Final Report Defra Project WT0750CSF

Identifying the Gap to Meet the Water Framework Directive - Lakes Baseline

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Catchment Sensitive Farming
and Water Quality Science

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1. Introduction

This report summarises technical work for Defra's Water Quality Division (Catchment Sensitive Farming group) on estimating the phosphorus loading and ecological status of freshwater standing waters throughout England and Wales. The work links to and builds upon the results of the Defra projects ES0205: Revised Diffuse Pollution Inventory; WQ0106: Underpinning Evidence and New Model Frameworks for Mitigation Multiple Diffuse Water Pollutants from Agriculture; and WT0719CSF: Identifying the Gap to Meet Water Framework Directive and Best Policies to Close the Gap.

The project aims to estimate annual average concentrations of total phosphorus in standing waters from estimates of the total phosphorus load from all diffuse and point sources, including: diffuse runoff from roads and urban areas; leakage from septic tanks and sewage effluent discharges; direct atmospheric deposition; river bank erosion; and diffuse runoff and soil drainage from agricultural land. Predicted concentrations are compared with threshold values for Good Ecological Status (GES) to determine the required percentage reduction in phosphorus inputs to meet the objectives of the Water Framework Directive (WFD). These required reductions are compared to forecasts of future phosphorus loads from diffuse agricultural sources that take account of changes in land management and livestock numbers.

Sections 2 to 3 of this report describe the models used for calculating the point and diffuse source phosphorus loads input to the standing waters, and the prediction of annual average total phosphorus concentrations. Section 4 reports on the validation of the combined models against recent measurements of total phosphorus concentrations in a large number of water bodies. Section 5 presents the results of applying the models to all standing waters in England and Wales, under present day (year 2004) environment conditions and under a forecast of diffuse agricultural phosphorus loadings for year 2015 under a Business as Usual (BAU) scenario. The results from this work are presented as statistical summaries of the proportions of standing waters achieving Good Ecological Status, as defined under the Water Framework Directive. Section 6 concludes with a discussion of the results and conclusions.

1.1 Background

Defra's Water Quality Division (Catchment Sensitive Farming Group) is working towards identifying practical on-farm methods for mitigating diffuse water pollution from agriculture (DWPA). Amongst other activities, this has involved commissioning a review by ADAS and IGER of diffuse pollution losses from different farming systems (Defra Project ES0203) and the development of a methodology for calculating the cost and effectiveness of policy instruments in reducing diffuse pollution losses at farm and national scale (Defra Projects ES0205 and WQ0106). A policy instrument results in the voluntary or enforced adoption of a group of pollution mitigation measures on a farm. Examples of policy instruments are codes of good agricultural practice and legislation under the Nitrate Vulnerable Zones (NVZs), and examples of measures are riparian buffer zones and reduction in fertiliser inputs. This work has supported the overall aim of the Catchment Sensitive Farming (CSF) programme: to identify policy packages (combinations of policy instruments) for inclusion into the programme of measures in River Basin Management Plans under the Water Framework Directive (WFD). An important aspect of this work is establishing that policy packages will result in good chemical and ecological status for water bodies. Defra require quantification of the gap between current status of water bodies and good status, and the impact of policy packages in reducing this gap.

Phosphorus is an important diffuse agricultural pollutant, contributing to the risk of eutrophication of fresh waters (Reynolds and Davies, 2001). Lake water total phosphorus concentration is a strong predictor of numerous indicators of lake water quality and overall ecosystem composition, including water clarity, phytoplankton biomass, and the abundance and diversity of macrophytes and fish (Brett and Benjamin, 2008). Concentrations of total phosphorus in English lakes and soluble reactive phosphorus in rivers are typically greater than 100 $\mu g\ l^{-1}$ P and are indicative of potential eutrophic conditions (Foy and Bailey-Watts, 1998; Muscutt and Withers, 1996). It has been previously estimated that 82,000 t of phosphorus enters UK surface waters annually, of which 43% derives from agriculture (Morse *et al*., 1993). The control of diffuse (agricultural) sources of phosphorus is, therefore, assumed to be an important aspect of a national strategy to combat eutrophication, in addition to controls on point (principally sewage effluent) sources (Environment Agency, 2000).

Previous work has focussed on a gap assessment of phosphorus loads delivered to rivers (Defra project WT0719CSF) but no work was done by Defra CSF to evaluate baseline loads to lakes, mainly because there was no agreed standard under WFD to establish a gap comparison. Standards have now been set for lakes based on a simplified typology (UKTAG, 2006) so a gap study can now be conducted. The forecasts of changes in agriculture structure and practices for 2015 have also recently been updated under a third BAU evaluation (Shepherd *et al*., 2007). Defra have therefore requested a quantitative assessment of the phosphorus gap and source apportionment for lakes, that takes account of the revised BAU scenario.

1.2 Objectives

The objectives of this project were therefore to:

- Calculate the total phosphorus load to lakes in England and Wales under the Business as Usual forecast for 2015, and provide an assessment of the percentage contribution from agricultural and other sources; and

- To predict the total phosphorus concentration in the lake waters, compare with standards for Good Ecological Status set under the Water Framework Directive, and to report on the extent of compliance and the load reduction required to achieve good status.

This project concerns only total phosphorus concentrations in the standing waters or lakes with a surface area greater than 1 ha, as defined by the GB-Lakes inventory (Hughes *et al*., 2004).

2. General Methodology

Total phosphorus loads to lakes within England and Wales were estimated using computer models for diffuse sources and an inventory approach for point sources (Section 3). The diffuse agricultural phosphorus loads were based on the existing model framework developed under Defra projects WQ0106 and WT0719CSF that employs the PSYCHIC (Phosphorus and Sediment Yield Characterisation in Catchments) model to calculate baseline phosphorus loads (Anthony, 2006; Anthony and Lyons, 2006).

Annual average lake total phosphorus concentrations are then estimated using the OECD (1982) combined lakes model that predicts concentrations as a function of the total lake phosphorus load and hydraulic residence time:

(EQ 1)
$$P_l = 1.55 \cdot \left(\frac{L}{Q} \cdot \frac{1}{1+\sqrt{T}} \right)^{0.82}$$

where P_l is the average total phosphorus concentration (μg l^{-1}), L is the annual area phosphorus load (mg m^{-2} yr^{-1}), Q is the area water load (m yr^{-1}) and T is the hydraulic residence time (yr). The area water load and hydraulic residence time were calculated on the basis of summary lake geometry and catchment runoff data in the GB Lakes Inventory (Bennion *et al*., 2003; Hughes *et al*., 2003).

This model conceptualises retention of phosphorus in a lake as a function of lake average depth and a particle settling velocity, and assumes that the lake is well mixed (Brett and Benjamin, 2008). This interpretation ignores the potential release of phosphorus from bed sediments that accumulate over time. It is emphasised that the model is empirical, and many variants have been derived for application to lake systems with specific character (see, for example Foy, 1992; Prairie, 1989). Differences between lake systems in the proportions of soluble and particulate phosphorus in the nutrient load, the morphology of the water bodies, trophic status and ecosystem structure, water and bed sediment chemistry, and stratification will result in differences between measurement and predictions and require further calibration of the model coefficients. Kronvang *et al*. (2004), for example, developed separate variants of the model for natural lakes and reservoirs, in which phosphorus retention is greater for the same value of residence time. Overall, however, the OECD (1982) model has been applied to the greatest number of lakes and was able to explain 86% of the variance in measured concentrations for 87 lake systems (Hakanson, 1999). It is emphasised that the measure of model fit is the result of the log-transformation of both measured and predicted concentrations. The 95% prediction confidence interval spans a five-fold variation in concentration and its use as a precise predictive management tool for individual lakes has been questioned (see, for example, Reynolds and Davies, 2001). However, its application to the calculation of population statistics for a very large number of lakes as in this project ought to be robust. The log-transformation also results in a bias when the predictions are re-transformed back to linear space. This occurs because the model predicts the geometric average concentration that will be less than the arithmetic average (Ferguson, 1986). Based on analyses by Brett and Benjamin (2008) a correction factor of *c*. 1.2 can be calculated by the method of Duan (1982). This correction is significant if the concentration standards are not also expressed as geometric averages. The concentration standards developed by the OECD (1982) and UK-TAG (2007), which are used in this report, are based on geometric averages.

2.1 Water Quality Standards

A number of phosphorus classifications and quality standards for lakes have previously been widely applied (e.g. OECD, 1982; Cardoso 2001). These schemes are, however, not WFD-compliant, in that they do not assess lake status in terms of deviation from a reference state that represents high ecological status. Although phosphorus is a supporting quality element in the WFD, Annex II of the directive outlines a requirement to establish type-specific reference conditions for supporting physicochemical (and hydromorphological) quality elements. The key ecological importance of phosphorus and its focus for catchment management, also mean that the UK has set a requirement for WFD-compliant environmental standards for phosphorus in freshwaters.

Earlier research established provisional reference conditions and good status threshold values for total phosphorus for GB lake types (Carvalho *et al*., 2005). A previous analysis of lakes at risk of failing WFD phosphorus standards was carried out using these preliminary phosphorus standards (Carvalho *et al.,* 2005)

A number of approaches were used to identify these provisional phosphorus standards, including analysis of a small set of about 50 UK reference lakes and application of the Morpho-Edaphic Index (MEI) developed by Vighi and Chiaudani (1985). The MEI model is an empirical regression model that predicts phosphorus concentrations in waters not impacted by pollution, and requires only data on lake mean depth and alkalinity or conductivity. The MEI-TP model of Vighi and Chiaudani (1985) is, however, based on only 53 'reference' lakes, of which only 12 were in Europe, most were of low alkalinity and very deep (mean depth was 37 m).

For this reason, more recent research was carried out at both a UK and European level to agree more applicable and consistent phosphorus standards for common European lake types. Nine lake types were defined by mean depth and alkalinity, with boundaries between types identified largely following UKTAG (2004) and intercalibration guidance (Van de Bund *et al*., 2004). The boundaries between the nine core lake types used in this study are indicated in Table 1.1. In terms of phosphorus standards, some of these lake types are further sub-divided by European region (Northern or Central European high alkalinity lakes) or specific geology (e.g. Marl high alkalinity lakes in limestone catchments). England and Wales is dominated by very shallow, high alkalinity lakes and has very few deep lakes (Table 1.2; Figure 1.1)

Table 1.1 Core lake type definitions used in the analysis (UK-TAG, 2008).

Lake Type Code	**Description**	**Broad Geology**	**Alkalinity (m equiv. l^{-1})**	**Mean depth (m)**
LA_VSh	Low alkalinity, very shallow	Siliceous		<3
LA_Sh	Low alkalinity, shallow	Siliceous	<0.2	3-15
LA_D	Low alkalinity, deep	Siliceous		>15
MA_VSh	Medium alkalinity, very shallow	Siliceous		<3
MA_Sh	Medium alkalinity, shallow	Siliceous	0.2 - 1.0	3-15
MA_D	Medium alkalinity, deep	Siliceous		>15
HA_VSh	High alkalinity, very shallow	Calcareous		<3
HA_Sh	High alkalinity, shallow	Calcareous	>1.0	3-15
HA_D	High alkalinity, deep	Calcareous		>15

Table 1.2 Numbers of lakes by lake type used in this analysis, across England and Wales

	Unclassified	Deep	Shallow	Very shallow	Total
Low Alkalinity	0	17	397	386	800
Medium Alkalinity	0	4	181	330	515
High Alkalinity	1	0	777	3276	4054
Marl	0	0	124	43	167
Total	1	21	1479	4035	5536

Table 1.3 Type specific annual mean total phosphorus concentrations (µg $l^{-1}P$) at lake status boundaries used in this analysis (Carvalho *et al.*, 2006).

Type	Depth	Region	Type Reference	High/ Good	Good / Moderate	Moderate / Poor	Poor / Bad
HA	Shallow	Central	20	25.6	35.7	71.4	142.8
		North	13	16.7	23.2	46.4	92.8
	Very shallow	Central	28	35.9	49.1	98.2	196.4
		North	18	23.1	31.6	63.2	126.4
LA	Very shallow	Central	7	9.3	13.7	27.4	54.8
		North	7	9.3	13.7	27.4	54.8
	Shallow	Central	5	6.7	9.8	19.6	39.2
		North	5	6.7	9.8	19.6	39.2
	Deeper	Central	6	8.1	12.5	25	50
		North	6	8.1	12.5	25	50
MA	Shallow	Central	8	10.7	15.7	31.4	62.8
		North	8	10.7	15.7	31.4	62.8
	Very shallow	Central	12	15.8	23.1	46.2	92.4
		North	12	15.8	23.1	46.2	92.4
	Deeper	Central	6	8.1	12.5	25	50
		North	6	8.1	12.5	25	50
Marl	Shallow		9		20	40	80
	Very shallow		10		24	48	96

