the Oak Mere catchment using the tier 2 approach



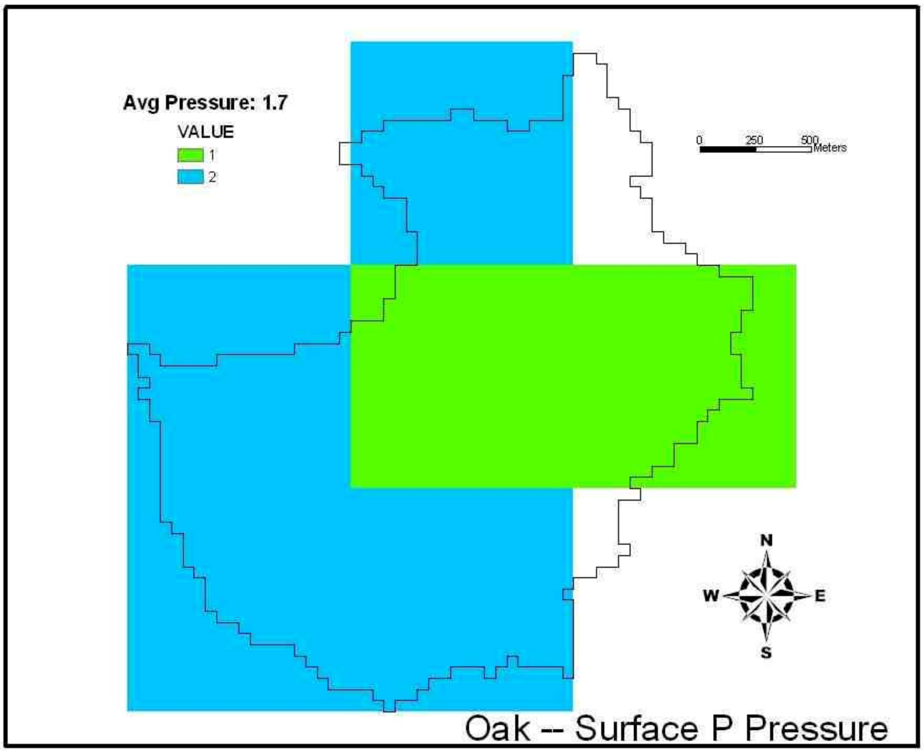
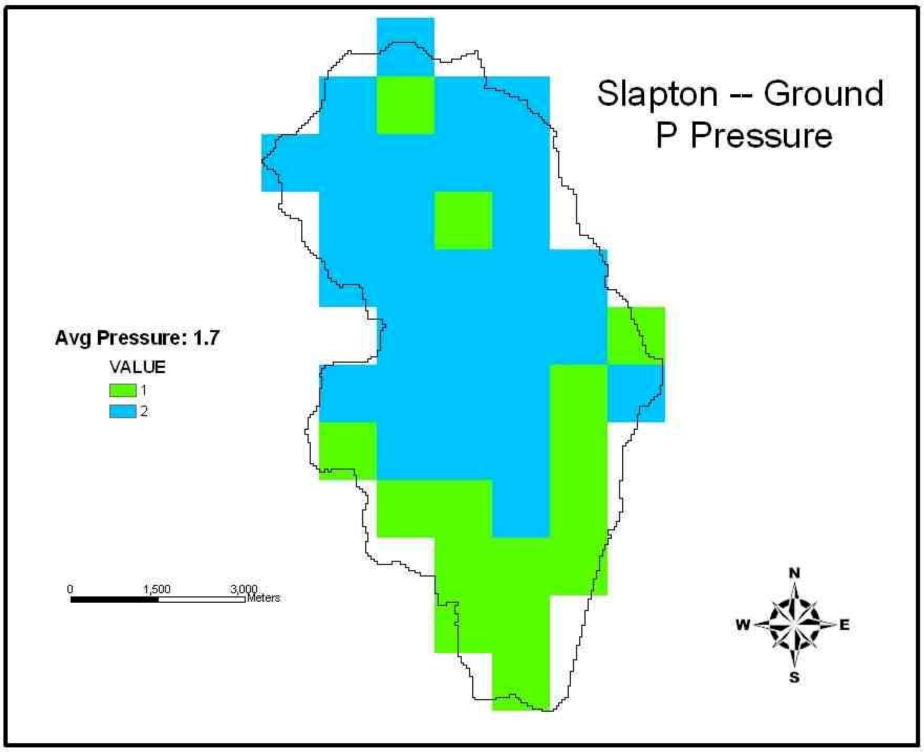


Figure A5.2. 6 Subsurface and surface P pressure maps for the Slapton Ley catchment using the tier 2 approach



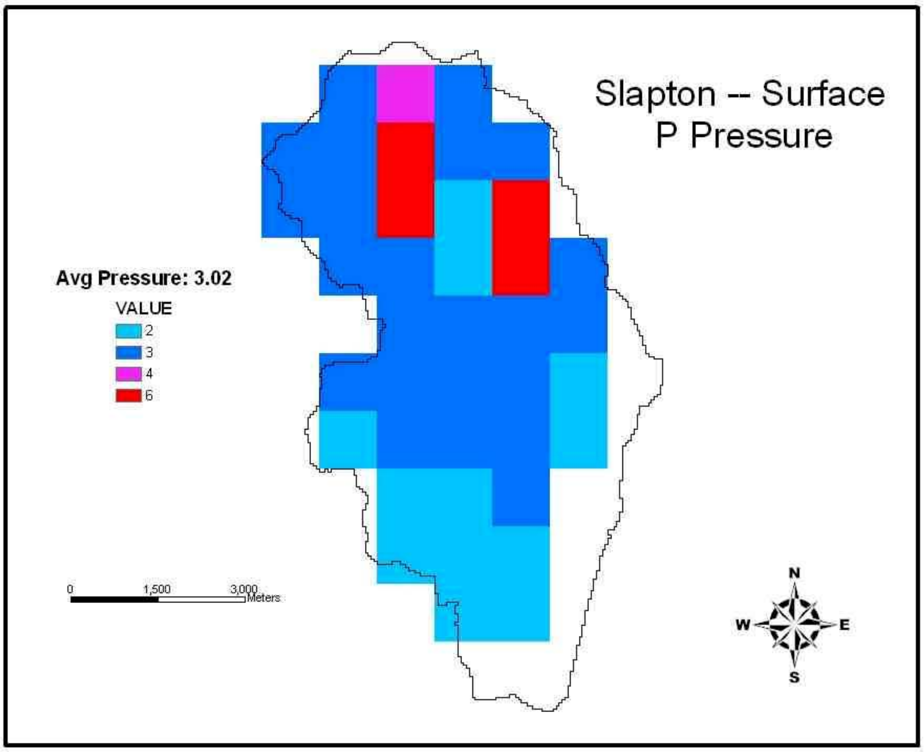
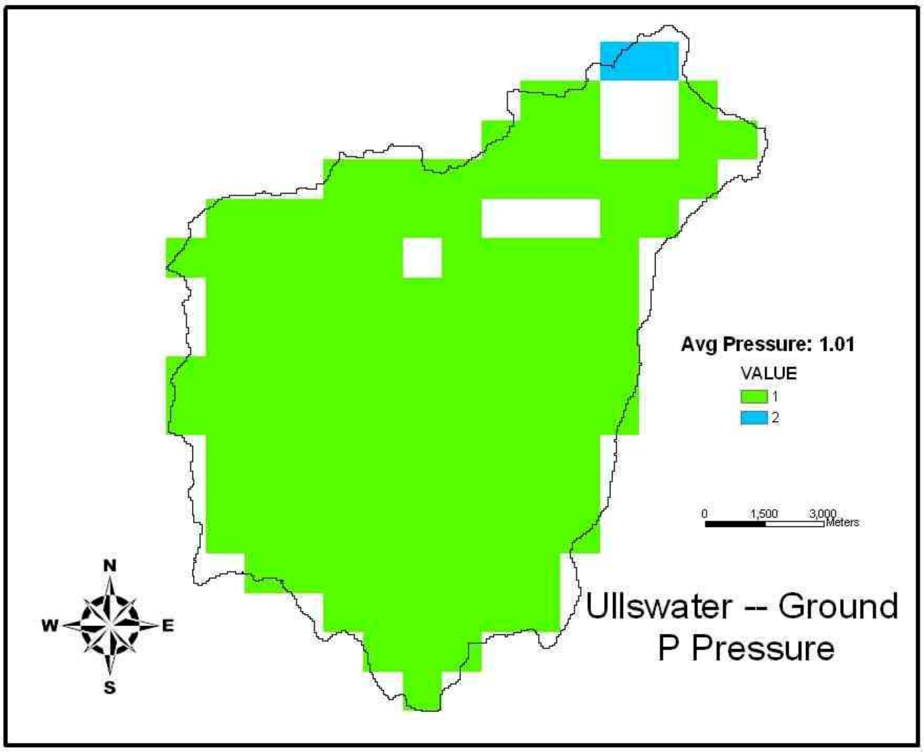


Figure A5.2. 7 Subsurface and surface P pressure maps for the Ullswater catchment using the tier 2 approach