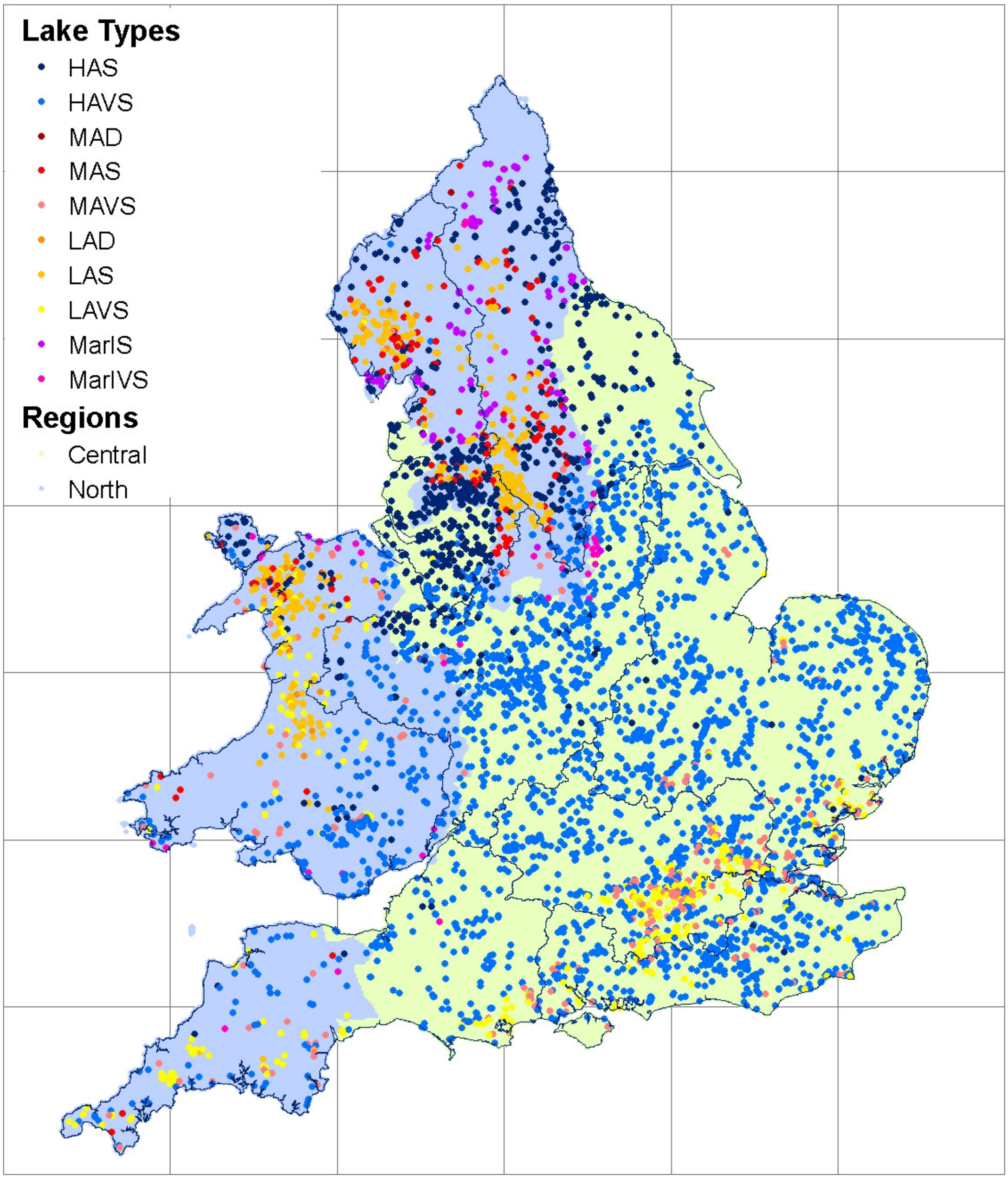

Figure 1.1 Distribution of lake types used in this analysis.

Type specific phosphorus concentrations for good ecological status were developed. This involved collating a large database of more than 500 European reference lakes and developing a European-version of the MEI model (Cardoso *et al*., 2007). UK Environmental Standards for phosphorus were developed in tandem with these to span all UK lake types (UK TAG, 2007). For full details of the method development see Carvalho *et al*. (2006).

Site-specific phosphorus standards are available for about 160 lakes in England and Wales with known depth and alkalinity data. All lakes use the clear-water MEI model outlined in Cardoso *et al*. (2007). Where these data are not known, type-specific standards have been set, as agreed by UK TAG in the Environmental Standards Report (outlined in Carvalho *et al*., 2006; Table 1.3). Lake types were estimated for

5,536 lakes in England and Wales for this analysis (Table 1.2) using information provided in the GB Lakes inventory dataset (version 2.2, November 2007).

3. Environment Data

This project is concerned only with the standing waters defined by the GB Lakes inventory (Hughes *et al*., 2004; Bennion *et al*., 2003). The inventory identifies a total of 17,941 waters in England and Wales, of which 6,309 have a surface area greater than 1ha. Our analysis considered only 5,536 lakes in England and Wales. This excluded a) some lakes where data in the GB Lakes inventory was inconsistent with a lake surface area greater than the catchment area; b) lakes located beyond the available national land use maps; and c) lakes for which a typology was not available.

The lake boundaries were originally extracted from Ordnance Survey PANORAMA mapping at a scale of 1:50,000, and the boundaries of the lake catchment areas were derived automatically from the Institute of Hydrology (IH) digital terrain model with a spatial resolution of 50 by 50 m^2 (Morris and Flavin, 1990). Phosphorus loads from point and diffuse sources were generally mapped at a spatial scale of 1 by 1 km^2 using statistical data on the mix of land uses within each cell. The loads impacting on an individual lake were then calculated by area-weighting the load map data in proportion to the fraction of each 1 by 1 km^2 cell that was located within the lake catchment boundary.

Because the catchment areas of the majority of lakes are small and the estimates of pollutant load are therefore uncertain, we caution that the analysis should be viewed as a statistical report on the proportion of the lake population that is at risk. The predictions made for an individual lake will have significant uncertainty, but the population average ought to be more robust.

3.1 Lake Morphometry

Application of the phosphorus concentration model required an estimate of the average lake depth, from which an estimate of the water body volume could be calculated. The GB Lakes inventory includes estimates of lake depth that were made using a multiple regression model based on lake surface area, altitude and perimeter for lakes throughout Great Britain. Estimation of lake volume in this way is uncertain. Although regression models with a high r^2 can be developed (see, for example, Rowan *et al*., 2006), the prediction confidence intervals for an individual lake remain large. Measured depth data were used where available, but this was only for a small percentage (2%) of the lakes.

The GB Lakes inventory also includes estimates of the annual average runoff for the period 1995 to 1997 for each lake, provided by the Centre for Ecology and Hydrology (CEH). Rainfall in the period was *c*. 10% lower than the long-term (1971-2000) average for England and Wales, according to the Hadley regional precipitation time series (http://www.metoffice.gov.uk/research/hadleycentre/obsdata/HadEWP.html). These runoff data were used to estimate the area water load in the combined lakes model (Section 2.1).

3.2 Diffuse Phosphorus Loading

3.2.1 Atmospheric Deposition

The direct deposition of phosphorus over the water surface area for each lake catchment was calculated by integrating annual average rainfall data with estimates of the total phosphorus concentration in rainfall.

Newman (1995) reviewed worldwide measurements of atmospheric total phosphorus deposition. Literature values were in the range of 0.07 to 1.7 kg ha^{-1} P per year. The major sources of atmospheric phosphorus are soil material, pollen and burnt organic material like plants, coal and oil. There are few regular measurements of phosphorus deposition in the United Kingdom, though soluble reactive phosphorus in rainfall is measured for the Environment Change Network (ECN) monitoring sites (www.ecn.ac.uk). The ECN network comprises 11 sites in the United Kingdom (7 in England, 1 in Wales, 2 in Scotland, and 1 in Northern Ireland). Average soluble phosphorus concentrations in rainfall in the period 1992 to 2005 were in the range 0.004 to 0.048 mg l^{-1} P with annual average loads of up to 0.6 kg ha^{-1} P.

Neal *et al*. (2004) measured annual average soluble phosphorus concentrations of 0.03 mg l^{-1} P in rainfall at a site in mid-Wales from 1994 to 2001, and average concentrations of 0.01 and 0.06 mg l^{-1} P in the Pang and Lambourne catchments in southeast England (Neal *et al*., 2004).

Wadsworth and Webber (1980) measured phosphate concentrations at 6 MAFF experimental farm sites. Average concentrations derived from the reported rainfall and average loads range between 0.03 and 0.1 mg l^{-1} P with highest concentrations in Drayton and lowest concentrations at the Great House and in Pwllpeiran. Williams (1967) further reported phosphorus concentrations and annual loads in rainfall at the experimental research stations at Rothamsted, Woburn and Saxmundham. The phosphorus concentrations at Woburn and Saxmundham were similar and relatively small (0.03 and 0.02 mg l^{-1} P) but a very large value was reported for Rothamsted (0.13 mg l^{-1} P).

The ratio of soluble to total phosphorus in rainfall has been reported in the range 0.05 to 0.9 (Kopacek *et al*., 1997). For the Pang and Lambourne catchments the ratios were 0.6 and 0.4 (Neal *at al*., 2004).

The brief review of phosphorus concentrations in rainfall revealed a wide range of data values that could not be correlated with catchment characteristics. Average soluble phosphorus concentrations in rainfall ranged from 0.004 to 0.13 mg l^{-1}P, with 50% of the reported values between 0.02 to 0.05 mg P l^{-1}. The ECN monitoring sites were considered most representative of the high rainfall and extensive grassland catchments in which atmospheric deposition would be a significant component of the total phosphorus budget. The average soluble phosphorus concentration based on only the ECN sites was 0.022 mg l^{-1} P. The total phosphorus concentration in rainfall was therefore fixed at 0.045 mg l^{-1} P based on the average ratio of soluble to total phosphorus reported by Neal *et al*. (2004). The total annual phosphorus deposition directly to rivers and lakes in England and Wales is 44 t P.

3.2.2 Urban Runoff

Phosphorus inputs from road runoff and diffuse urban sources to watercourses across England and Wales, were estimated using an Event Mean Concentration (EMC) methodology wherein calculations of average annual runoff were combined with event mean concentrations to estimate the annual phosphorus load (Mitchell, 2005). Representative total phosphorus EMC for roads and urban areas were selected, based on the database developed by Mitchell *et al.* (2001). The concentrations were in the range 0.2 to 0.4 mg l^{-1}. The typology of urban areas was based on the CORINE land cover dataset.

The annual average runoff *L* (mm) from the urban land area was calculated using the Wallingford Procedure:

$$L = R \cdot (0.829 \cdot P + 25.0 \cdot SOIL + 0.078 \cdot U - 20.7)$$

where *P* is the proportion of the land area that is impermeable, *R* is the annual average rainfall (mm), SOIL is the winter rainfall acceptance value of the local soil type (0.15 to 0.50; NERC, 1975), and *U* is the urban catchment wetness index that is determined from annual average rainfall (Mitchell *et al*., 2001). The urban area within each of the river catchments was estimated from the Land Cover Map of Great Britain. The proportion of land that is impermeable was determined from the density of households in the Ordnance Survey Address Point database, and assumptions used in sewer modelling systems (Ellis, 1986). This calculation was carried out for individual 1km^2 cells across the study area and the results aggregated to each of the lake catchment areas.

The urban diffuse phosphorus load is in the range of 0 to 4.3 kg ha^{-1} P of urban land, with an average of 0.7 kg ha^{-1} P of urban land. Averaged over the total land area of England and Wales the diffuse urban phosphorus load was equivalent to 0.06 kg ha^{-1} P. The calculated total annual phosphorus loss in England and Wales to rivers and lakes is 851 t P.

3.2.3 River Bank Erosion

River bank erosion was calculated using the model developed by Anthony and Collins (2007). This model estimates bank erosion as a function of the duration of excess bank shear stress following the general methodology of Julian and Torres (2006). It was previously calibrated against measurements of the bank erosion contribution to catchment sediment yield derived from sediment finger-printing studies. Bank erosion was calculated to be in the range 10 to 670 kg ha^{-1} sediment of the local catchment area, with a national average value of 30 kg ha^{-1}. The associated phosphorus loss was calculated for individual river catchments using data on the average soil total phosphorus content for each catchment, and then the data interpolated onto a 1km^2 grid for summarising to lake catchment areas. The topsoil phosphorus content was in the range 175 to 780 mg P kg^{-1} soil with an average value of 650 mg P kg^{-1} soil, based on NSRI data. The average total phosphorus load was then estimated to be 0.02 kg ha^{-1}. The annual total phosphorus loss by bank erosion to rivers and lakes in England and Wales was calculated to be 292 t P.

3.2.4 Agricultural Load

The PSYCHIC model (Davison *et al*., 2008; Collins *et al*., 2007) is a process based, monthly time-stepping, model with explicit representation of surface and drain flow hydrological pathways, particulate and solute mobilisation, and incidental losses associated with fertiliser and manure spreading. The model has previously been integrated with the soils, climate and agricultural census data held in the MAGPIE decision support system (Lord and Anthony, 2000) to calculate total phosphorus losses from all agricultural land, including rough grazing and runoff from hardstandings. The model predictions of total phosphorus loss take account of landscape retention, and are therefore the best available estimate of net delivery to lakes and rivers. The total phosphorus loss from all agricultural land was 5.7 kt P, and was equivalent to a national average loss of 0.44 kg ha^{-1} P across all agricultural land (including woodland and rough grazing). The total agricultural land area is 12.6×10^{6} ha of which 35% is improved grassland, 17% is rough grazing, 12% is woodland or forest, and 36% is cultivated. The model predictions of total phosphorus loss used year 2000 agricultural census and land cover data (Figure 3.1). The baseline output from the PSYCHIC model also assumes that landowners implement no mitigation methods that might reduce phosphorus losses.

The diffuse agricultural loadings used in this project to predict lake phosphorus concentrations are for the years 2004 and 2015. This required adjusting the PSYCHIC model output to take account of changes in the agricultural land area and number of livestock in the intervening period. A previous project developed a much simpler export coefficient model that predicted phosphorus losses as a function of crop areas and total inputs of phosphorus in fertiliser and animal manures (Anthony, 2006). The model was applied to agricultural census data for representative farms located in a small number of regions defined by combinations of soil texture and climate, and further by the boundaries of the Nitrate Vulnerable Zone and Catchment Sensitive Farming priority catchments (Figures 3.2 and 3.3). This simpler model facilitated rapid calculation of the impact of changes in the agricultural census on phosphorus losses within each of the regions. The percentage changes in the export coefficient model results under each scenario were applied as scalars to the more detailed PSYCHIC model results; the scalar for each 1 by 1 km^2 model cell being determined by the region in which it is located (Anthony, 2006).

The same approach was used in this project to scale the PSYCHIC model output to account for changes in the crop areas and the number and excreta production by livestock between 2000 and either 2004 or 2015. To accomplish this, we made use of the Business as Usual forecasts of agricultural structure and livestock excreta production made by Shepherd *et al*. (2007). Agricultural census data for 2004 and 2015 were mapped on a 10 by 10 km^2 grid under this project. Regional summaries of census data for 2000 and 2004 were used to derive scalars for mapping the 2000 census on to the same grid. Forecasts of total excreta production and the phosphorus content of excreta by livestock type, also from the Business as Usual project, were integrated with the agricultural census data to map the total pollutant potential in manures. The summary data at 10 by 10 km^2 resolution were then used to derive scalars of change in crop areas, stock counts and manure phosphorus production for each individual cell. These were then spatially integrated with census data for 2000 mapped on a 1 by 1km^2 grid to produce fine scale census maps for 2000, 2004 and 2015. Total crop areas, stock counts and manure phosphorus production were then summarised for each of the soil, climate and scheme zones to derive zone coefficients for structural change in the agricultural industry (Anthony, 2006). The results of these analyses are summarised (Table 3.1). Overall, a 10 to 20% reduction in phosphorus in cattle excreta was forecast by 2015 relative to the year 2000, and a 20 to 30% reduction in pig phosphorus in excreta.

The export coefficient model also permitted calculation of the impact of mitigation methods. A standardised list of diffuse pollution mitigation methods were reviewed by an expert group and reported on in a User Manual (Cuttle *et al*., 2006). This manual summarised empirical evidence and model based assessments to give a reduction in the pollutant loss following implementation of each method of the representative farms. A methodology was developed that estimated the percentage implementation and efficiency of the mitigation methods by land owners within each of the geographical zones, taking account of the impact of existing legislative and supportive drivers for changes in agricultural practices. The methodology was integrated with the export coefficient model to give estimates of diffuse pollution reduction due to adoption of mitigation methods within each of the zones. The full detail of the methodology, including algorithms and coefficients can be found in Anthony (2006).

This same approach was again implemented in this project to scale the PSYCHIC model output for changes in agricultural practice. This required preparation of new estimates of mitigation method implementation for 2004 and 2015 (Tables 3.2 and 3.3).

The net effect of the changes in crop areas, livestock numbers and of mitigation method implementation were calculated using the export coefficient model to derive scalars of change applied to the baseline output from the PSYCHIC model for the year 2000. Overall total phosphorus losses were calculated to reduce to 5.0 kt P in 2004 and to 4.6 kt P by 2015. The long-term reduction in total phosphorus loss was due equally to the changes in industry structure and to adoption of mitigation methods. Figures 3.4 and 3.5 summarise the spatial patterns of changes in phosphorus loss from diffuse agricultural sources relative to the baseline agricultural census year 2000 and assuming no mitigation methods in place.

3.2.5 Groundwater

Measurements of phosphorus in groundwater are limited and frequently below the limits of detection. Neal (2004) reported average concentrations of 0.01 to 0.04 mg l^{-1} P for borehole samples in the Kennet and Pang catchments. Jarvie *et al.* (2005) reported concentrations up to 0.97 mg l^{-1} with an average value of 0.05 mg l^{-1} for the groundwater in the Avon catchment.

Environment Agency measurements of groundwater orthophosphate in the period 1996 to 2000 were summarised by this project. Concentrations below the limit of detection were replaced with values equal to one half the measured value. Measured concentrations greater than 10 mg l^{-1} P were excluded to minimise the risk of incorrect data being included in the analyses. This subset included a large number of measurements in the range 100 to 500 mg l^{-1} P and suggested that sample units had been misreported as micro-grammes (μg) in place of milli-grammes (mg). Data were available for a total of 1,240 sample sites. There was no information on the depth of sampling or the type of borehole, but sites at landfills were excluded on the basis of the site name. Orthophosphate concentrations were in the range < 0.002 to 5.828 mg l^{-1} P with an average value of 0.115 ± 0.290 mg l^{-1} P. The sample sites were generally located over the major Upper Cretaceous Chalk, Jurassic Limestone and Permian Sandstone aquifers. The average orthophosphate concentrations in these groundwater units were 0.059 ± 0.125 mg l^{-1} P in the Chalk (*n* 343), 0.077 ± 0.203 mg l^{-1} P in the Limestone (*n* 89) and 0.171 ± 0.364 mg l^{-1} P in the Sandstone (*n* 328). The highest concentrations in boreholes were possibly associated with industrial or urban pollution, or point source pollution from farmsteads such as leakage from manure stores. However, the measured concentrations were also clearly influenced by the analytical minimum levels of detection that varied regionally. For example, the typical minimum level of detection was 0.50 mg l^{-1} in the north-west region and 0.02 mg l^{-1} in the north-east. Average orthophosphate concentrations in the Permian Sandstone aquifer in the north-west were 0.19 mg l^{-1} compared to 0.02 mg l^{-1} P in the north-east region.

The analysis was therefore restricted to the Anglian, Thames and South-West regions where the laboratory limits of detection were 0.02 or 0.03 mg l^{-1} P. The calculated average orthophosphate concentrations were 0.038 ± 0.089 mg l^{-1} P in the Chalk (*n* 192) and 0.025 ± 0.024 mg l^{-1} P in the Limestone (*n* 50) aquifers. The sample distributions were positively skewed. The median values were 0.020 mg l^{-1} P in the Chalk and 0.015 mg l^{-1} P in the Limestone aquifers. In river catchments dominated by groundwater flows, these concentrations would be sufficient to contribute an additional annual phosphorus load of up to 0.1 kg ha^{-1} P to rivers feeding lake systems.

It was believed that the calculated average groundwater concentrations were higher than the true values due to the minimum levels of detection in the laboratory analyses. These calculated averages were also characteristic of groundwater catchments with intensive agriculture. It was reasoned that concentrations would generally be less in surface water catchments and in those with less intensive agriculture. The final phosphorus load from groundwater to lakes was therefore calculated by integrating the PSYCHIC model calculations of soil drainage to groundwater (as opposed to surface and lateral flow) with an national average total phosphorus concentration of 0.01 mg l^{-1} P. This calculation assumed that regional groundwater flows fully interact with rivers feeding the lake systems. There remains an uncertainty in the estimate of the groundwater contribution to the total phosphorus load in a lake catchment. Loads may be under-estimated in catchments with a large area of unconfined aquifer.

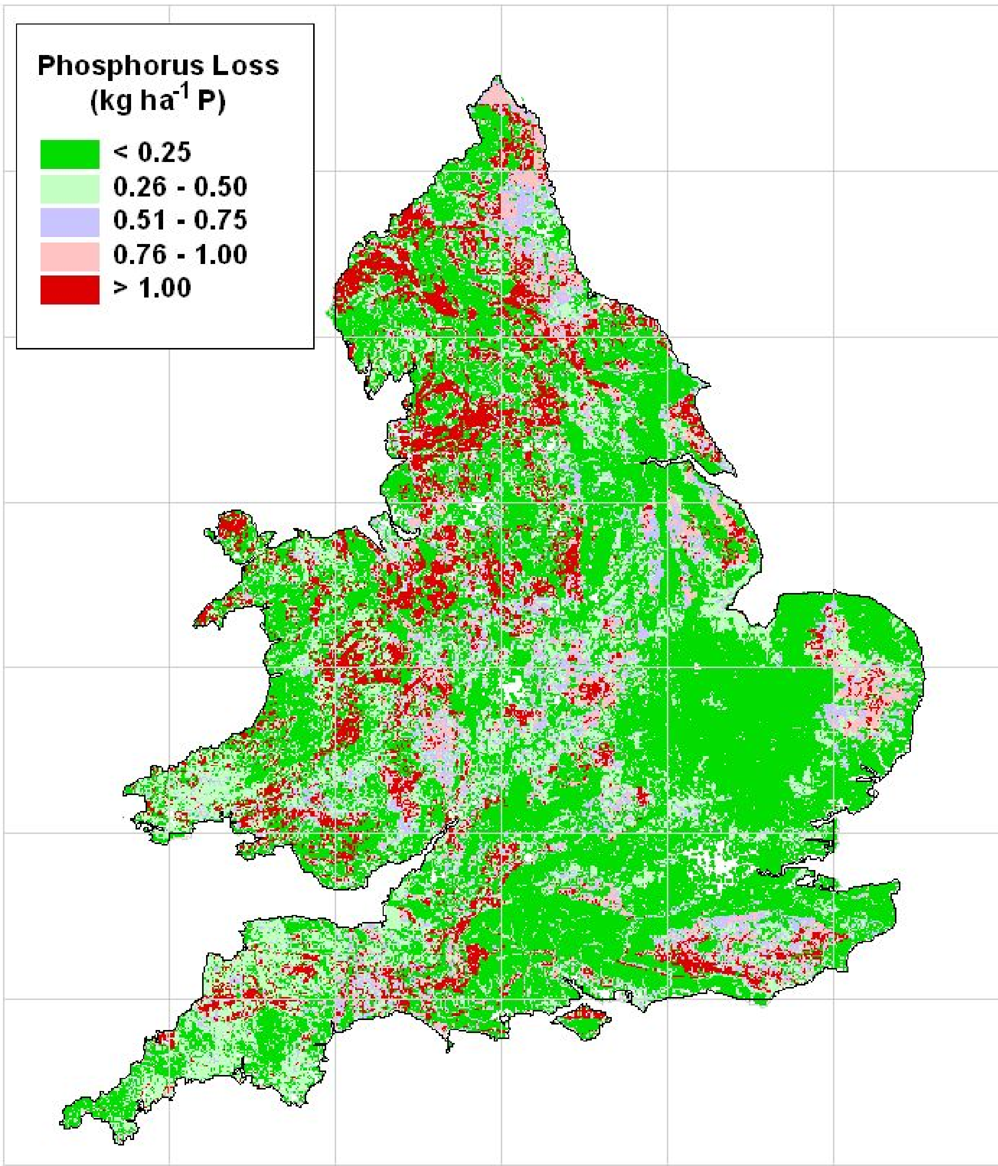

Figure 3.1 Annual average total phosphorus loss from diffuse agricultural sources, expressed per hectare of all agricultural land, as calculated by the PSYCHIC model using agricultural census data for the year 2000, and assuming that no mitigation methods are implemented.

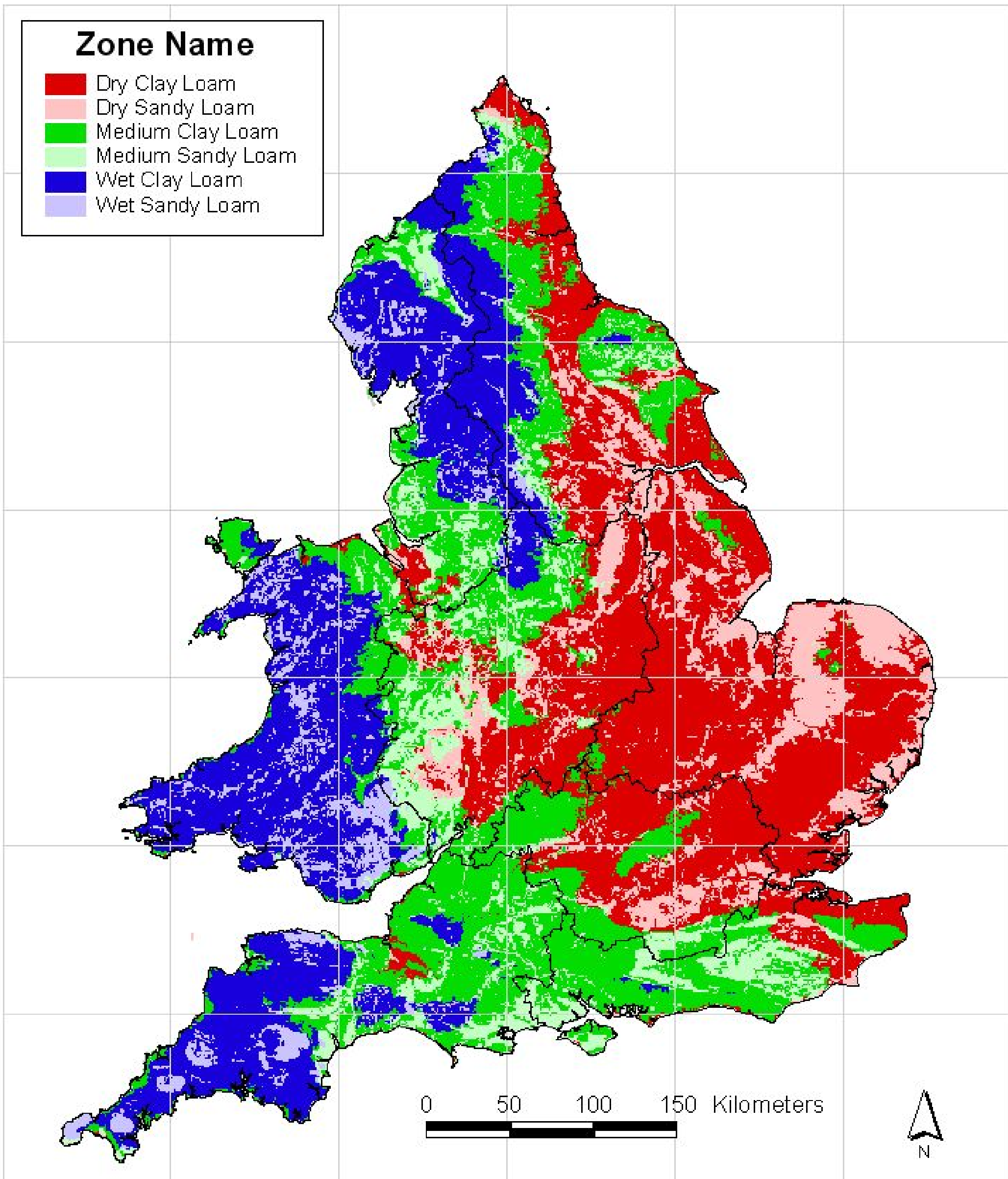

Figure 3.2 Climate and soil zones defined for the extrapolation of the results of field scale models, applied to representative farm systems, across England and Wales.