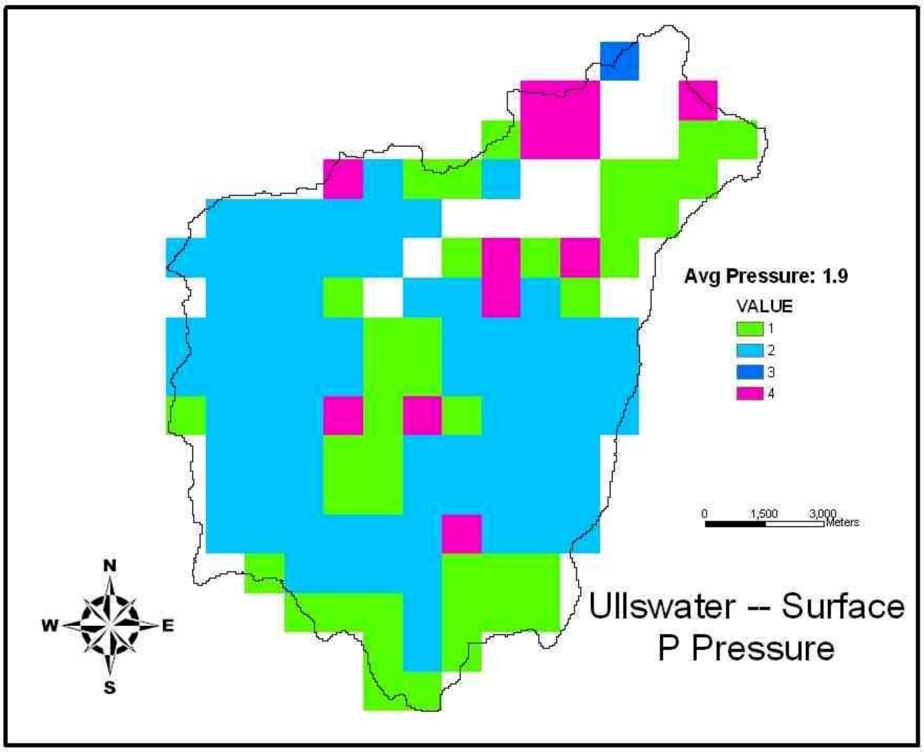
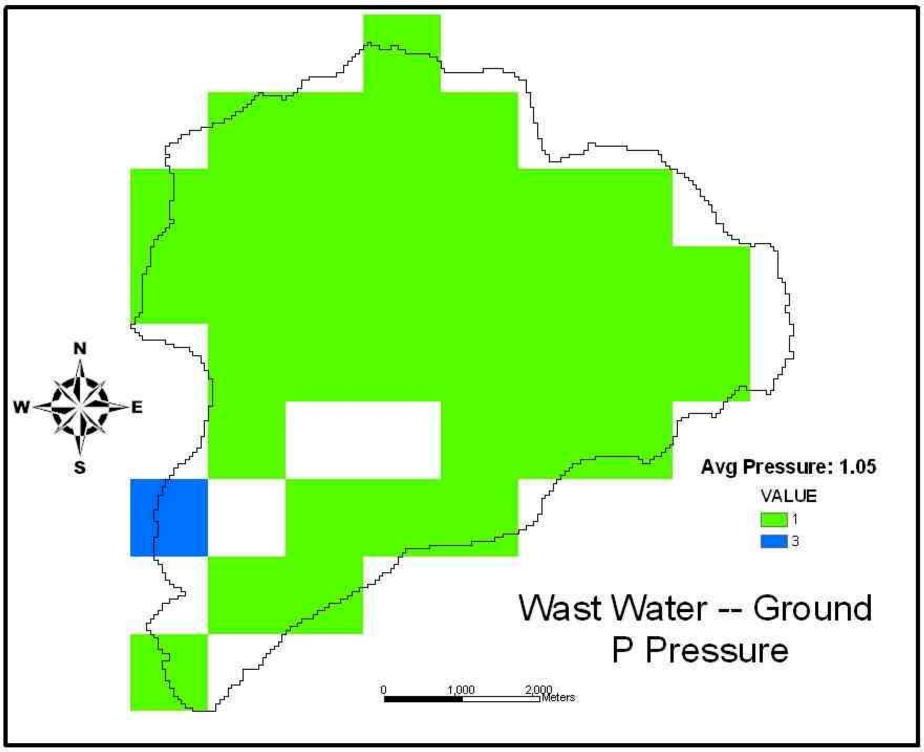
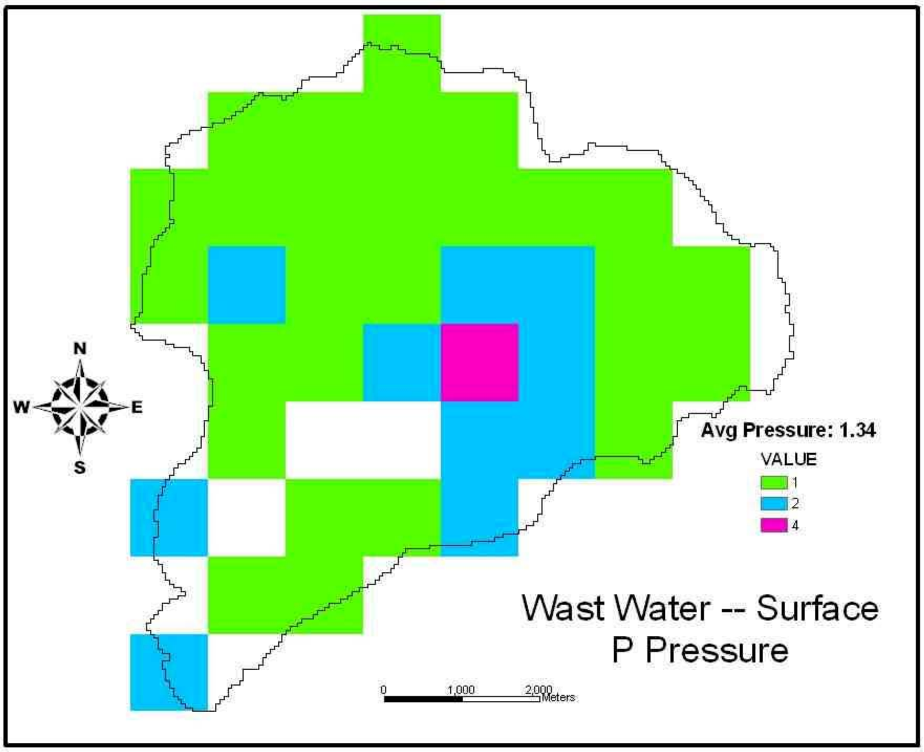


Figure A5.2. 8 Subsurface and surface P pressure maps for the Wast Water catchment using the tier 2 approach