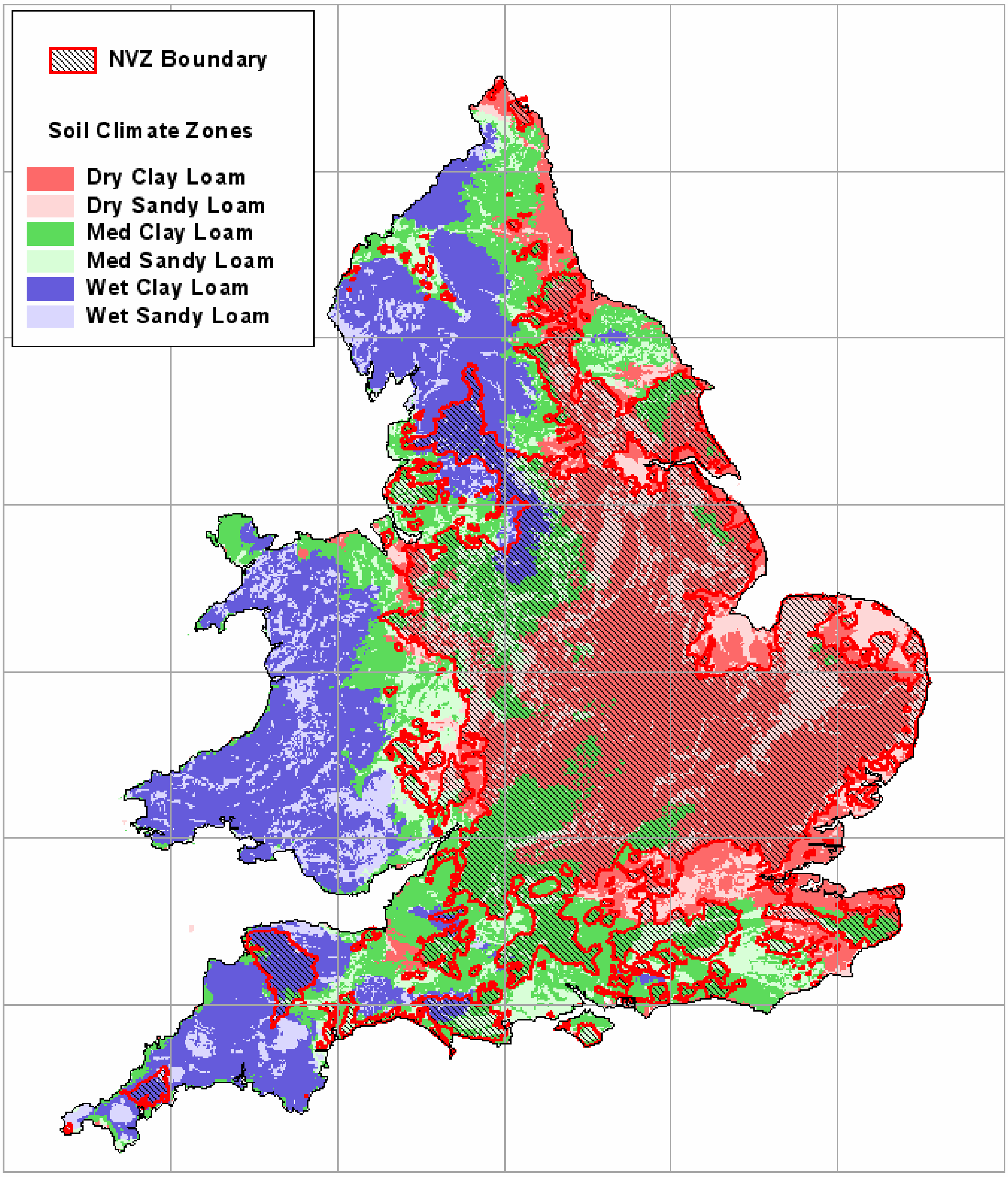

Figure 3.3a Climate and soil zones defined for the extrapolation of the results of field scale models applied to representative farm systems across England and Wales (see Section 3) with superimposed boundaries of the existing Nitrate Vulnerable Zones (Defra, 2002).

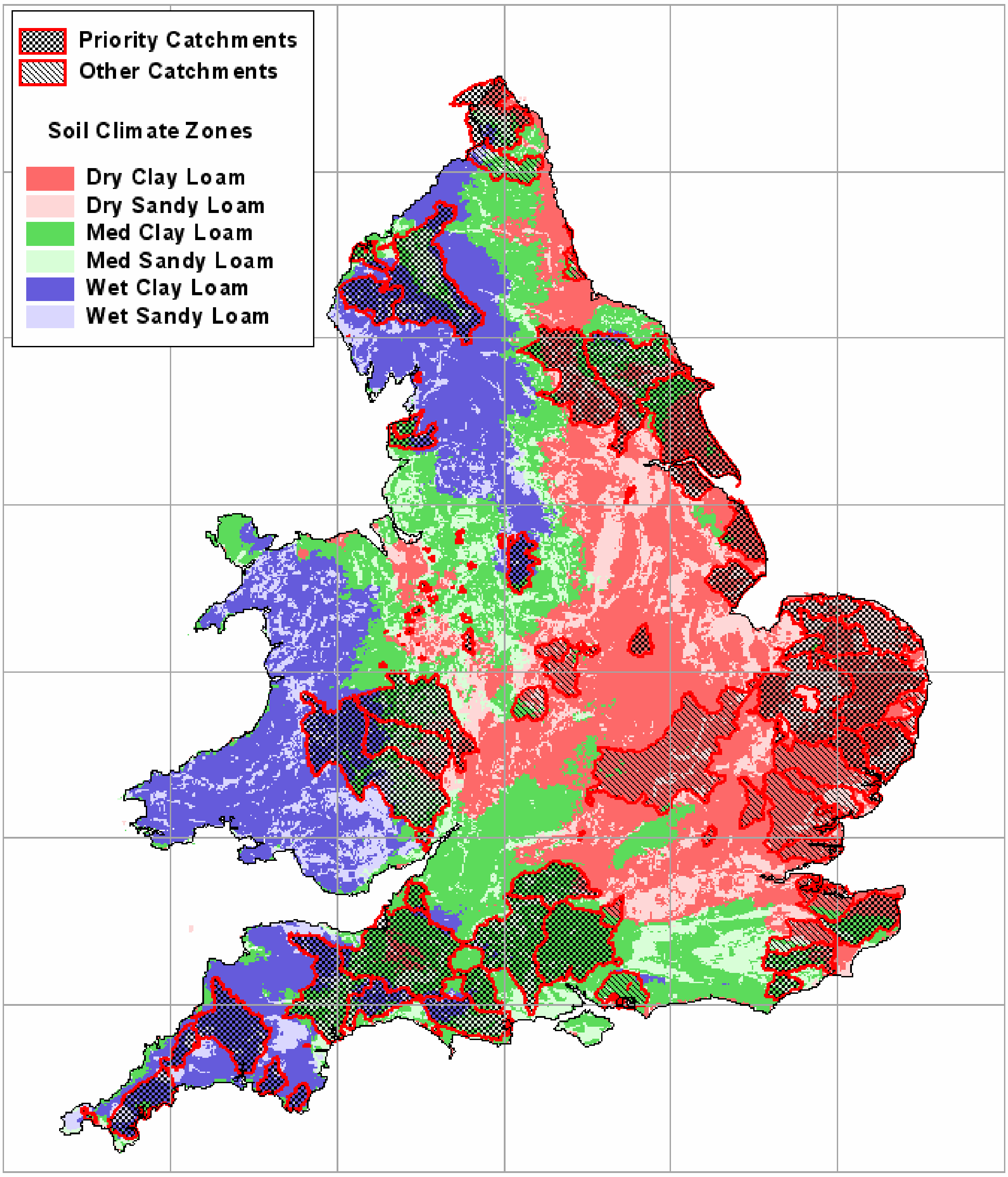

Figure 3.3b Climate and soil zones defined for the extrapolation of the results of field scale models applied to representative farm systems across England and Wales (see Section 3) with superimposed boundaries of the existing 40 priority catchments and additional catchments identified under the England Catchment Sensitive Farming Delivery Initiative (Defra, 2002).

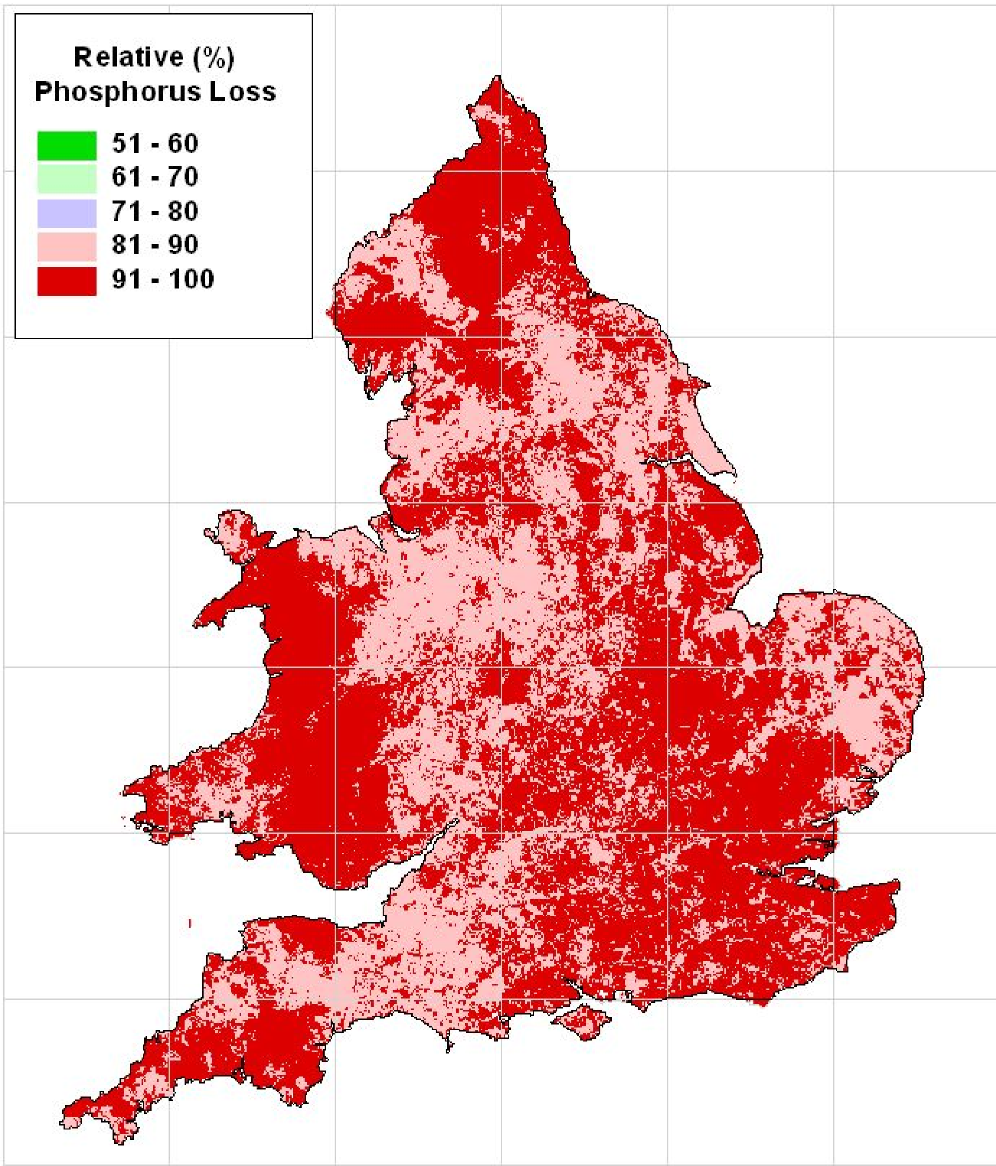

Figure 3.4 Modelled phosphorus loss from agricultural land (including rough grazing and woodland) in the year 2004, expressed as a percentage of baseline PSYCHIC model output for the year 2000 (see Figure 3.1). This presents the combined impact of changes in agricultural census and the implementation of mitigation methods by 2004.

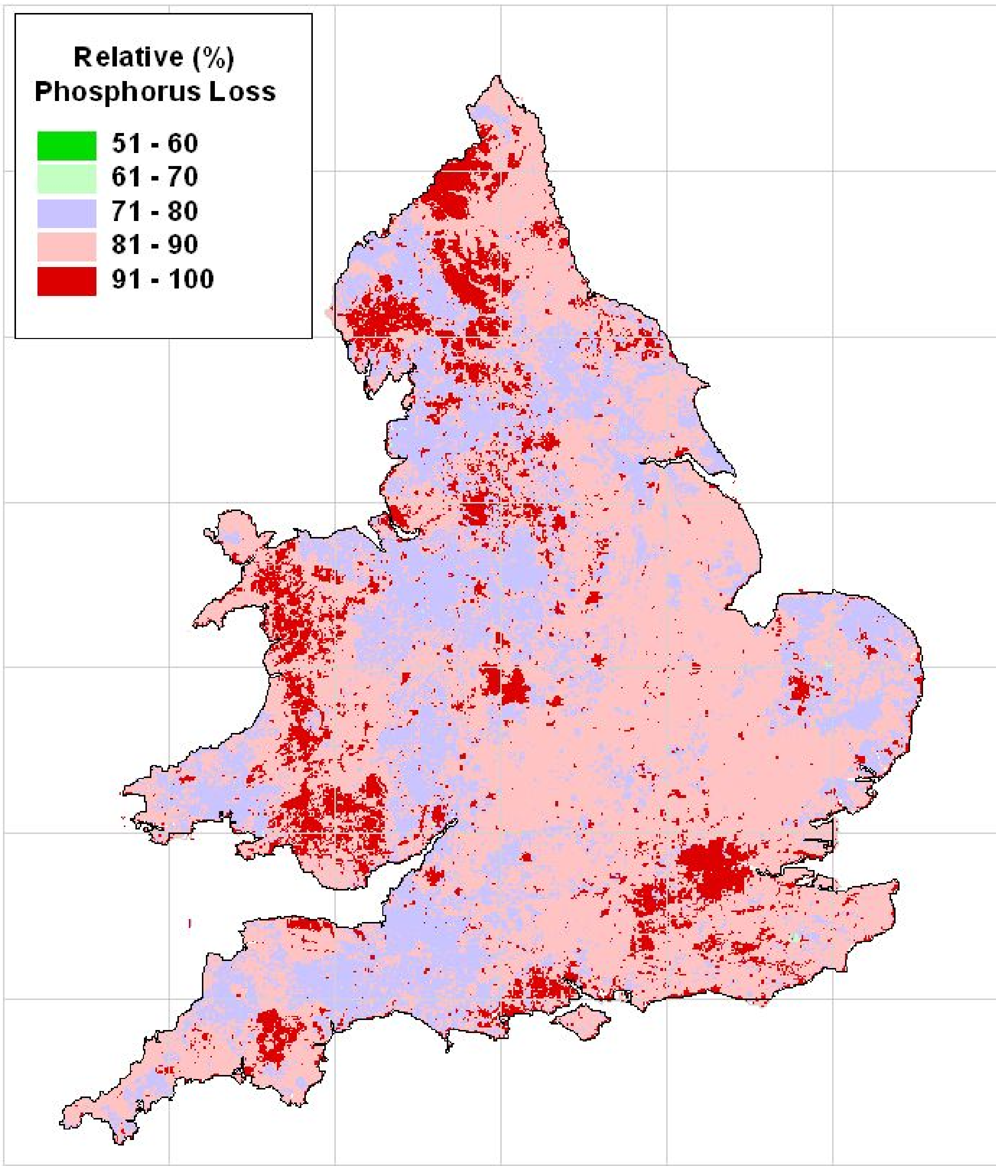

Figure 3.5 Modelled phosphorus loss from agricultural land (including rough grazing and woodland) in the year 2015, expressed as a percentage of baseline PSYCHIC model output for the year 2000 (see Figure 3.1). This presents the combined impact of changes in agricultural census and the implementation of mitigation methods by 2015.

	Arable		Grass		Sheep		Poultry		Pig		Beef		Dairy	
Zone	**2004**	**2015**	**2004**	**2015**	**2004**	**2015**	**2004**	**2015**	**2004**	**2015**	**2004**	**2015**	**2004**	**2015**
Dry Clay Loam	97	94	105	107	84	82	117	108	78	69	95	89	89	81
Dry Clay Loam NVZ	98	95	106	110	83	82	104	95	78	69	97	91	89	82
Dry Clay Loam CSF	97	94	106	109	83	81	107	106	81	71	97	90	89	82
Dry Clay Loam CSF NVZ	99	96	108	113	83	81	105	98	85	75	98	91	87	80
Dry Sandy Loam	98	94	105	107	84	82	107	97	76	67	95	89	90	81
Dry Sandy Loam NVZ	99	96	106	110	83	82	102	94	79	70	97	90	89	82
Dry Sandy Loam CSF	98	94	106	107	83	82	109	112	83	73	94	88	89	82
Dry Sandy Loam CSF NVZ	99	95	109	113	82	80	107	101	87	77	96	90	87	80
Med Clay Loam	98	94	105	105	85	82	102	92	72	64	94	88	91	82
Med Clay Loam NVZ	98	93	105	107	83	81	99	91	73	65	97	90	89	82
Med Clay Loam CSF	98	94	105	106	84	82	107	96	71	63	95	89	90	83
Med Clay Loam CSF NVZ	97	94	106	108	84	82	103	95	75	66	98	91	89	82
Med Sandy Loam	98	93	105	106	84	82	107	94	74	65	95	89	89	82
Med Sandy Loam NVZ	99	94	105	107	83	81	89	83	73	65	96	90	89	82
Med Sandy Loam CSF	98	94	105	106	84	82	105	95	72	64	94	88	90	83
Med Sandy Loam CSF NVZ	98	94	105	107	84	82	107	99	73	65	96	90	90	83
Wet Clay Loam	99	95	105	105	86	81	104	96	65	57	94	88	94	83
Wet Clay Loam NVZ	97	92	105	106	84	82	123	118	71	63	97	91	90	82
Wet Clay Loam CSF	98	94	105	105	85	81	118	109	69	61	96	89	90	83
Wet Clay Loam CSF NVZ	98	93	105	106	83	81	106	100	69	61	97	91	90	83
Wet Sandy Loam	101	97	105	105	86	81	93	84	62	54	95	88	93	82
Wet Sandy Loam NVZ	99	94	105	105	83	81	84	80	72	63	96	90	89	82
Wet Sandy Loam CSF	98	94	105	106	85	81	119	109	68	60	96	89	90	83
Wet Sandy Loam CSF NVZ	98	93	105	106	83	81	132	126	69	61	98	92	90	83

Table 3.1 Summary of the calculated areas of arable and grassland, and of the total phosphorus contents of livestock excreta in 2004 and 2015, expressed as a percentage of the baseline values for the year 2000.

Arable (2004)

ID	Measure	Measure Name	Dry Clay Loam	Dry Clay Loam NVZ	Dry Clay Loam CSF	Dry Clay Loam CSF NVZ	Dry Sandy Loam	Dry Sandy Loam NVZ	Dry Sandy Loam CSF	Dry Sandy Loam CSF NVZ	Med Clay Loam	Med Clay Loam NVZ	Med Clay Loam CSF	Med Clay Loam CSF NVZ	Med Sandy Loam	Med Sandy Loam NVZ	Med Sandy Loam CSF	Med Sandy Loam CSF NVZ	Wet Clay Loam	Wet Clay Loam NVZ	Wet Clay Loam CSF	Wet Clay Loam CSF NVZ	Wet Sandy Loam	Wet Sandy Loam NVZ	Wet Sandy Loam CSF	Wet Sandy Loam CSF NVZ
0	0	Convert arable land to non cropped land	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1	1	Convert arable land to extensive grassland: including livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2	2	Establish cover crops in the autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	3	Cultivate land for crop establishment in spring rather than autumn	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8
4	4	Adopt minimal cultivation systems	45	45	45	45	25	25	25	25	45	45	45	45	25	25	25	25	30	30	30	30	15	15	15	15
5	5	Cultivate compacted tillage soils	2	2	2	2	2	2	2	2	3	3	3	3	3	3	3	3	4	4	4	4	3	3	3	3
6	6	Cultivate and drill across the slope	5	5	5	5	5	5	5	5	5	5	5	5	6	6	6	6	5	5	5	5	8	8	8	8
7	7	Leave autumn seedbeds rough	20	20	20	20	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30	30
8	8	Avoid tramlines over winter	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
9	9	Establish in-field grass buffer strips	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
10	10	Loosen compacted soil layers in grassland fields	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	11	Maintain and enhance soil organic matter levels	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10
12	12	Allow field drainage systems to deteriorate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
13	13	Reduce overall stocking rates on livestock farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
14	14	Reduce the length of the grazing day or grazing season	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
15	15	Reduce field stocking rates when soils are wet	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
16	16	Move feed and water troughs at regular intervals	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	17	Reduce dietary N and P intakes	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
18	18	Adopt phase feeding of livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
19	19	Use a fertiliser recommendation system	60	70	60	70	60	70	60	70	60	70	60	70	60	70	60	70	60	70	60	70	60	70	60	70
20	20	Integrate fertiliser and manure nutrient supply	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60
21	21	Reduce N fertiliser application rates	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
22	21	Reduce P fertiliser application rates	25	30	25	30	30	35	30	35	20	20	20	20	30	20	30	20	20	25	20	25	20	25	20	25
23	22	Do not apply P fertilisers to high P index soils	10	10	10	10	15	15	15	15	10	10	10	10	15	15	15	15	10	10	10	10	15	15	15	15
24	23	Do not apply fertiliser to high-risk areas	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50
25	24	Avoid spreading fertiliser to fields at high-risk times	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60	50	60
26	25	Increase the capacity of farm manure stores	10	20	10	20	10	20	10	20	10	20	10	20	10	20	10	20	10	20	10	20	10	20	10	20
27	26	Minimise the volume of dirty water produced	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
28	27	Adopt batch storage of slurry	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
29	28	Adopt batch storage of solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
30	29	Compost solid manure	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	2	2	2	2	2	2	2	2
31	30	Change from slurry to a solid manure handling system	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	31	Site solid manure heaps away from watercourses and field drains	40	50	40	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50	50
33	32	Site solid manure heaps on concrete and collect the effluent	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
34	33	Do not apply manure to high-risk areas	20	20	20	20	60	60	60	60	30	40	30	40	60	65	60	65	20	30	20	30	50	60	50	60
35	34	Do not spread farmyard manure to fields at high-risk times	50	55	50	55	60	65	60	65	50	55	50	55	60	65	60	65	50	55	50	55	60	65	60	65
36	35	Do not spread slurry or poultry manure to fields at high-risk times	40	60	40	60	40	70	40	70	40	55	40	55	30	70	30	70	40	65	40	65	40	65	40	65
37	36	Incorporate manure into the soil	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	30	35	30	35
38	37	Transport manure to neighbouring farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	38	Incinerate poultry litter	5	10	5	10	5	10	5	10	5	10	5	10	5	15	5	15	15	22	15	22	15	22	15	22
40	39	Fence off rivers and streams from livestock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
41	40	Construct bridges for livestock crossing rivers and streams	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
42	41	Re-site gateways away from high-risk areas	1	1	1	1	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	5	5	5	5
43	42	Establish new hedges	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
44	43	Establish riparian buffer strips	40	45	40	45	40	45	40	45	40	45	40	45	40	45	40	45	40	45	40	45	40	45	40	45
45	44	Establish and maintain artificial (constructed) wetlands	0	1	0	1	0	0	0	0	0	1	0	1	0	0	0	0	0	1	0	1	0	1	0	1

Table 3.2a Summary of expected implementation of mitigation methods for climate and soil zones within and outside NVZ and ECSFDI catchment boundaries in 2004. Results are expressed as a percentage of the arable land area affected.

Grassland (2004)

ID	Measure	Measure Name	Dry Clay Loam	Dry Clay Loam NVZ	Dry Clay Loam CSF	Dry Clay Loam CSF NVZ	Dry Sandy Loam	Dry Sandy Loam NVZ	Dry Sandy Loam CSF	Dry Sandy Loam CSF NVZ	Med Clay Loam	Med Clay Loam NVZ	Med Clay Loam CSF	Med Clay Loam CSF NVZ	Med Sandy Loam	Med Sandy Loam NVZ	Med Sandy Loam CSF	Med Sandy Loam CSF NVZ	Wet Clay Loam	Wet Clay Loam NVZ	Wet Clay Loam CSF	Wet Clay Loam CSF NVZ	Wet Sandy Loam	Wet Sandy Loam NVZ	Wet Sandy Loam CSF	Wet Sandy Loam CSF NVZ
0	0	Convert arable land to non cropped land	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1	1	Convert arable land to extensive grassland: including livestock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	2	Establish cover crops in the autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	3	Cultivate land for crop establishment in spring rather than autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
4	4	Adopt minimal cultivation systems	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
5	5	Cultivate compacted tillage soils	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
6	6	Cultivate and drill across the slope	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
7	7	Leave autumn seedbeds rough	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
8	8	Avoid tramlines over winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	9	Establish in-field grass buffer strips	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	10	Loosen compacted soil layers in grassland fields	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
11	11	Maintain and enhance soil organic matter levels	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
12	12	Allow field drainage systems to deteriorate	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
13	13	Reduce overall stocking rates on livestock farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
14	14	Reduce the length of the grazing day or grazing season	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
15	15	Reduce field stocking rates when soils are wet	1	1	1	1	1	1	1	1	1	1	1	1	2	3	2	3	2	5	2	5	2	5	2	5
16	16	Move feed and water troughs at regular intervals	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
17	17	Reduce dietary N and P intakes	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
18	18	Adopt phase feeding of livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
19	19	Use a fertiliser recommendation system	30	40	30	40	30	40	30	40	20	25	20	25	20	25	20	25	20	25	20	25	20	25	20	25
20	20	Integrate fertiliser and manure nutrient supply	30	40	30	40	30	40	30	40	20	25	20	25	20	25	20	25	20	25	20	25	20	25	20	25
21	21	Reduce N fertiliser application rates	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40
22	21	Reduce P fertiliser application rates	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40	30	40
23	22	Do not apply P fertilisers to high P index soils	30	40	30	40	30	40	30	40	20	25	20	25	20	25	20	25	20	25	20	25	20	25	20	25
24	23	Do not apply fertiliser to high-risk areas	30	40	30	40	30	40	30	40	20	25	20	25	20	25	20	25	20	25	20	25	20	25	20	25
25	24	Avoid spreading fertiliser to fields at high-risk times	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55
26	25	Increase the capacity of farm manure stores	5	10	5	10	5	10	5	10	5	10	5	10	5	10	5	10	5	10	5	10	5	10	5	10
27	26	Minimise the volume of dirty water produced	3	5	3	5	3	5	3	5	3	5	3	5	3	5	3	5	3	5	3	5	3	5	3	5
28	27	Adopt batch storage of slurry	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
29	28	Adopt batch storage of solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
30	29	Compost solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
31	30	Change from slurry to a solid manure handling system	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	31	Site solid manure heaps away from watercourses and field drains	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55	50	55
33	32	Site solid manure heaps on concrete and collect the effluent	10	12	10	12	10	12	10	12	10	12	10	12	10	12	10	12	10	12	10	12	10	12	10	12
34	33	Do not apply manure to high-risk areas	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35
35	34	Do not spread farmyard manure to fields at high-risk times	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35	30	35
36	35	Do not spread slurry or poultry manure to fields at high-risk times	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40	40
37	36	Incorporate manure into the soil	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
38	37	Transport manure to neighbouring farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	38	Incinerate poultry litter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
40	39	Fence off rivers and streams from livestock	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
41	40	Construct bridges for livestock crossing rivers and streams	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
42	41	Re-site gateways away from high-risk areas	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
43	42	Establish new hedges	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
44	43	Establish riparian buffer strips	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
45	44	Establish and maintain artificial (constructed) wetlands	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table 3.2b Summary of expected implementation of mitigation methods for climate and soil zones within and outside NVZ and ECSFDI catchment boundaries in 2004. Results are expressed as a percentage of the grassland area affected.