A5. Appendix 5, Tier 3a

Figure A5.3AA. 1 Tier 3a results (PIT Silver standard) for Barton Broad

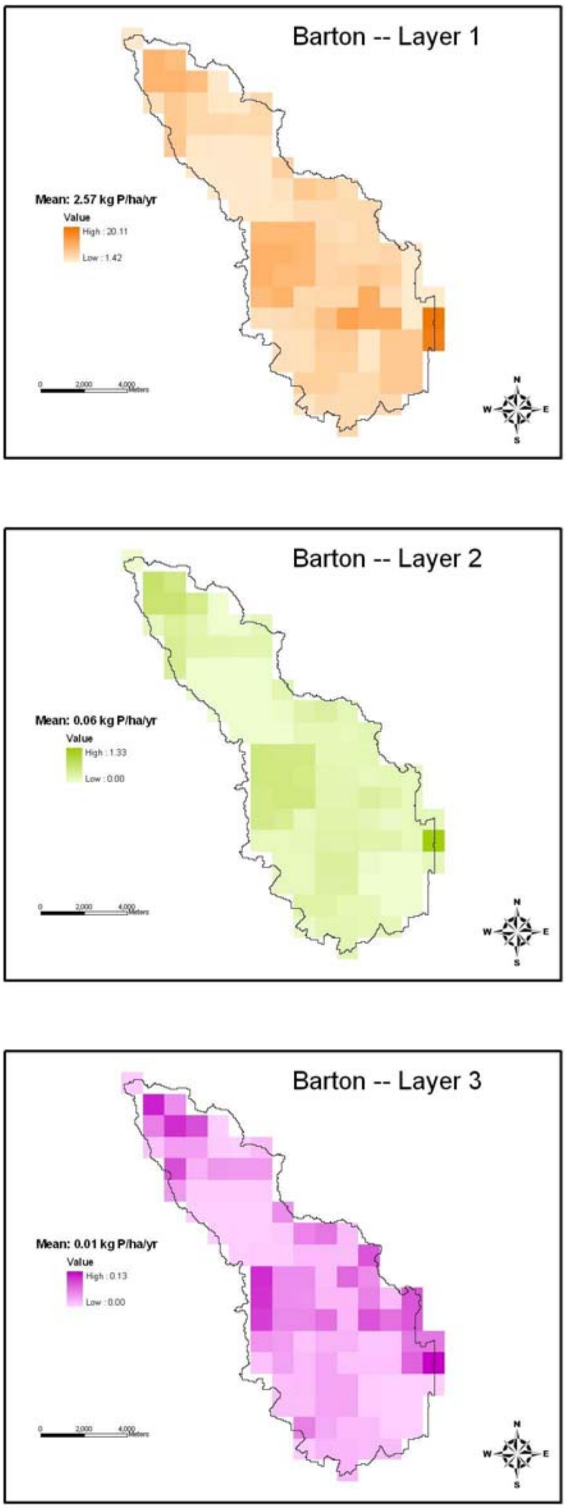


Figure A5.3A. 2 Tier 3a results (PIT Silver standard) for Bassenthwaite Lake

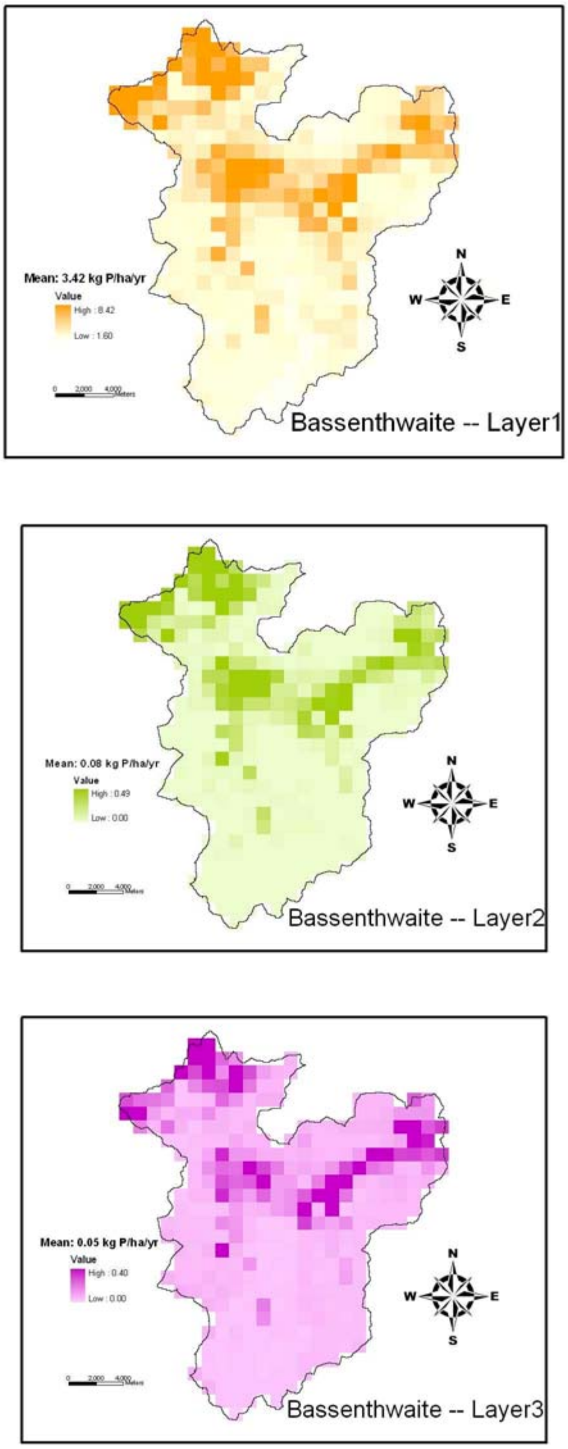
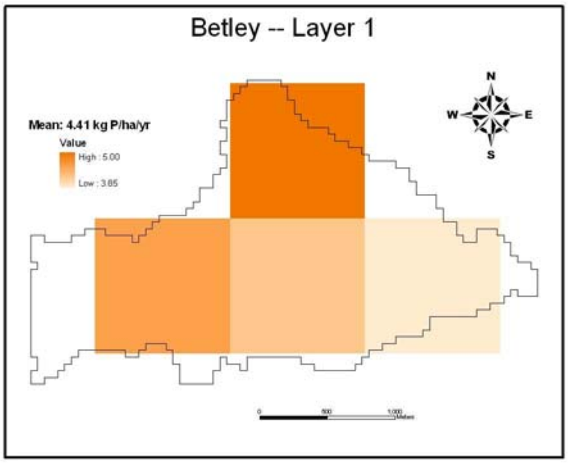
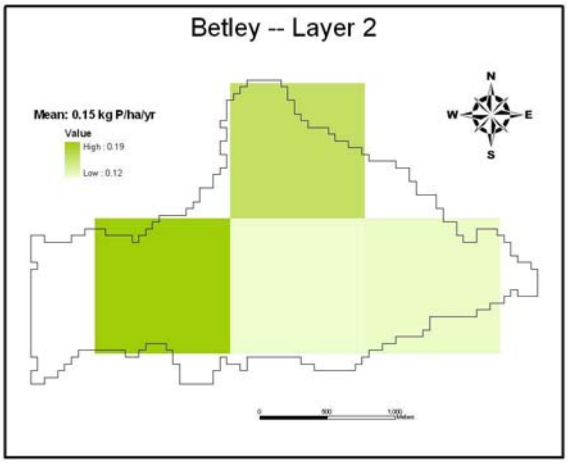


Figure A5.3A. 3 Tier 3a results (PIT Silver standard) for Betley Mere





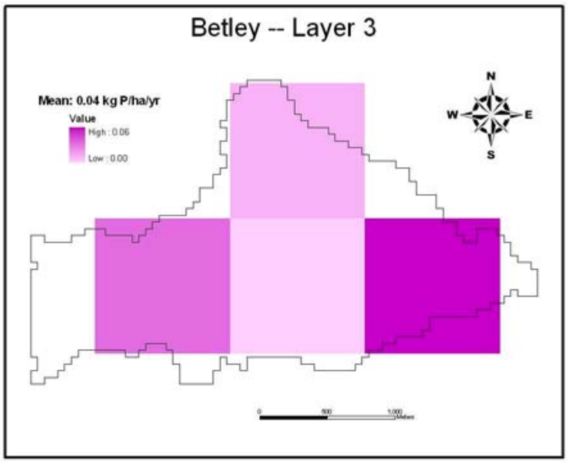


Figure A5.3A. 4 Tier 3a results (PIT Silver standard) for Blelham Tarn

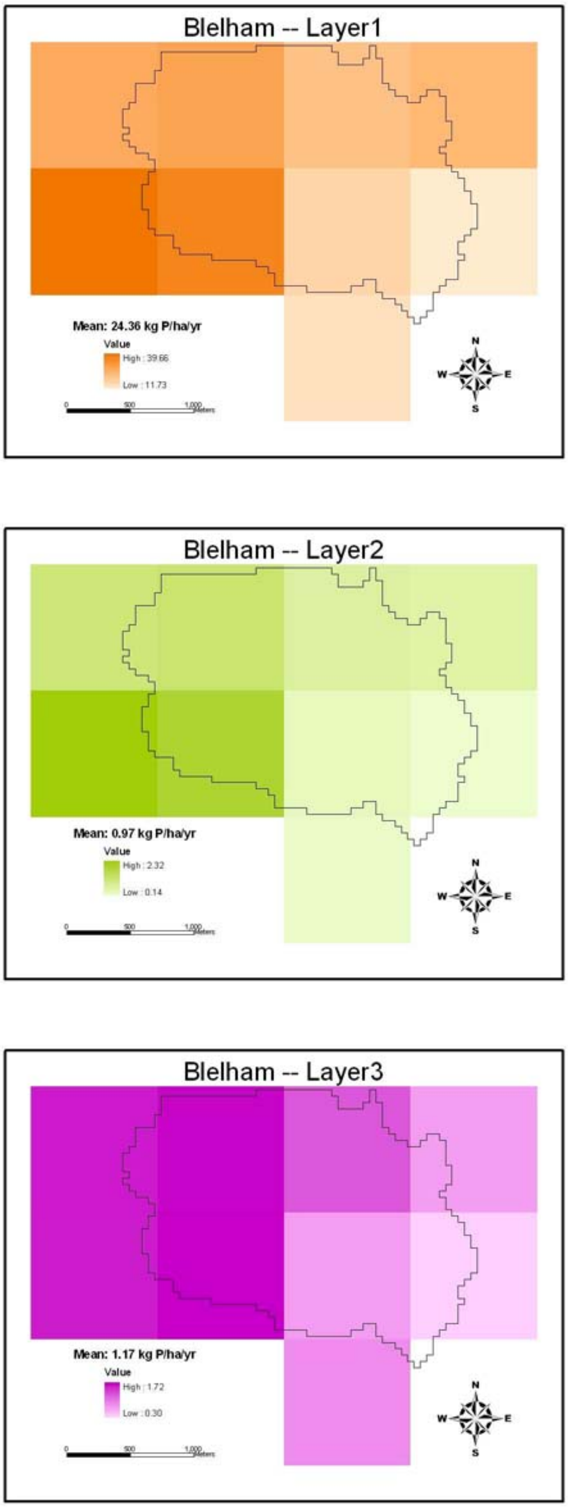


Figure A5.3A. 5 Tier 3a results (PIT Silver standard) for Comber Mere

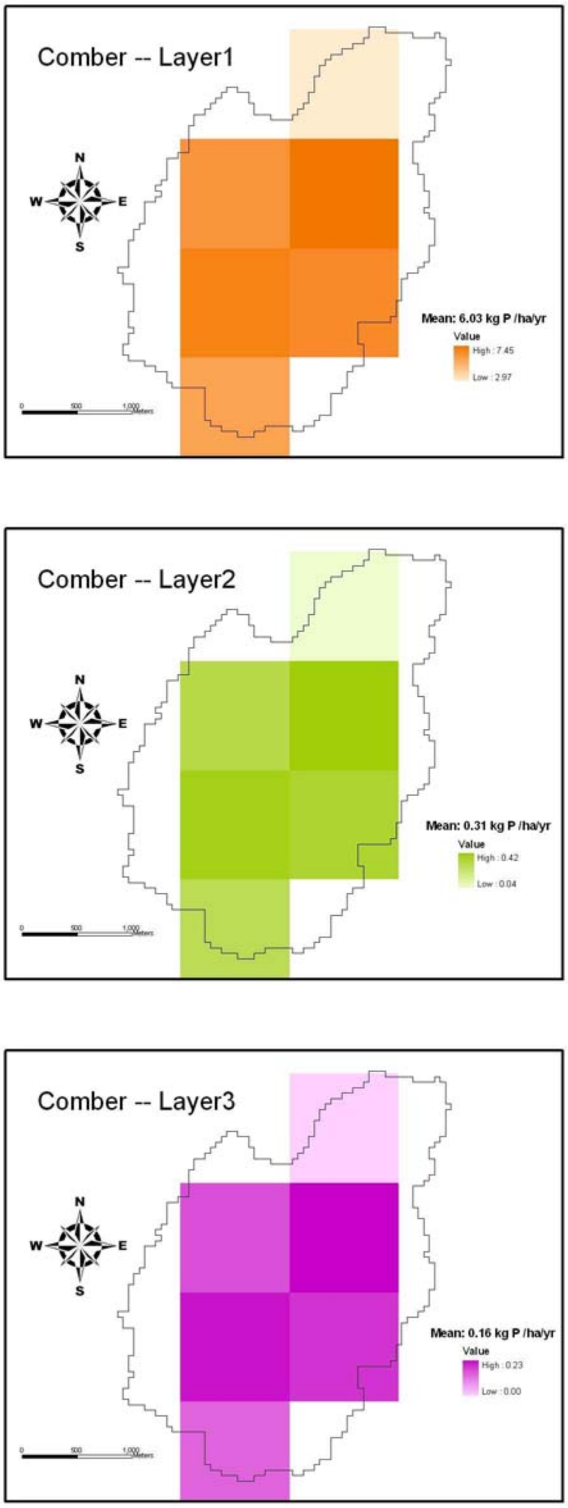


Figure A5.3A. 6 Tier 3a results (PIT Silver standard) for Coniston Water

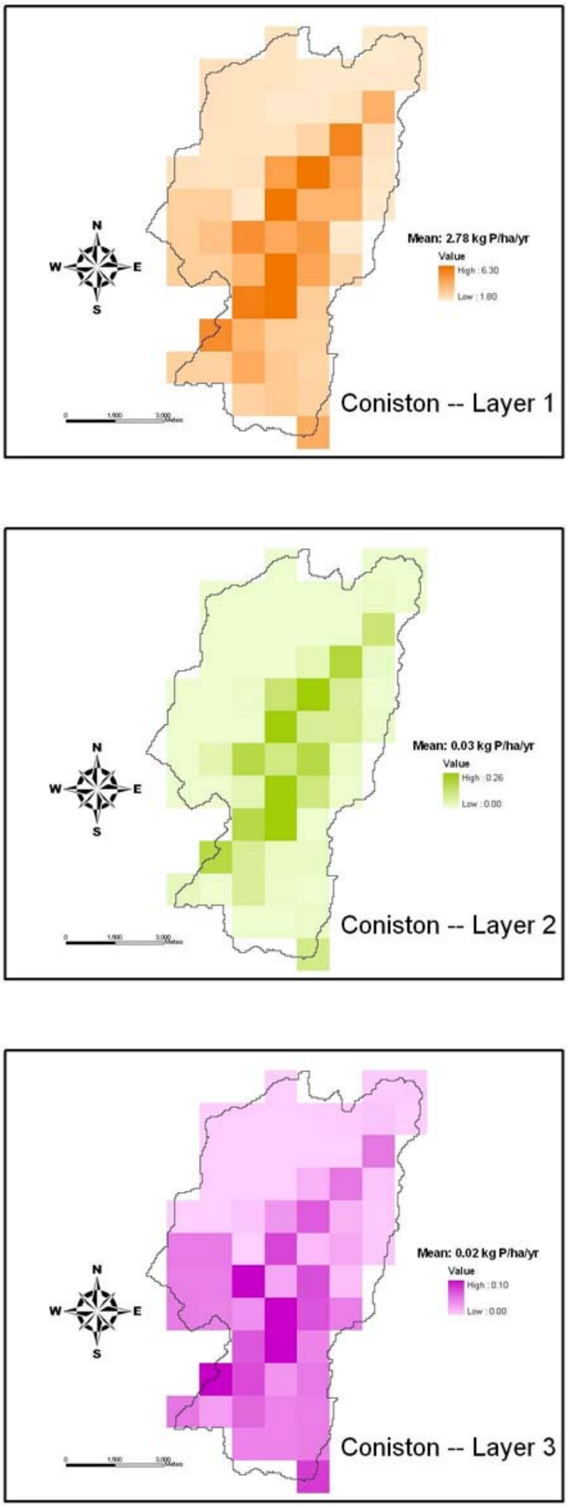


Figure A5.3A. 7 Tier 3a results (PIT Silver standard) for Llyn Coron

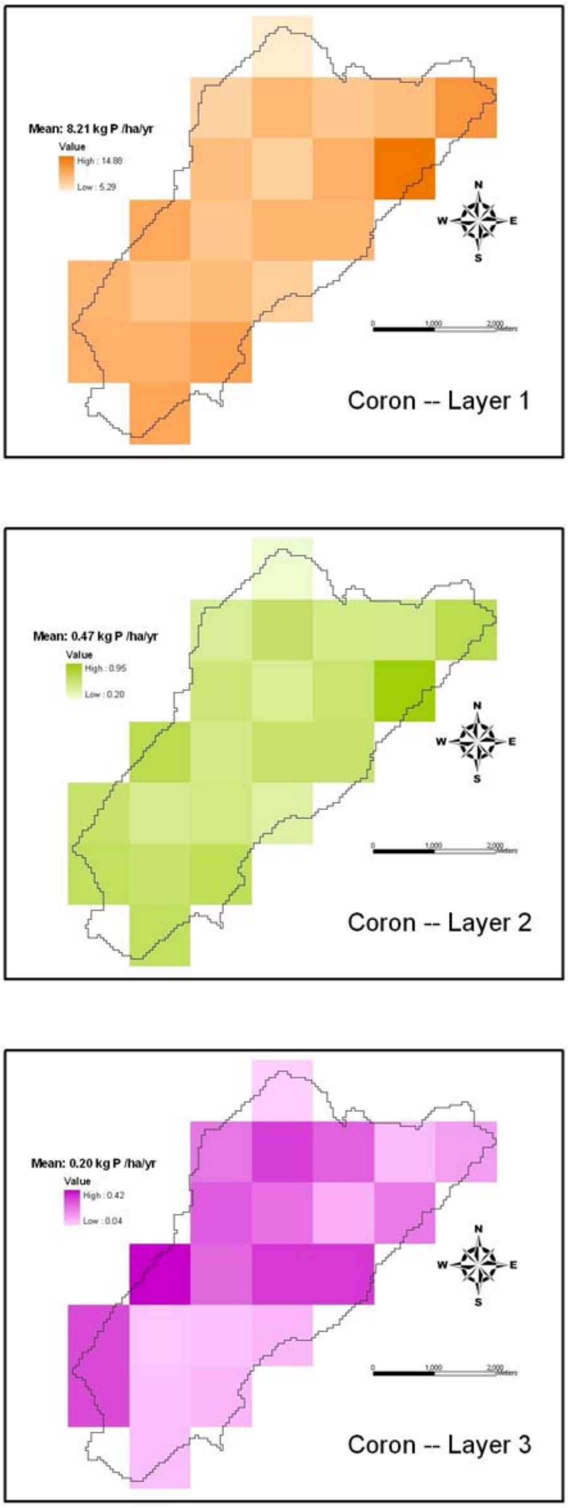


Figure A5.3A. 8 Tier 3a results (PIT Silver standard) for Derwent Water

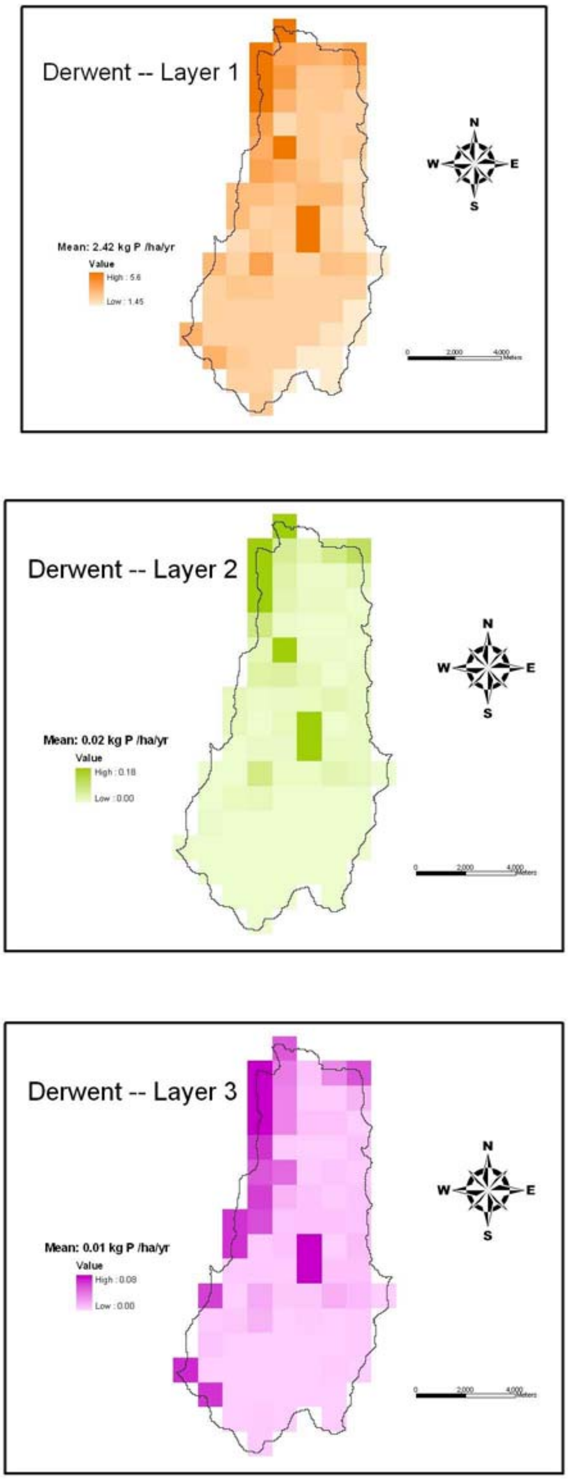


Figure A5.3A. 9 Tier 3a results (PIT Silver standard) for Esthwaite Water

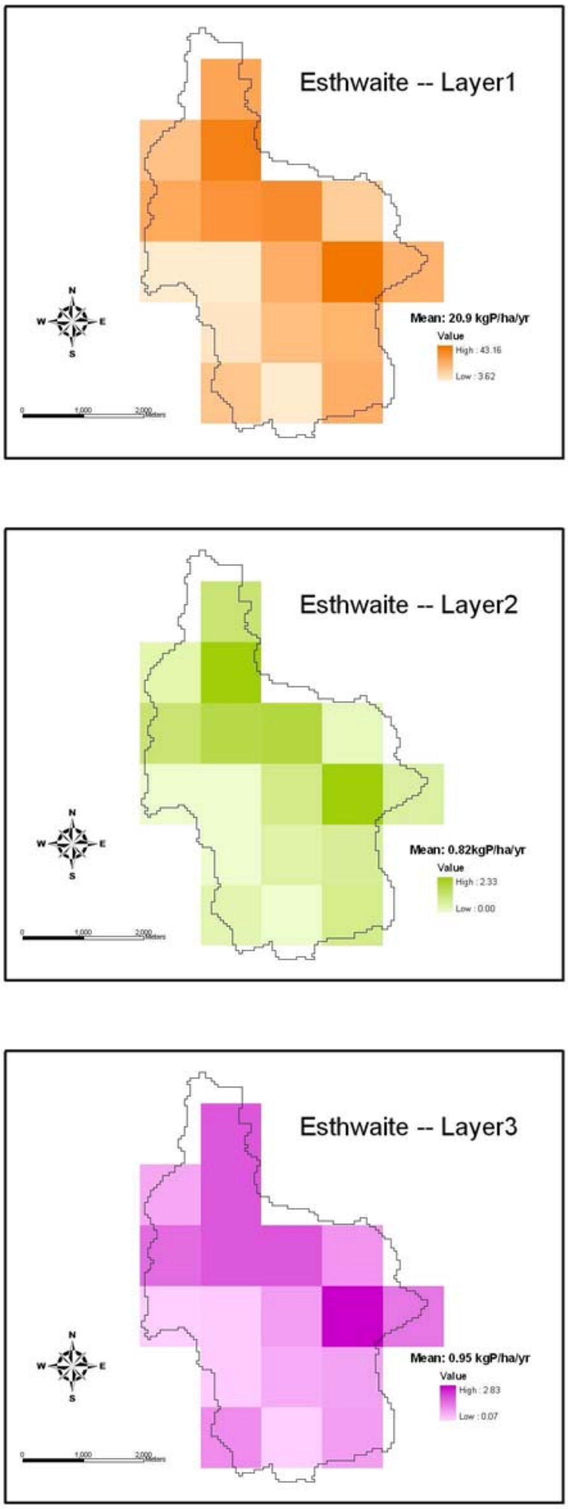
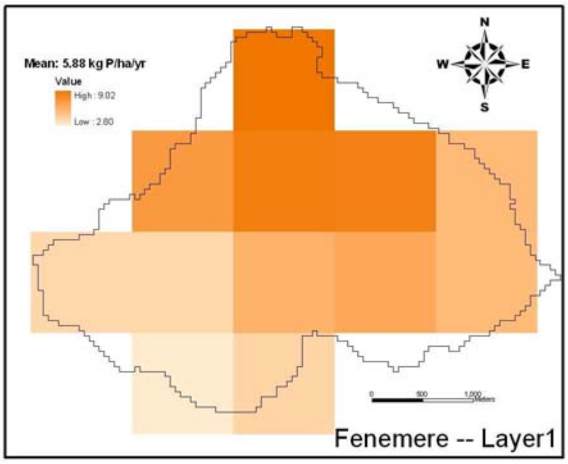
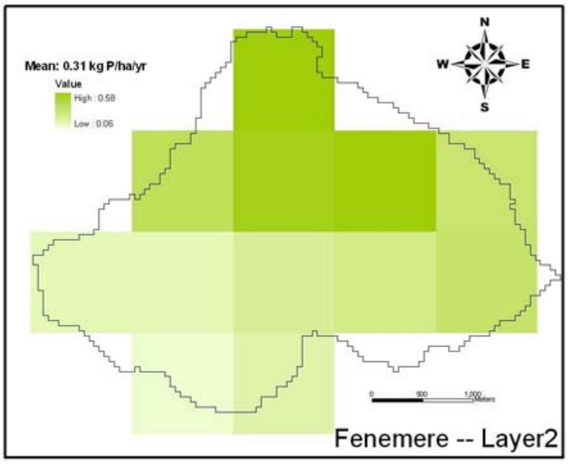