Arable (2015)

ID	Measure	Measure Name	Dry Clay Loam	Dry Clay Loam NVZ	Dry Clay Loam CSF	Dry Clay Loam CSF NVZ	Dry Sandy Loam	Dry Sandy Loam NVZ	Dry Sandy Loam CSF	Dry Sandy Loam CSF NVZ	Med Clay Loam	Med Clay Loam NVZ	Med Clay Loam CSF	Med Clay Loam CSF NVZ	Med Sandy Loam	Med Sandy Loam NVZ	Med Sandy Loam CSF	Med Sandy Loam CSF NVZ	Wet Clay Loam	Wet Clay Loam NVZ	Wet Clay Loam CSF	Wet Clay Loam CSF NVZ	Wet Sandy Loam	Wet Sandy Loam NVZ	Wet Sandy Loam CSF	Wet Sandy Loam CSF NVZ
0	0	Convert arable land to non cropped land	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
1	1	Convert arable land to extensive grassland: including livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2	2	Establish cover crops in the autumn	0	0	1	1	0	0	1	1	0	0	1	1	0	0	1	1	0	0	1	1	0	0	1	1
3	3	Cultivate land for crop establishment in spring rather than autumn	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8
4	4	Adopt minimal cultivation systems	65	65	70	70	45	45	50	50	65	65	70	70	45	45	55	55	50	50	55	55	35	35	40	40
5	5	Cultivate compacted tillage soils	2	2	3	3	2	2	3	3	3	3	4	4	3	3	4	4	4	4	5	5	3	3	4	4
6	6	Cultivate and drill across the slope	5	5	5	5	5	5	8	8	5	5	5	5	6	6	9	9	5	5	5	5	8	8	10	10
7	7	Leave autumn seedbeds rough	40	40	45	45	50	50	55	55	50	50	55	55	50	50	55	55	50	50	55	55	50	50	55	55
8	8	Avoid tramlines over winter	1	1	1	1	1	1	3	3	1	1	1	1	1	1	4	4	1	1	3	3	1	1	5	5
9	9	Establish in-field grass buffer strips	3	3	5	5	3	3	5	5	3	3	5	5	3	3	5	5	3	3	5	5	3	3	5	5
10	10	Loosen compacted soil layers in grassland fields	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	11	Maintain and enhance soil organic matter levels	20	20	22	22	20	20	22	22	20	20	22	22	20	20	22	22	20	20	22	22	20	20	22	22
12	12	Allow field drainage systems to deteriorate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
13	13	Reduce overall stocking rates on livestock farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
14	14	Reduce the length of the grazing day or grazing season	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
15	15	Reduce field stocking rates when soils are wet	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
16	16	Move feed and water troughs at regular intervals	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	17	Reduce dietary N and P intakes	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
18	18	Adopt phase feeding of livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
19	19	Use a fertiliser recommendation system	80	90	85	95	80	90	85	95	80	90	85	95	80	90	85	95	80	90	85	95	80	90	85	95
20	20	Integrate fertiliser and manure nutrient supply	70	80	75	85	70	80	75	85	70	80	75	85	70	80	75	85	70	80	75	85	70	80	75	85
21	21	Reduce N fertiliser application rates	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
22	21	Reduce P fertiliser application rates	45	50	55	55	50	55	60	60	40	40	45	50	50	40	45	45	40	45	50	50	40	45	50	50
23	22	Do not apply P fertilisers to high P index soils	10	10	15	15	15	15	20	20	10	10	15	15	15	15	20	20	10	10	15	15	15	15	20	20
24	23	Do not apply fertiliser to high-risk areas	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75
25	24	Avoid spreading fertiliser to fields at high-risk times	80	90	85	90	80	90	85	90	80	90	85	90	80	90	85	90	80	90	85	90	80	90	85	90
26	25	Increase the capacity of farm manure stores	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30
27	26	Minimise the volume of dirty water produced	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
28	27	Adopt batch storage of slurry	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
29	28	Adopt batch storage of solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
30	29	Compost solid manure	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	2	2	2	2	2	2	2	2
31	30	Change from slurry to a solid manure handling system	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	31	Site solid manure heaps away from watercourses and field drains	80	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90	90
33	32	Site solid manure heaps on concrete and collect the effluent	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
34	33	Do not apply manure to high-risk areas	40	40	50	60	80	80	85	85	50	60	60	65	80	85	85	85	40	50	60	60	70	80	80	80
35	34	Do not spread farmyard manure to fields at high-risk times	70	75	75	75	80	85	85	85	70	75	75	75	80	85	85	85	70	75	75	75	80	85	85	85
36	35	Do not spread slurry or poultry manure to fields at high-risk times	60	80	80	80	60	90	80	90	60	75	70	80	50	90	70	90	60	85	80	85	60	85	80	85
37	36	Incorporate manure into the soil	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	80	70	75	75	75
38	37	Transport manure to neighbouring farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	38	Incinerate poultry litter	20	25	20	25	20	25	20	25	20	25	20	25	20	30	30	30	30	35	30	35	30	35	30	35
40	39	Fence off rivers and streams from livestock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
41	40	Construct bridges for livestock crossing rivers and streams	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
42	41	Re-site gateways away from high-risk areas	1	1	3	3	2	2	6	6	2	2	6	6	2	2	6	6	2	2	6	6	5	5	10	10
43	42	Establish new hedges	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
44	43	Establish riparian buffer strips	70	75	80	80	70	75	80	80	70	75	80	80	70	75	80	80	70	75	80	80	70	75	80	80
45	44	Establish and maintain artificial (constructed) wetlands	0	1	1	1	0	0	0	0	0	1	1	1	0	0	0	0	0	1	1	1	0	1	1	1

Table 3.3a Summary of expected implementation of mitigation methods for climate and soil zones within and outside NVZ and ECSFDI catchment boundaries in 2015. Results are expressed as a percentage of the arable land area affected.

Grassland (2015)

ID	Measure	Measure Name	Dry Clay Loam	Dry Clay Loam NVZ	Dry Clay Loam CSF	Dry Clay Loam CSF NVZ	Dry Sandy Loam	Dry Sandy Loam NVZ	Dry Sandy Loam CSF	Dry Sandy Loam CSF NVZ	Med Clay Loam	Med Clay Loam NVZ	Med Clay Loam CSF	Med Clay Loam CSF NVZ	Med Sandy Loam	Med Sandy Loam NVZ	Med Sandy Loam CSF	Med Sandy Loam CSF NVZ	Wet Clay Loam	Wet Clay Loam NVZ	Wet Clay Loam CSF	Wet Clay Loam CSF NVZ	Wet Sandy Loam	Wet Sandy Loam NVZ	Wet Sandy Loam CSF	Wet Sandy Loam CSF NVZ
0	0	Convert arable land to non cropped land	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1	1	Convert arable land to extensive grassland: including livestock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	2	Establish cover crops in the autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	3	Cultivate land for crop establishment in spring rather than autumn	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
4	4	Adopt minimal cultivation systems	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
5	5	Cultivate compacted tillage soils	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
6	6	Cultivate and drill across the slope	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
7	7	Leave autumn seedbeds rough	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
8	8	Avoid tramlines over winter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	9	Establish in-field grass buffer strips	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	10	Loosen compacted soil layers in grassland fields	1	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1	2	2	1	1	2	2
11	11	Maintain and enhance soil organic matter levels	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
12	12	Allow field drainage systems to deteriorate	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
13	13	Reduce overall stocking rates on livestock farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
14	14	Reduce the length of the grazing day or grazing season	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
15	15	Reduce field stocking rates when soils are wet	1	1	2	2	1	1	2	2	1	1	2	2	2	3	3	5	2	5	5	10	2	5	5	10
16	16	Move feed and water troughs at regular intervals	1	2	5	7	1	2	5	7	1	2	5	7	1	2	5	7	1	2	5	7	1	2	5	7
17	17	Reduce dietary N and P intakes	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
18	18	Adopt phase feeding of livestock	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
19	19	Use a fertiliser recommendation system	50	60	65	70	50	60	65	70	40	45	50	55	40	45	50	55	40	45	50	55	40	45	50	55
20	20	Integrate fertiliser and manure nutrient supply	50	60	65	70	50	60	65	70	40	45	50	55	40	45	50	55	40	45	50	55	40	45	50	55
21	21	Reduce N fertiliser application rates	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70
22	21	Reduce P fertiliser application rates	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70	50	60	65	70
23	22	Do not apply P fertilisers to high P index soils	50	60	65	70	50	60	65	70	40	45	50	55	40	45	50	55	40	45	50	55	40	45	50	55
24	23	Do not apply fertiliser to high-risk areas	50	60	65	70	50	60	65	70	40	45	50	55	40	45	50	55	40	45	50	55	40	45	50	55
25	24	Avoid spreading fertiliser to fields at high-risk times	60	65	70	75	60	65	70	75	60	65	70	75	60	65	70	75	60	65	70	75	60	65	70	75
26	25	Increase the capacity of farm manure stores	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30	15	25	25	30
27	26	Minimise the volume of dirty water produced	5	10	12	15	5	10	12	15	5	10	12	15	5	10	12	15	5	10	12	15	5	10	12	15
28	27	Adopt batch storage of slurry	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
29	28	Adopt batch storage of solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
30	29	Compost solid manure	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
31	30	Change from slurry to a solid manure handling system	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
32	31	Site solid manure heaps away from watercourses and field drains	65	70	75	80	65	70	75	80	65	70	75	80	65	70	75	80	65	70	75	80	65	70	75	80
33	32	Site solid manure heaps on concrete and collect the effluent	15	17	20	25	15	17	20	25	15	17	20	25	15	17	20	25	15	17	20	25	15	17	20	25
34	33	Do not apply manure to high-risk areas	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65
35	34	Do not spread farmyard manure to fields at high-risk times	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65	50	55	60	65
36	35	Do not spread slurry or poultry manure to fields at high-risk times	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60	60
37	36	Incorporate manure into the soil	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
38	37	Transport manure to neighbouring farms	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	38	Incinerate poultry litter	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
40	39	Fence off rivers and streams from livestock	5	5	10	10	5	5	10	10	5	5	10	10	5	5	10	10	5	5	10	10	5	5	10	10
41	40	Construct bridges for livestock crossing rivers and streams	2	2	5	5	2	2	5	5	2	2	5	5	2	2	5	5	2	2	5	5	2	2	5	5
42	41	Re-site gateways away from high-risk areas	1	10	1	10	2	10	1	10	1	10	1	10	1	10	1	10	1	10	1	10	1	10	1	10
43	42	Establish new hedges	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
44	43	Establish riparian buffer strips	3	3	8	8	3	3	8	8	3	3	8	8	3	3	8	8	3	3	8	8	3	3	8	8
45	44	Establish and maintain artificial (constructed) wetlands	1	1	2	2	1	1	2	2	1	1	2	2	1	1	2	2	1	1	2	2	1	1	2	2

Table 3.3b Summary of expected implementation of mitigation methods for climate and soil zones within and outside NVZ and ECSFDI catchment boundaries in 2015. Results are expressed as a percentage of the grassland area affected.

3.3 Point Phosphorus Loading

3.3.1 Septic Tanks

Phosphorus losses from households not connected to mains sewerage are potentially very significant in rural areas. It had previously been estimated that 750,000 properties are without mains sewerage in England, representing 4% of national connections (DWI0632, 1989).

Data on the number of properties connected for sewerage in 2001 were collated for each sewerage company area from annual reports made to the Office of Water Services (OFWAT; Table 3.4). These were compared to the total number of properties in each service area, estimated from Ordnance Survey Address Point data for 2001. The number of unconnected properties in each area was calculated by difference, and was found to be in the range 0.5 to 12% (Table 3.4). The unconnected properties were expected to be concentrated in rural areas. Previous analyses of service data for Scotland and the North West of England under earlier Environment Agency and SNIFFER funded projects for post-code group areas demonstrated that the percentage of properties connected increased with an increasing density of properties (Figure 3.6). The Countryside Agency and Office of the Deputy Prime Minister (ODPM) recently launched a national Rural and Urban Area Classification (Bibby and Shepherd, 2004) that defines settlement area types and sparsity of properties. The available service data for the North West region were summarised by each of the settlement types (Table 3.5). The percentage of properties unconnected varied from 27% in areas of sparse hamlets and isolated dwellings, to less than 0.1% within densely populated urban areas. The septic tank population within each service company area was therefore distributed across a map of the Rural and Urban Area Classification in proportion to the number of properties in each 1 km^2 cell and a weight was given to each of the area classification types based on the North West summary data (Table 3.5). It was assumed that each unconnected property was occupied by an average of 2.1 people. The total phosphorus load from the mapped septic tanks was estimated as 0.3 kg $capita^{-1}$ served by the tanks (Carvalho *et al*., 2005).

Table 3.4 Numbers of properties connected for sewerage within each service company area of England and Wales, and the estimated number of unconnected properties by differencing with the total property count (OFWAT, 2001).

Service Company	Connected Properties (10^6)	Total Property Count (10^6)	Unconnected Properties (%)
Anglian	2.4	2.5	0.5
Northumbrian	1.2	1.2	1.0
Severn Trent	3.7	3.9	4.9
South West	0.6	0.7	11.7
Southern	1.8	1.9	7.9
Thames	5.3	5.8	7.3
United Utilities	3.1	3.1	1.9
Welsh	1.3	1.5	8.4
Wessex	1.1	1.2	12.0
Yorkshire	2.1	2.1	0.2
England and Wales	22.7	23.9	5.1

Table 3.5 Average percentage of properties not connected to mains sewerage within the North West service area, by Rural and Urban Area Classification.

Rural and Urban Area Classification	Percent of Properties Unconnected
Hamlet & Isolated Dwelling - Sparse	27
Hamlet & Isolated Dwellings - Less Sparse	16
Village - Sparse	14
Village - Less Sparse	10
Town & Fringe - Sparse	0.3
Town & Fringe - Less Sparse	1.3
Urban >10k - Sparse	0.1
Urban >10K - Less Sparse	0.1

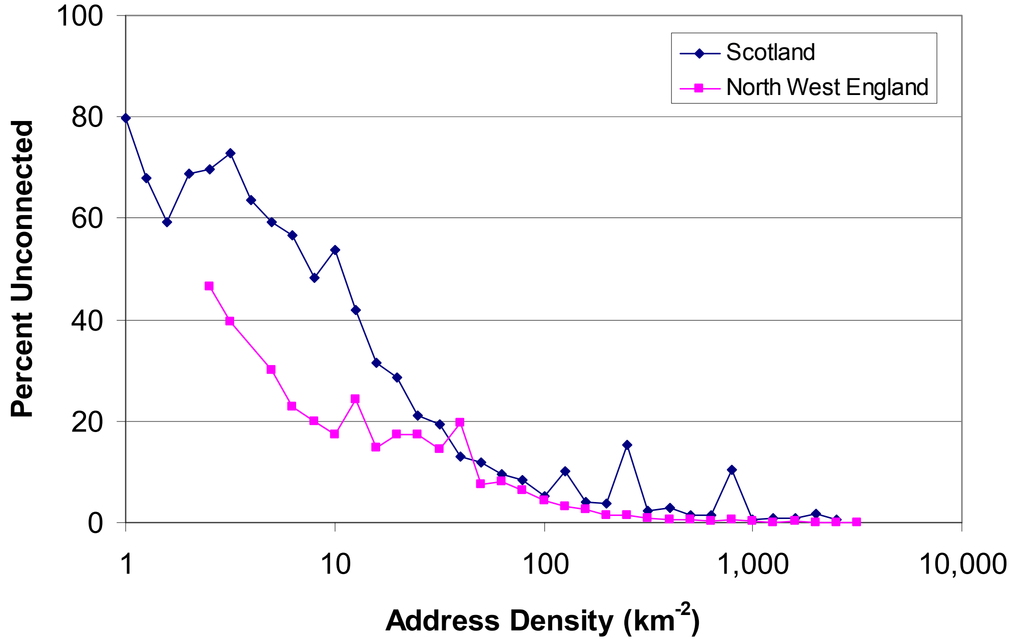

Figure 3.6 Calculated percentage of properties unconnected for sewerage as a function of property density by post-code group areas within Scotland and the North West of England.