Figure A5.3A. 10 Tier 3a results (PIT Silver standard) for Fenemere





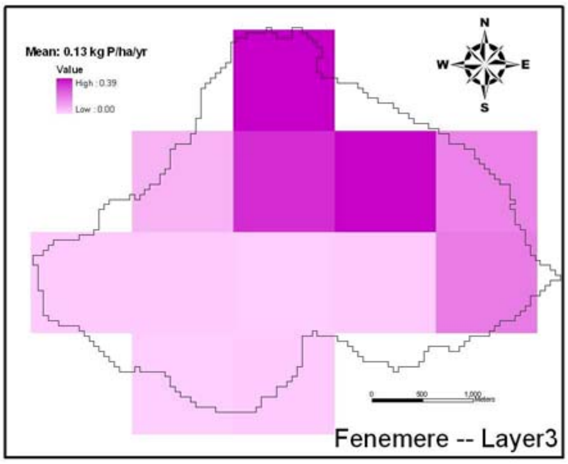


Figure A5.3A. 11 Tier 3a results (PIT Silver standard) for Llangorse Lake

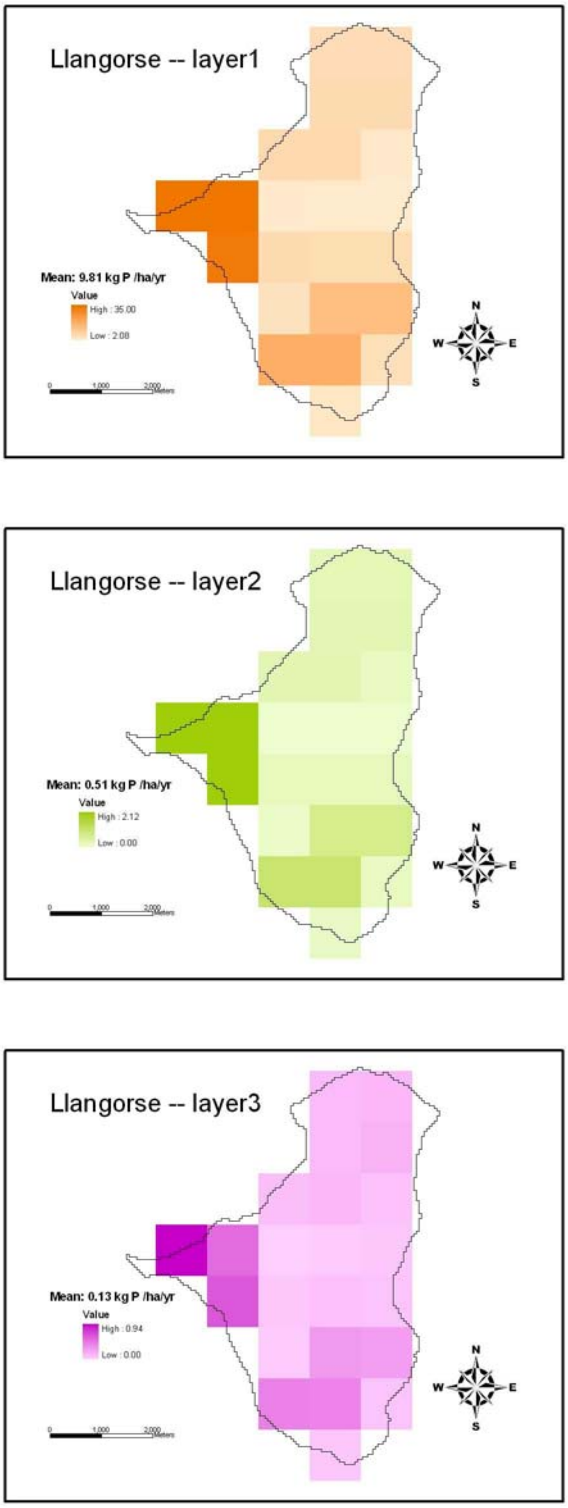


Figure A5.3A. 12 Tier 3a results (PIT Silver standard) for Loch Leven

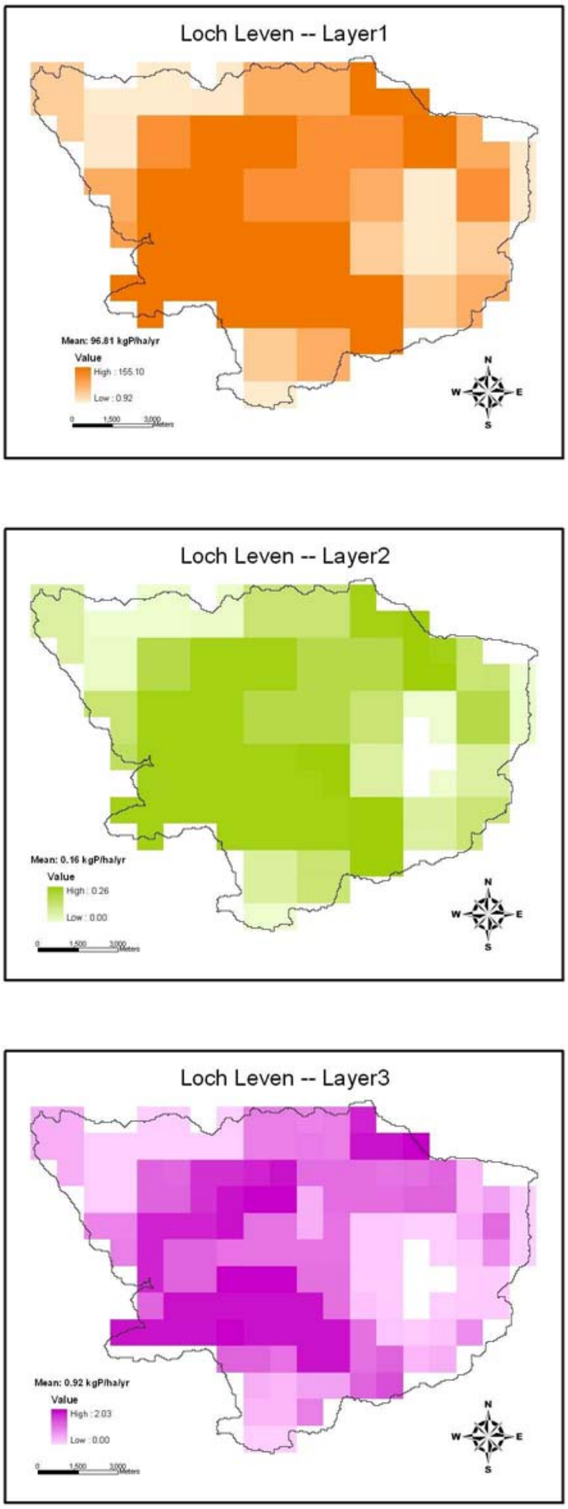
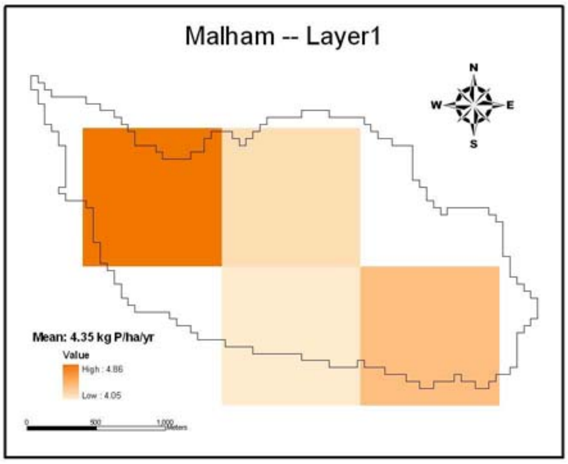
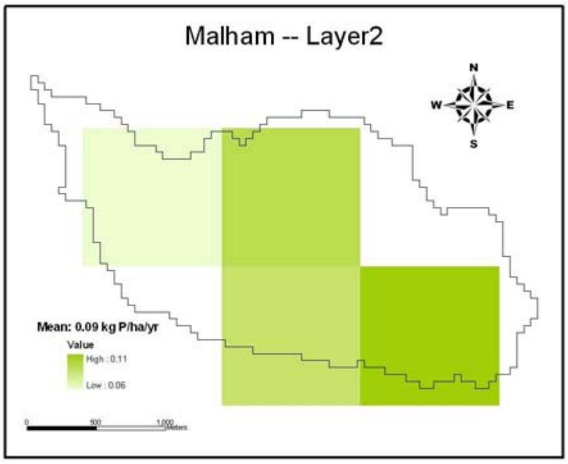


Figure A5.3A. 13 Tier 3a results (PIT Silver standard) for Malham Tarn





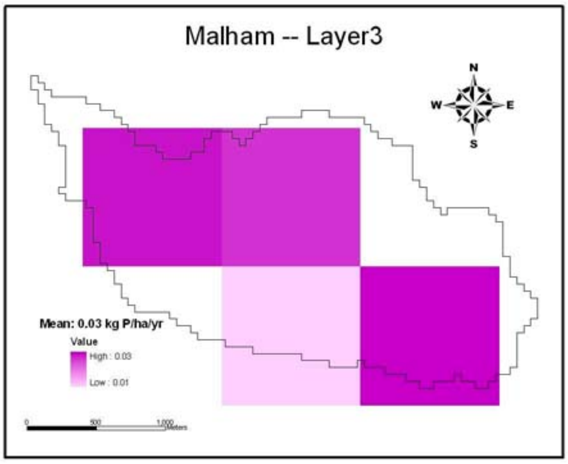
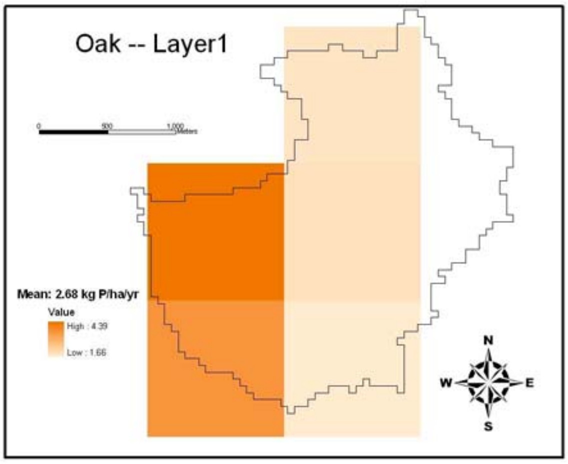
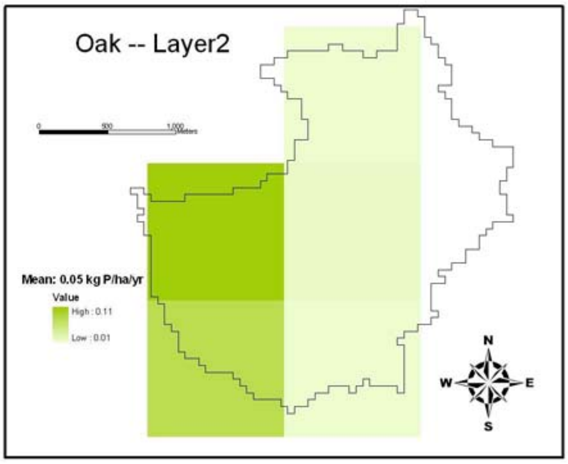


Figure A5.3A. 14 Tier 3a results (PIT Silver standard) for Oak Mere





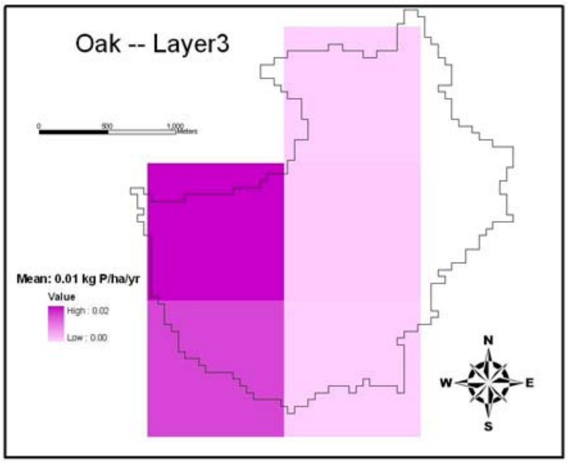


Figure A5.3A. 15 Tier 3a results (PIT Silver standard) for Rostherne Mere

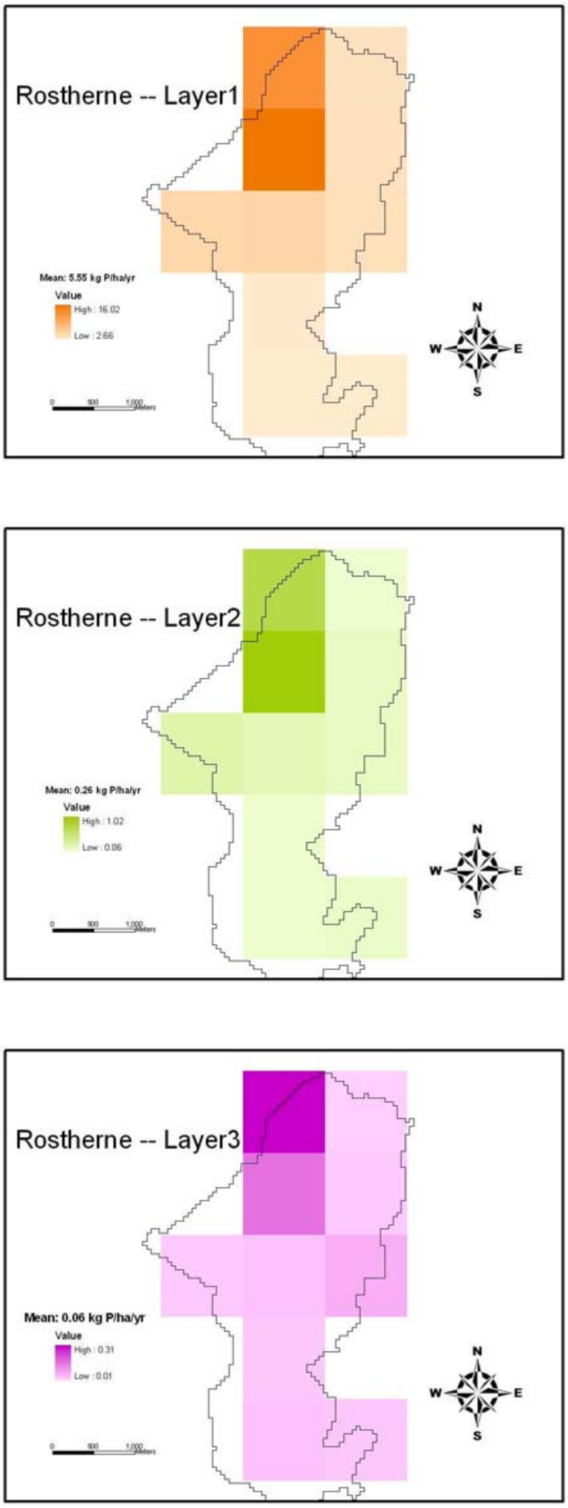
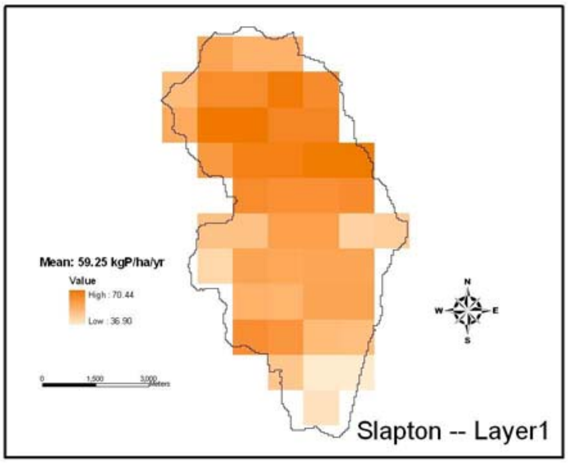
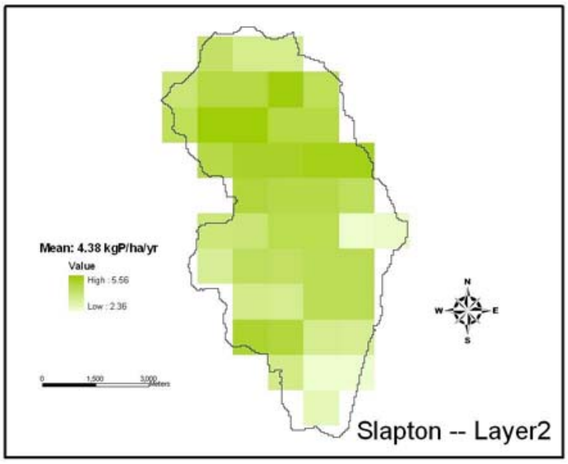