3.3.2 Sewage Effluent Discharges

Phosphorus in sewage effluent discharges was calculated using data provided by the Environment Agency. The database had been developed by the Water Research Centre (WRc) for the Agency as part of a national water quality model framework (Kelly *et al*., 2006). The Environment Agency database provided estimates of consented annual average flows for *c*. 3,550 effluent discharges. This database included only discharges with a population equivalent greater than 250. Previously, Anthony and Lyons (2006) constructed a database of only sewage effluent discharges for England and Wales using the Environment Agency consents register as of 2003. This database did not include any discharge threshold. The effluent discharge contributed by the smaller works was less than 5% of the total for England and Wales.

The database also provided estimates of annual average orthophosphate concentrations in the effluent based on compliance monitoring. These data were not available for every discharge. Missing concentration data were therefore replaced with averages, derived separately for each Environment Agency region and for effluent discharges in three flow bands (Tables 3.6 and 3.7). The flow bands were centred on population equivalents in the ranges 0 to 2,000; 2,000 to 10,000; and greater than 10,000.

The impact of tertiary phosphorus stripping was calculated using an Environment Agency database that identified 227 individual sites for which treatment was in place by the end of 2005 (Nowosielski, *pers. comm*.). The database gave maximum permitted total phosphorus concentrations in the effluent discharge that were used in place of the compliance monitoring data as appropriate.

The total phosphorus load in the sewage effluent discharges was calculated on the basis of an average ratio of orthophosphate to total phosphate in effluent discharges of 0.85 (Demars *et al*., 2005). The calculated annual average total phosphorus load to all lakes and rivers in England and Wales was 20.2 kt P. The calculated load was reassuringly close to the estimate of 18.2 kt P developed by Anthony and Lyons (2006) who had independently integrated available Environment Agency consents and compliance monitoring data. Use of the Environment Agency discharge database in this project will ensure consistency with on-going work in the Agency to further characterise diffuse pollution pressures.

Table 3.6 Total numbers of fresh water sewage effluent discharges and total consented annual average discharge by Environment Agency region (Environment Agency; data for 2000 to 2004).

EA Region	Number of Sewage Discharge Consents	Consented Average Discharge Ml day^{-1}	Consented Annual Average Discharge 10^9 m^3
Anglian	675	1383	0.50
Wales	348	683	0.25
Midlands	644	3223	1.18
North East	433	2623	0.96
North West	319	2495	0.91
South West	535	1608	0.59
Southern	233	673	0.25
Thames	343	2387	0.87

Table 3.7 Total numbers of fresh water sewage effluent discharge consents matched to measured effluent quality data, and the annual average orthophosphate concentrations at the matching sites (Environment Agency; data for 2000 to 2004).

a) Numbers of fresh water sewage effluent discharge consents matched to measured effluent quality data.

EA Region	Population Equivalent <2000	2,000-10,000	>10,000
Anglian	263	241	95
Wales	173	85	41
Midlands	258	165	141
North East	30	45	89
North West	105	62	102
South West	32	38	63
Southern	65	84	60
Thames	110	87	92

b) Annual average orthophosphate concentrations at the matching sites.

EA Region	Population Equivalent <2000 (mg l^{-1} P)	2,000-10,000 (mg l^{-1} P)	>10,000 (mg l^{-1} P)
Anglian	6.1	5.9	5.2
Wales	4.7	4.0	3.4
Midlands	6.5	6.2	4.3
North East	5.1	4.8	2.9
North West	4.8	4.4	3.0
South West	5.4	6.0	2.1
Southern	6.0	5.0	2.6
Thames	5.3	5.2	3.8

3.3 Retention

A proportion of the phosphorus input to the contributing lake catchment area is retained in the river system above the lake, for example in storage in bed sediments, and not delivered to the lake. This would not normally be significant for small river systems. The annual average proportion of phosphorus inputs retained was estimated using the empirical model of Behrendt and Optiz (2000). This model estimates the proportion of total phosphorus retained, without regard to particulate and soluble proportions, and is based on statistical analyses of modelled total phosphorus inputs and measured outputs from *c*. 100 catchments in north west Europe. The model requires estimates of the river and lake surface area within a catchment. River surface area is estimated from total catchment area (Venohr *et al*., 2005) and lake surface area was taken from the GB Lakes Inventory (Hughes *et al*., 2004). Modelled retention for individual lake catchments was generally in the range 5 to 40% with the highest values associated with large catchments of low annual rainfall and other lakes located in the headwaters.

4. Model Validation

The regular monitoring of lake water quality in England and Wales is limited in comparison to the monitoring of river systems. Phosphorus measurements are taken regularly at *c*. 7,000 river monitoring sites by the Environment Agency (EA) whereas the GB Lakes inventory contains measurements for only *c*. 200 lakes, the majority of which were the result of specific and short-lived investigations. The available measurements might therefore be unrepresentative of the majority of lakes modelled from the GB Lakes database, and possibly also influenced by changes in lake management such as the introduction of phosphorus stripping at important water bodies. The order of magnitude uncertainty in the predictions from the OECD (1982) model meant that we also could not expect accurate predictions for the individual lakes for which we had measurement data. Our intention by model validation was therefore only to demonstrate that the output from the model framework was generally unbiased and appropriate for calculating population statistics at the regional and national scale.

Two data sets were potentially available for validation of the phosphorus lake loads and prediction of annual average concentrations. The first was a set of average concentrations reported by Carvalho *et al*. (2003) in the final report of the NUPHAR project (Appendix 2). Values provided were taken from the GB Lakes inventory and were annual averages based on at least four regularly spaced sampling occasions throughout the year. The data were provided only for lakes with measured depth data. A second data set of predominantly Environment Agency data was provided by the Centre for Ecology & Hydrology and dates from 2005. This data set was potentially more appropriate for comparison with catchment predictions made after implementation of phosphorus stripping in the period 2001 to 2005 (see Section 3.3.2). The annual averages in this dataset were derived by first calculating quarterly averages, and then taking an average of the quarterly values. This was intended to give a best estimate of the annual average that was not biased by more frequent sampling of water quality in the spring and summer months.

The 2005 data were predominantly sourced from the Environment Agency. Recent concerns have, however, been raised over the quality of these data, specifically in relation to detecting total phosphorus at concentrations below 100 µg l^{-1} P that are characteristic of natural lake waters (UK TAG, *pers. comm.*). This is due to the Environment Agency's traditional laboratory calibrations that were set up to measure total phosphorus in sewage effluent and rivers, generally at much higher concentrations than in lakes. For this reason, the model validation with 2005 data was rejected. The earlier measured data taken from the GB Lakes inventory was largely derived from CEH and university laboratories which have the necessary low detection limits for measuring phosphorus in natural lake waters. For this reason, only these were used for model validation.

The NUPHAR project had previously identified that White Mere (35091) was trap for phosphorus with no outflow (Moss *et al*., 1997) and had removed the data from the validation dataset (Carvalho *et al*., 2003). A number of other lakes were removed from the validation dataset by this project. Data for Lyn Penrhyn (32968) and Grafham Water (38310) were excluded as the measurement data pre-dated implementation of phosphorus stripping at contributing sewage effluent works. Similarly, data for Rostherne (32650), Alderfen Broad (35791) and Walsham (36143) were excluded as sewage effluent had been re-directed out of the catchment. Finally, an unusual case at Seathwaite Tarn (29290) was also excluded. This lake was the subject of an acidification experiment between 1990 and 1993 in which phosphate fertiliser was added to the lake to raise the pH of the water. The author of a paper on the research observed that the lake became very eutrophic (May, 2004). Finally, Hanmer Mere (34780) was also excluded from the analysis.

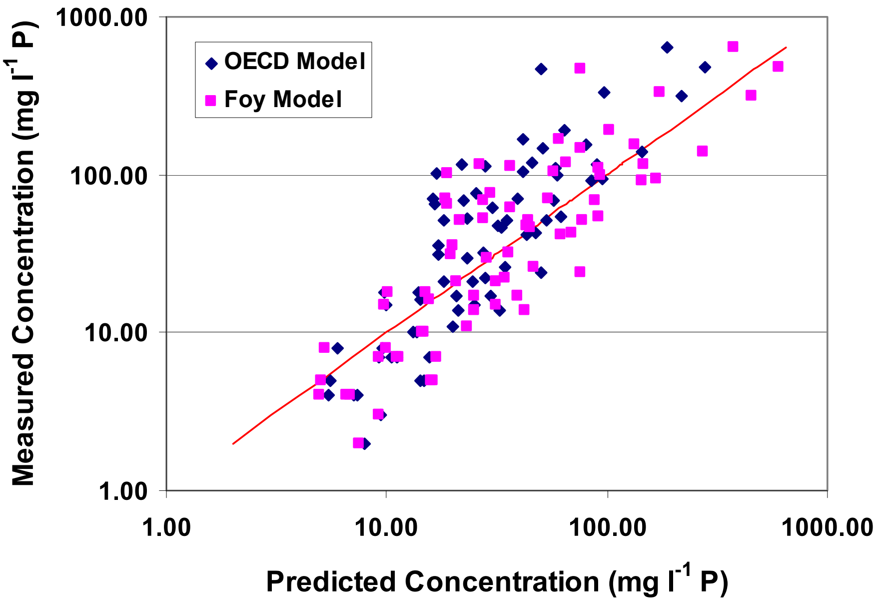

Figure 4.1 Baseline modelled and measured annual average total phosphorus concentrations using the OECD (1982) and Foy (1992) models for selected lakes in the GB Lakes inventory.

There were a total of 72 lake measurements remaining that were compared with concentration predictions using the OECD model (Section 2.1) and agricultural census and practice data for the year 2004 (Section 3.2.4). A strong positive correlation was observed between the predicted and measured annual average total phosphorus concentrations using the OECD model (Figure 4.1), explaining 71% of the observed variance (EQ 2). The regression model intercept was only just statistically significant (*P* 0.043) and when removed the slope of the model was 1.088(±0.026) indicating that the model under-estimated at high measured concentrations. Application of the Kolmogorov-Smirnov test indicated that the statistical distributions of modelled and measured phosphorus concentrations were not significantly different ($P > 0.20$).

(EQ 2) $\mathrm{Log}_{10}(\text{Measured TP}) = 1.283(\pm 0.098) \times \mathrm{Log}_{10}(\text{Modelled TP}) - 0.299(\pm 0.145)$

Alternatively, the draft guidance on lake assessment provided by the UK-TAG (2003; WP7a) reported that experience in parts of the United Kingdom suggested that the OECD equation over-estimated retention and therefore under-estimated concentrations and risk. An alternative equation derived from lakes in Northern Ireland was recommended for the risk assessment in the preliminary phases of catchment characterisation (Foy, 1992):

(EQ 3)
$$C = \frac{1.118 \cdot \dfrac{10^5 \cdot L}{H}}{\left(1 + \sqrt{t}\right)^{1.135}}$$

where C is the predicted concentration ($\mu g\ l^{-1}$), L is the modelled total phosphorus loss from all land in the catchment (kg P ha^{-1}), H is the modelled average catchment drainage (mm), and t is the average hydraulic residence time of the lake (yr).

This model was also applied to the calibration dataset and resulted in significantly higher predicted concentrations for the most polluted of lake systems (Figure 4.1), again explaining 71% of the variance (EQ 4). The regression model intercept was also not significant and when removed the slope of the model (1.004±0.023) was not significantly different from one. The Foy (1992) retention model was a better fit to the measured data. However, the retention estimates for the lake systems due to the Foy (1992) model were very low and included one negative value that could not be physically justified.

(EQ 4) $Log_{10}(\text{Measured TP}) = 1.041(\pm 0.079) \times Log_{10}(\text{Modelled TP}) - 0.062(\pm 0.128)$

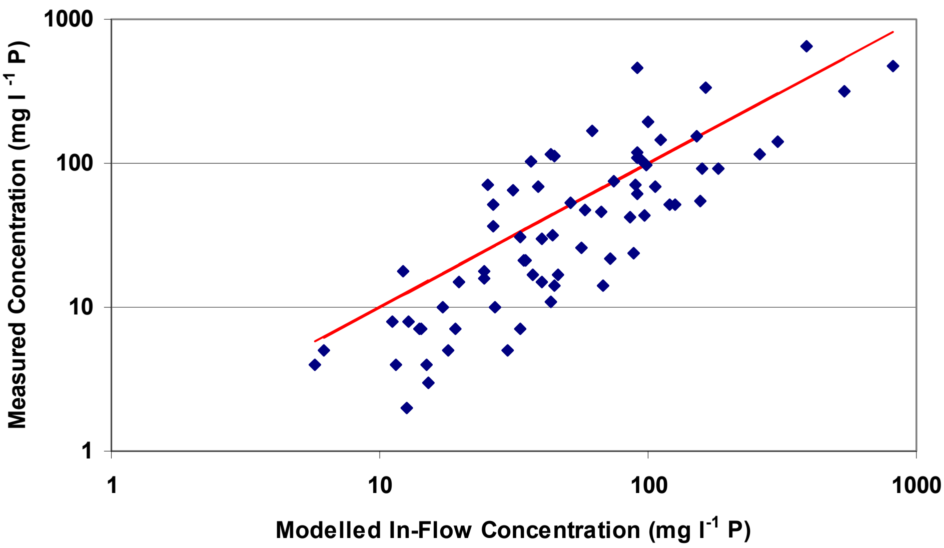

Figure 4.2 Modelled in-flow phosphorus concentration (prior to in lake retention) and measured average total phosphorus concentrations in lake waters, for selected lakes in the GB Lakes inventory.