Figure A5.3A. 16 Tier 3a results (PIT Silver standard) for Slapton Ley





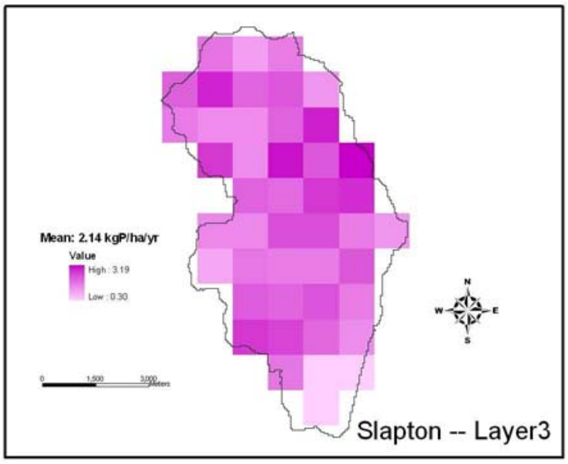


Figure A5.3A. 17 Tier 3a results (PIT Silver standard) for Llyn Tegid

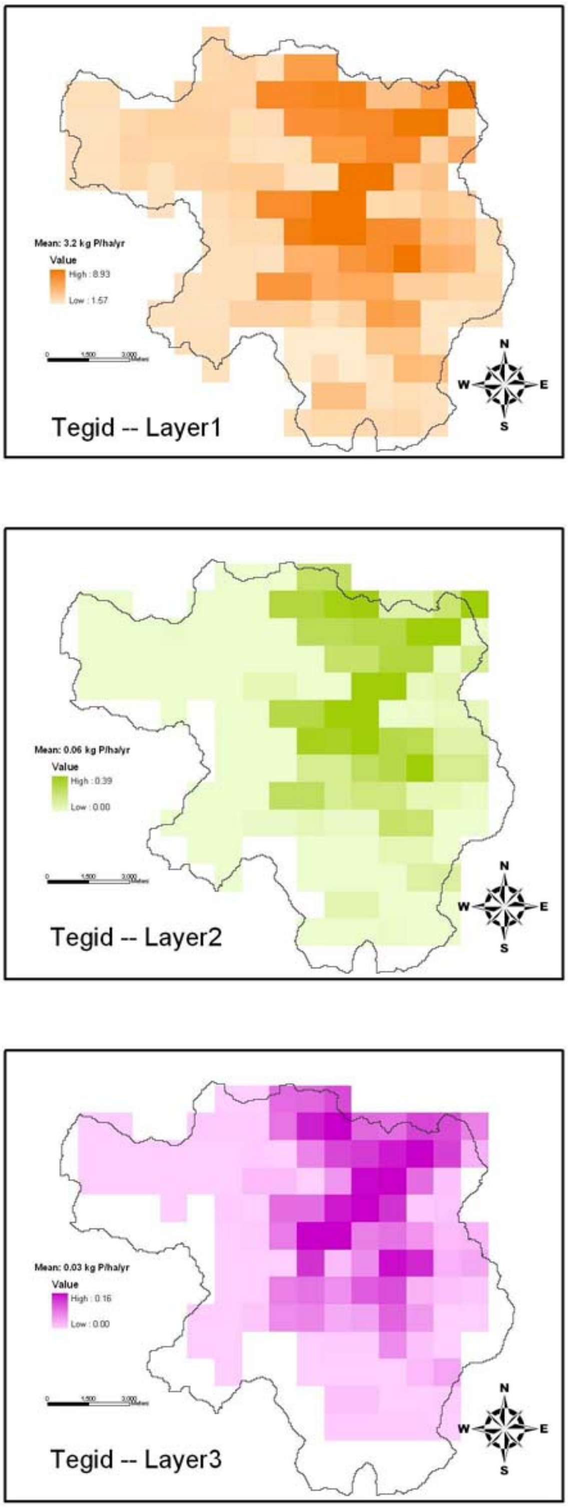


Figure A5.3A. 18 Tier 3a results (PIT Silver standard) for Ullswater

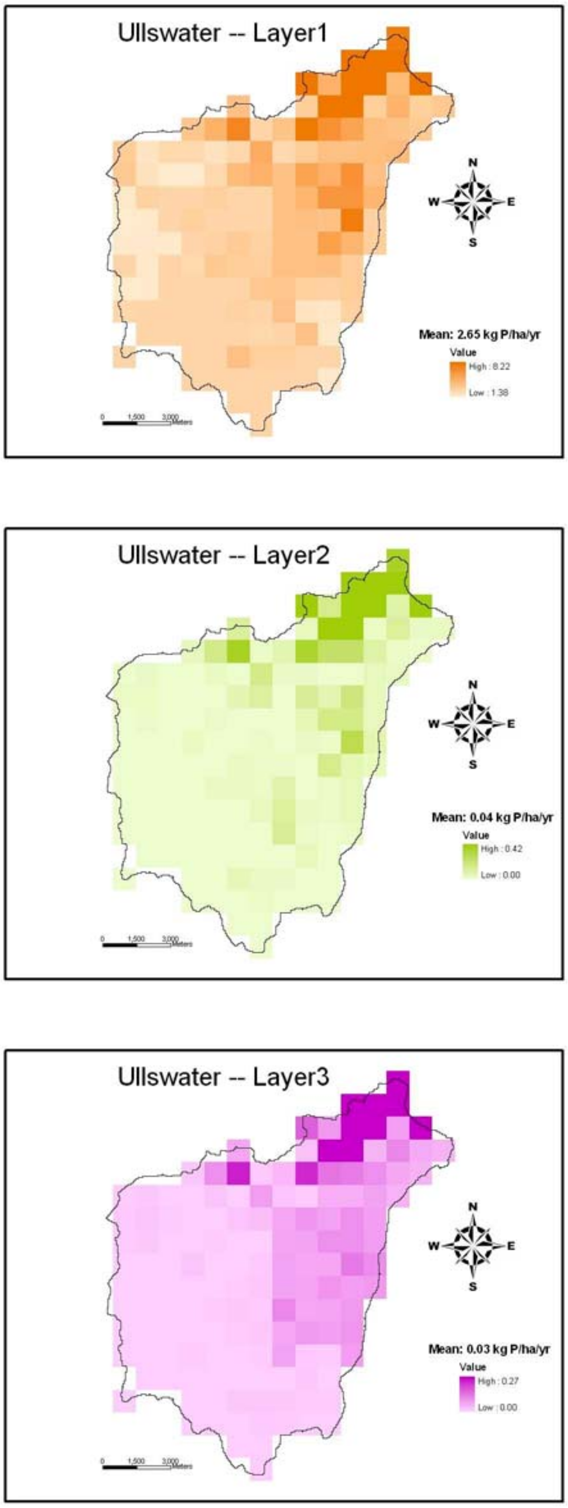


Figure A5.3A. 19 Tier 3a results (PIT Silver standard) for Wast Water

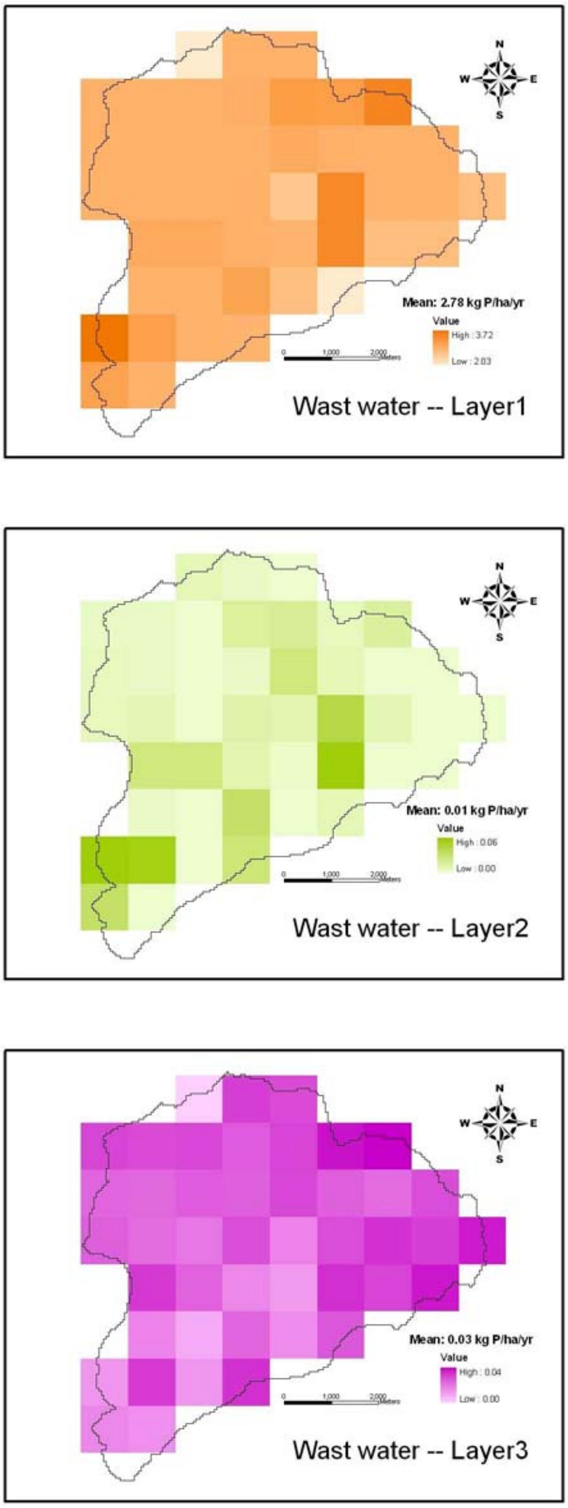
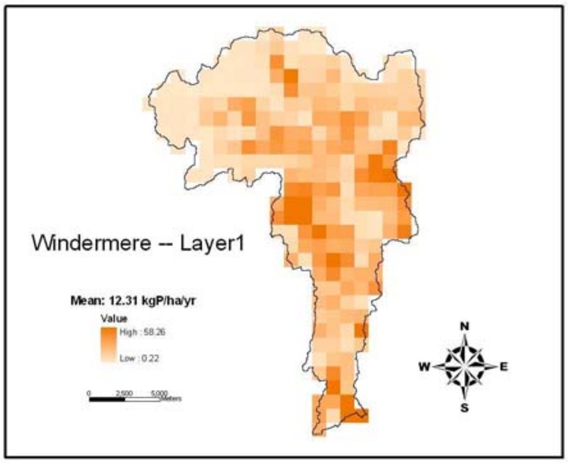
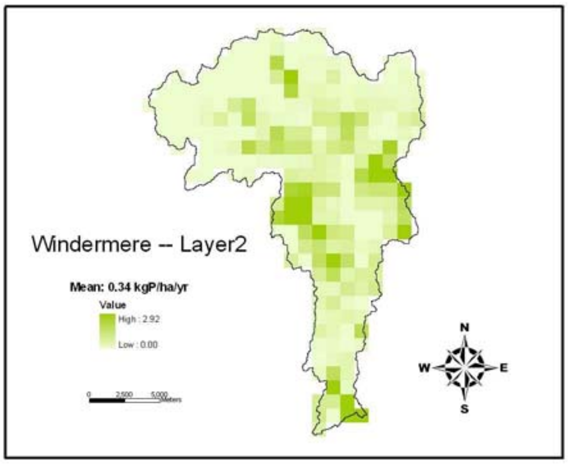
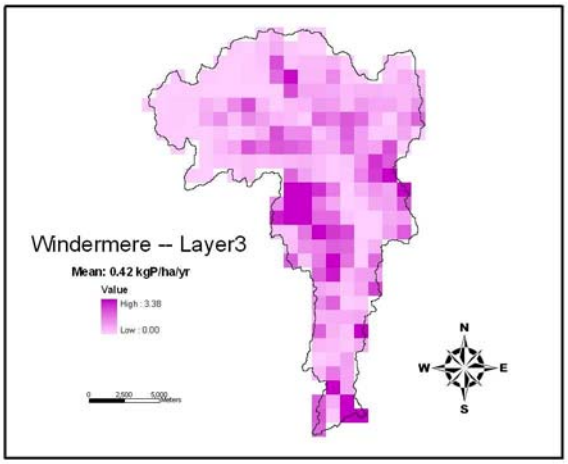


Figure A5.3A. 20 Tier 3a results (PIT Silver standard) for Windermere

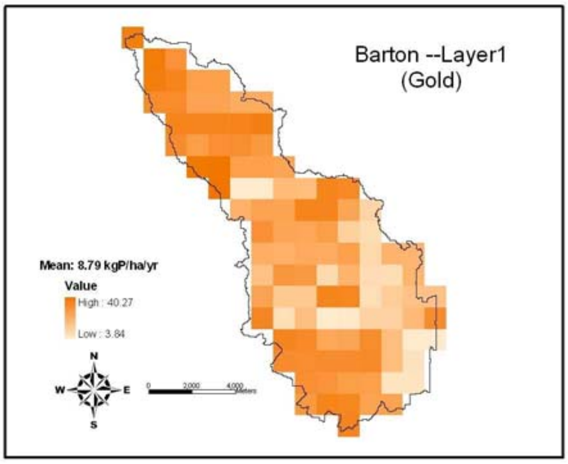


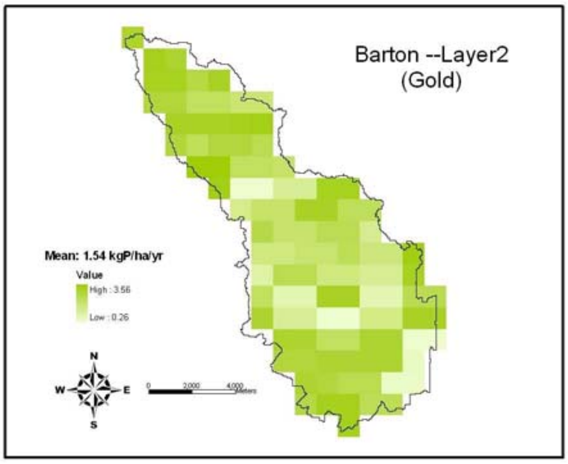




A5. Appendix 5, Tier 3b

Figure A5.3B. 1 Tier 3b results (PIT Gold standard) for Barton Broad





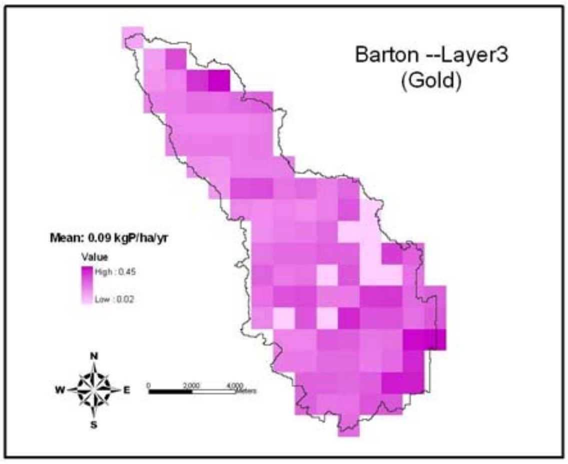
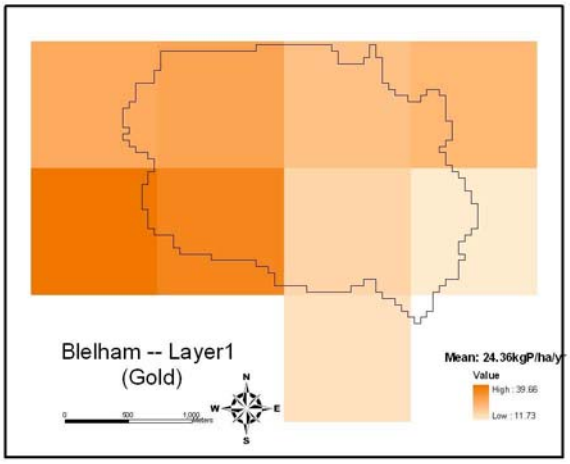
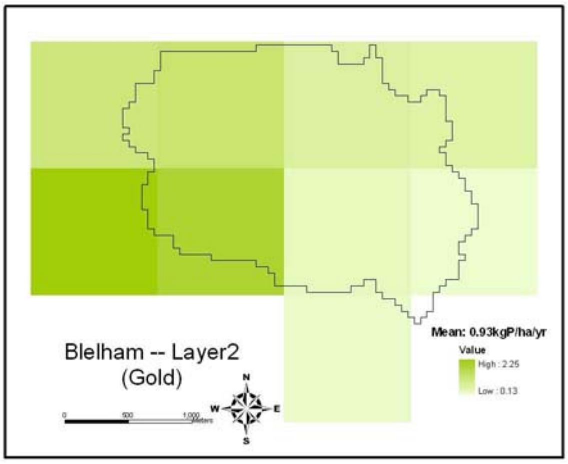


Figure A5.3B. 2 Tier 3b results (PIT Gold standard) for Blelham Tarn





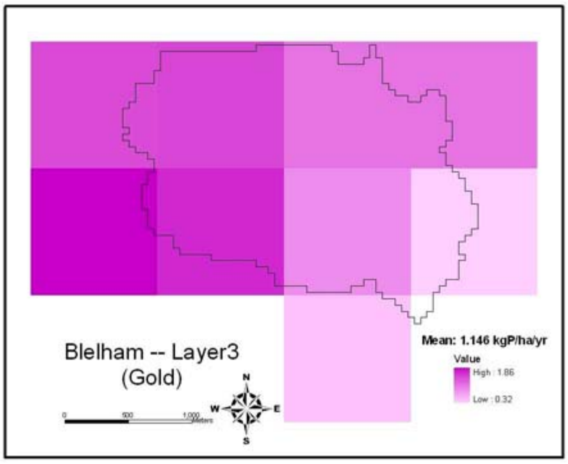
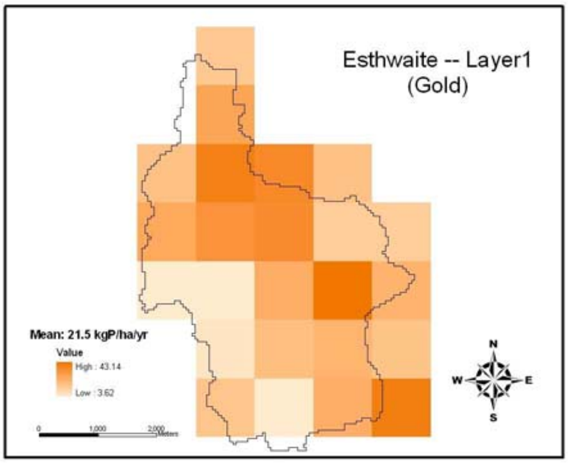
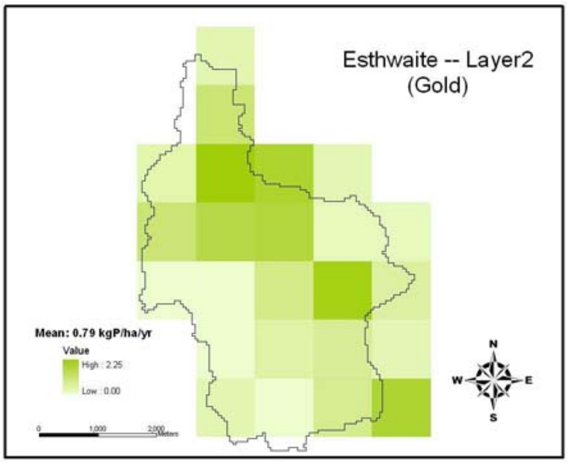


Figure A5.3B. 3 Tier 3b results (PIT Gold standard) for Esthwaite Water





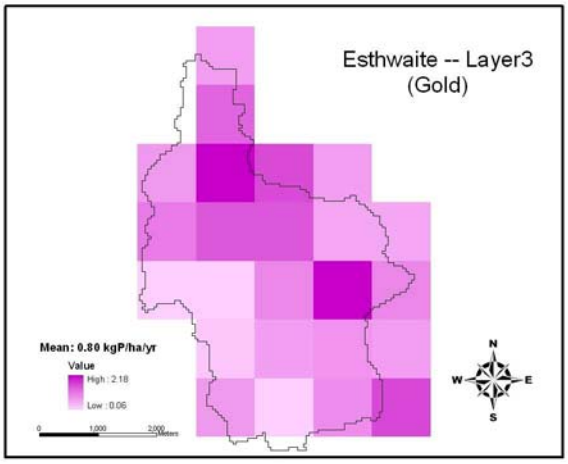


Figure A5.3B. 4 Tier 3b results (PIT Gold standard) for Slapton Ley

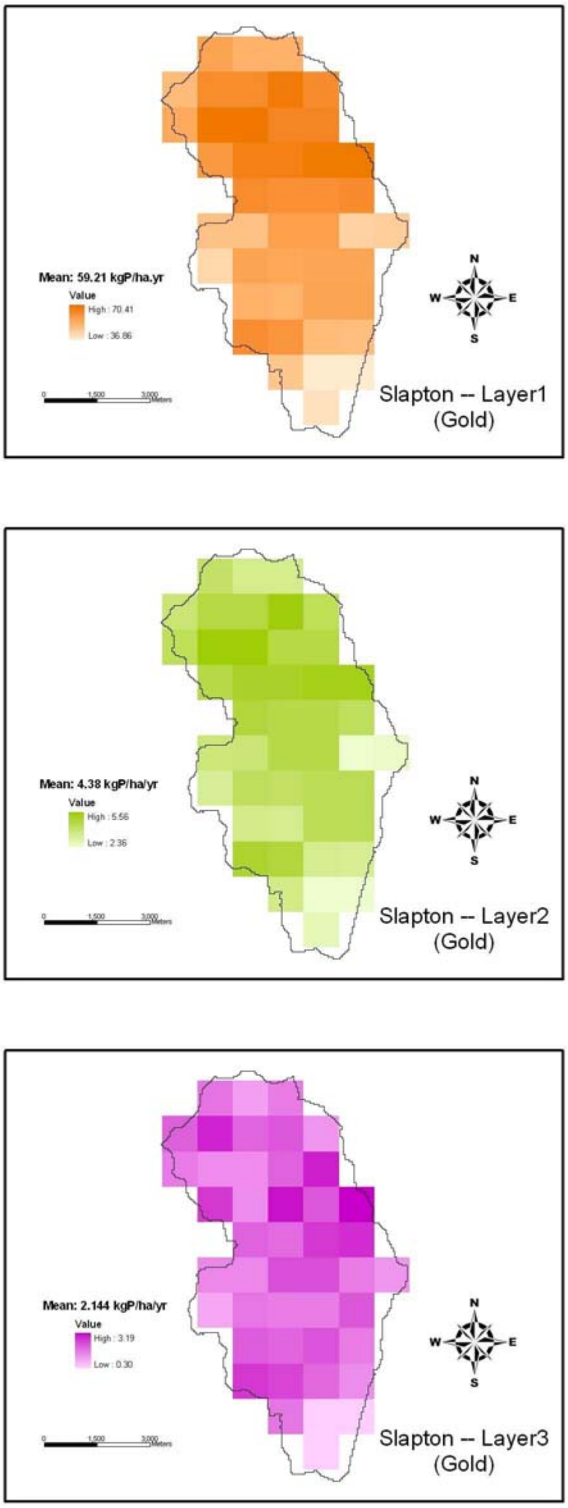


Figure A5.3B. 5 Tier 3b results (PIT Gold standard) for Windermere