The model prediction errors were within the five-fold prediction intervals associated with the original OECD (1982) model derivation. Furthermore, the Environment Agency has adopted the OECD (1982) model for their independent lake nutrient eutrophication risk assessment as part of the River Basin Characterisation process. We therefore selected to use the OECD (1982) model for the remainder of this analysis. One consequence of this was a potential under-estimate lake phosphorus concentration in the more polluted systems.

The under-estimation of measured phosphorus concentrations may have reflected an under-estimation of modelled phosphorus loads by the PSYCHIC model which is known to predict lower loads than published export coefficient models (see Section 6). However, the under-estimation was apparent only for the most polluted lake systems where the calculated dominant sources of phosphorus were known to be either sewage effluent outfalls, septic tanks or diffuse urban runoff. It was possible that the phosphorus loads from these sources had been under-estimated in these specific catchments due to the statistical average data used to characterise effluent discharges and concentrations, density of septic tanks and the connectivity of urban areas.

To attempt to better understand the causes of the under-estimation, measured concentrations were plotted against calculated phosphorus concentrations in the inflow prior to any in-lake retention (Figure 4.2). There ought to be no measured values greater than the modelled values unless there was a significant net release of phosphorus from the lake sediments associated with historically higher loadings. The lake with the largest relative under-estimate of phosphorus loading was Fleet Pond (43315), the largest fresh water lake in Hampshire. This lake is immediately adjacent to a large urban development and it is possible that the urban component has been under-estimated. The lake is also very shallow and subject to siltation problems. The other problem lakes included Ranworth Broad (36050) and Hoveton Little Broad (35923) in Norfolk.

It was concluded that the uncertainty in model predictions was not with the OECD (1982) retention model but with the estimate of the total loads input to the lake systems. Given the small number of lakes in the validation dataset, it was not possible to demonstrate conclusively that the model framework has provided unbiased estimates of total phosphorus loads to the lake systems.

5. Model Application

The PSYCHIC model was used to predict total phosphorus losses from diffuse agricultural sources in 2004 and 2015. These losses were combined with fixed losses for 2004 from urban areas, atmospheric deposition, bank erosion, septic tanks and sewage effluent discharges (Sections 3.2 and 3.3). The loads were input to the OECD (1982) model to predict lake total phosphorus concentrations and infer lake ecological status according to type specific standards.

5.1 Baseline Results

The baseline results are for the year 2004. They represent best estimates for phosphorus loads under present day agricultural practices, crop areas and livestock numbers. The phosphorus loads calculated for sewage effluent discharges are consistent with the inventory used by the Environment Agency for river water quality modelling. The loads calculated for diffuse urban runoff follow a methodology that has been developed as a generic risk assessment methodology for urban areas (Mitchell, 2005).

5.1.1 Load Apportionment

The total annual phosphorus load from all sectors to rivers and lakes across England and Wales was calculated to be 27.4 kt P, equivalent to 1.9 kg ha^{-1} P. The calculated total pollutant loads are summarised by sector and Environment Agency region in Table 5.1. It is shown the sewage effluent discharges are responsible for 74% of the total phosphorus input to lakes and rivers across England and Wales, and agriculture is responsible for only 18%. This is consistent with earlier work reported by Anthony and Lyons (2006) that used an independent working of the effluent consents and compliance monitoring databases (Section 3.3.2), and also by White and Hammond (2007) that used a per capita export coefficient model. The diffuse total phosphorus contribution varied spatially in association with the intensity of agriculture and population density from 20% in East Anglia to 50% in Wales.

In contrast to this national source apportionment, diffuse agricultural (including forestry) sources are the dominant sources of phosphorus within the lake catchments, with diffuse urban runoff the next most important source for catchment areas less than 10 km^2 (Figure 5.1). This is a consequence of effluent discharges generally being positioned on main river systems, rather than the tributaries associated with the lake systems. Figure 5.2 maps the dominant phosphorus source within each of the modelled lake catchments. Atmospheric deposition is significant only for the smallest lake catchments in upland areas; and bank erosion, groundwater, and sewage effluent discharges are dominant in a scattering of catchments. Diffuse urban runoff and septic tanks are dominant in areas of high population density, principally in the Midlands and South East. Overall, groundwater, bank erosion and atmospheric deposition each account for only a small percentage (generally less than 10%) of the total phosphorus input to each lake catchment (Figure 5.3). The calculated septic tank contribution is comparable to the range of 1.2 to 22% reported for 8 lake studies in Carvalho *et al.* (2003).

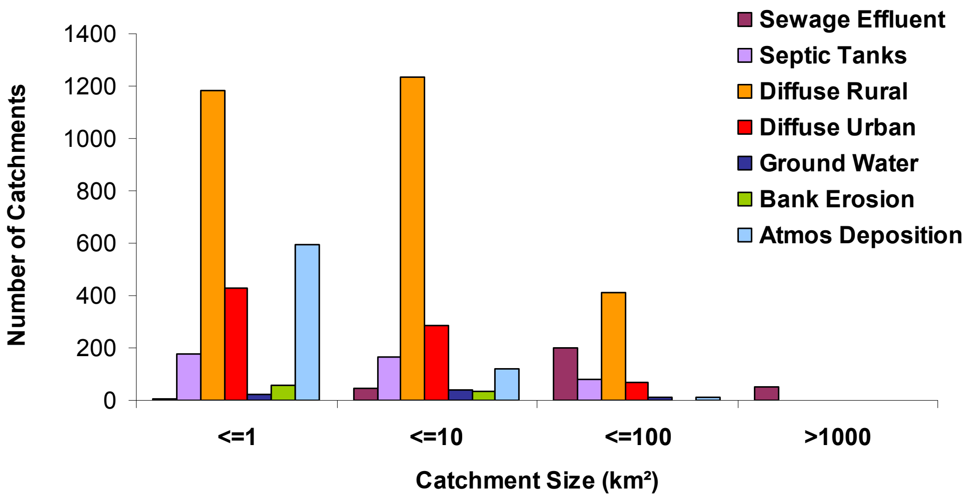

Figure 5.1 Count of modelled lake catchments in which each of the modelled pollutant sectors is the dominant source, by catchment size across England and Wales.

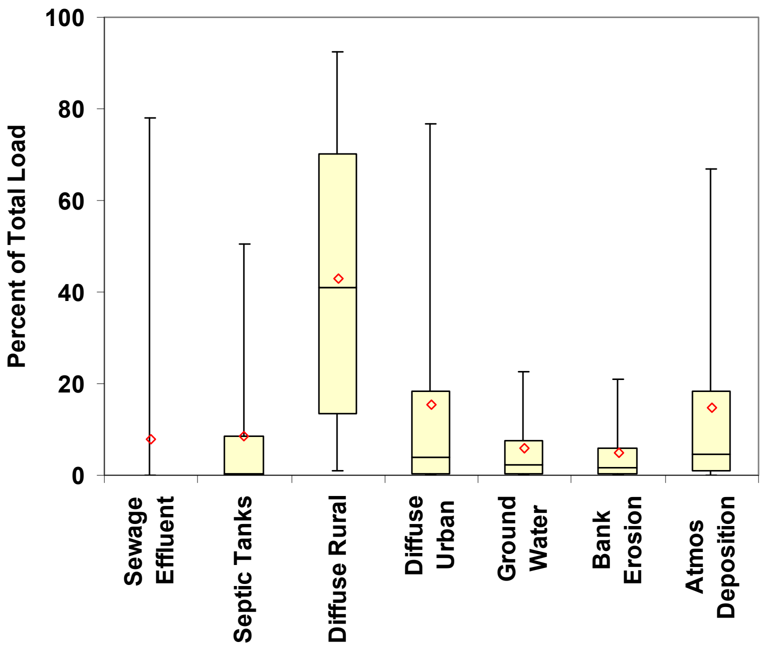

Figure 5.3 Range of percentage contributions to the total modelled phosphorus load input to the modelled lake catchments across England and Wales. Bars and whiskers mark the inter-quartile range, and the minimum and maximum values.

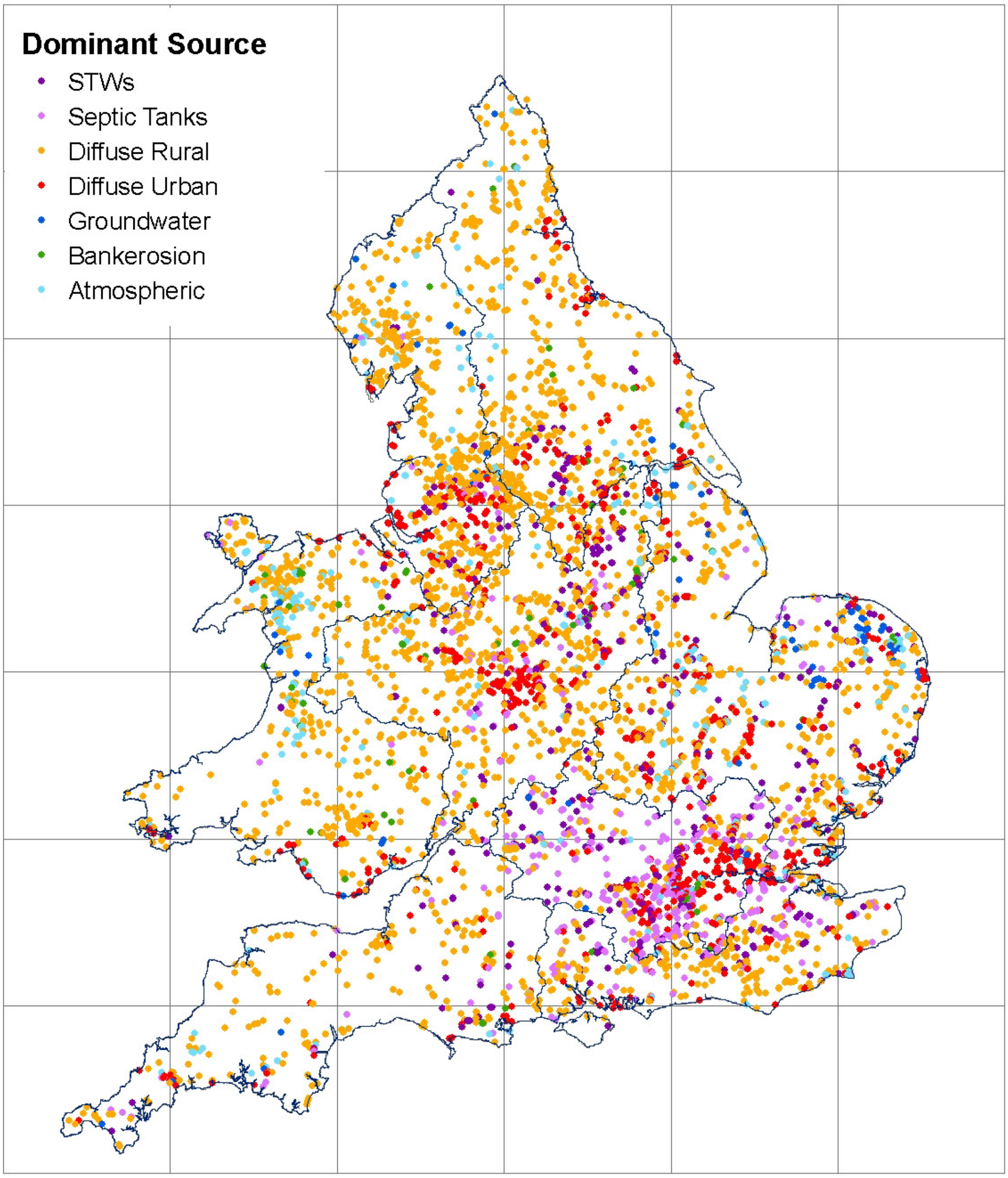

Figure 5.2 Dominant source of phosphorus input to the modelled lake catchments across England and Wales in 2004.

a)

	Sewage Effluent kg ha^{-1} y^{-1}	Diffuse Rural kg ha^{-1} y^{-1}	Diffuse Urban kg ha^{-1} y^{-1}	Septic Tanks kg ha^{-1} y^{-1}	Ground Water kg ha^{-1} y^{-1}	Bank Erosion kg ha^{-1} y^{-1}	Atmos. Depos. kg ha^{-1} y^{-1}
Anglian	0.85	0.19	0.03	0.01	0.01	0.00	0.002
Wales	0.52	0.45	0.04	0.04	0.05	0.04	0.005
Midlands	2.55	0.38	0.07	0.05	0.01	0.02	0.003
North East	1.57	0.40	0.05	0.01	0.02	0.02	0.002
North West	2.26	0.57	0.10	0.03	0.03	0.02	0.006
South West	0.91	0.34	0.04	0.07	0.04	0.02	0.002
Southern	0.81	0.27	0.06	0.11	0.01	0.01	0.002
Thames	2.02	0.15	0.12	0.18	0.01	0.01	0.003
England and Wales	1.40	0.34	0.06	0.05	0.02	0.02	0.003

b)

	Sewage Effluent t y^{-1}	Diffuse Rural t y^{-1}	Diffuse Urban t y^{-1}	Septic Tanks t y^{-1}	Ground Water t y^{-1}	Bank Erosion t y^{-1}	Atmos. Depos. t y^{-1}
Anglian	2183	476	77	31	19	9	5
Wales	1031	898	87	71	97	89	10
Midlands	5310	787	145	109	28	50	6
North East	3456	877	115	15	35	53	5
North West	3088	781	141	40	39	29	8
South West	1774	672	77	136	70	39	4
Southern	823	279	66	112	15	9	2
Thames	2492	184	144	217	15	15	4
England and Wales	20157	4953	851	730	318	292	44

Table 5.1 Summary of calculated total phosphorus loads by source and Environment Agency region, with agricultural census and practices for 2004. Loads are expressed as a) a load per hectare of all land in England in Wales, or b) as a total load per year.

5.1.2 Phosphorus Concentrations

Figure 5.4 maps the modelled annual average total phosphorus concentrations in each of the modelled lake systems for 2004. Modelled concentrations are obviously higher in areas of high population density and areas of high phosphorus loads from agricultural land (see Figure 3.1). Overall, only 22% of modelled lake phosphorus concentrations exceeded 100 $\mu g\ l^{-1}$ P. This is in contrast to 70% of measured lake concentrations from three studies located in England, the West Midland Meres and the south-east of England (Carvalho and Moss, 1995; Moss *et al*., 1994; Bennion, 1994 respectively).

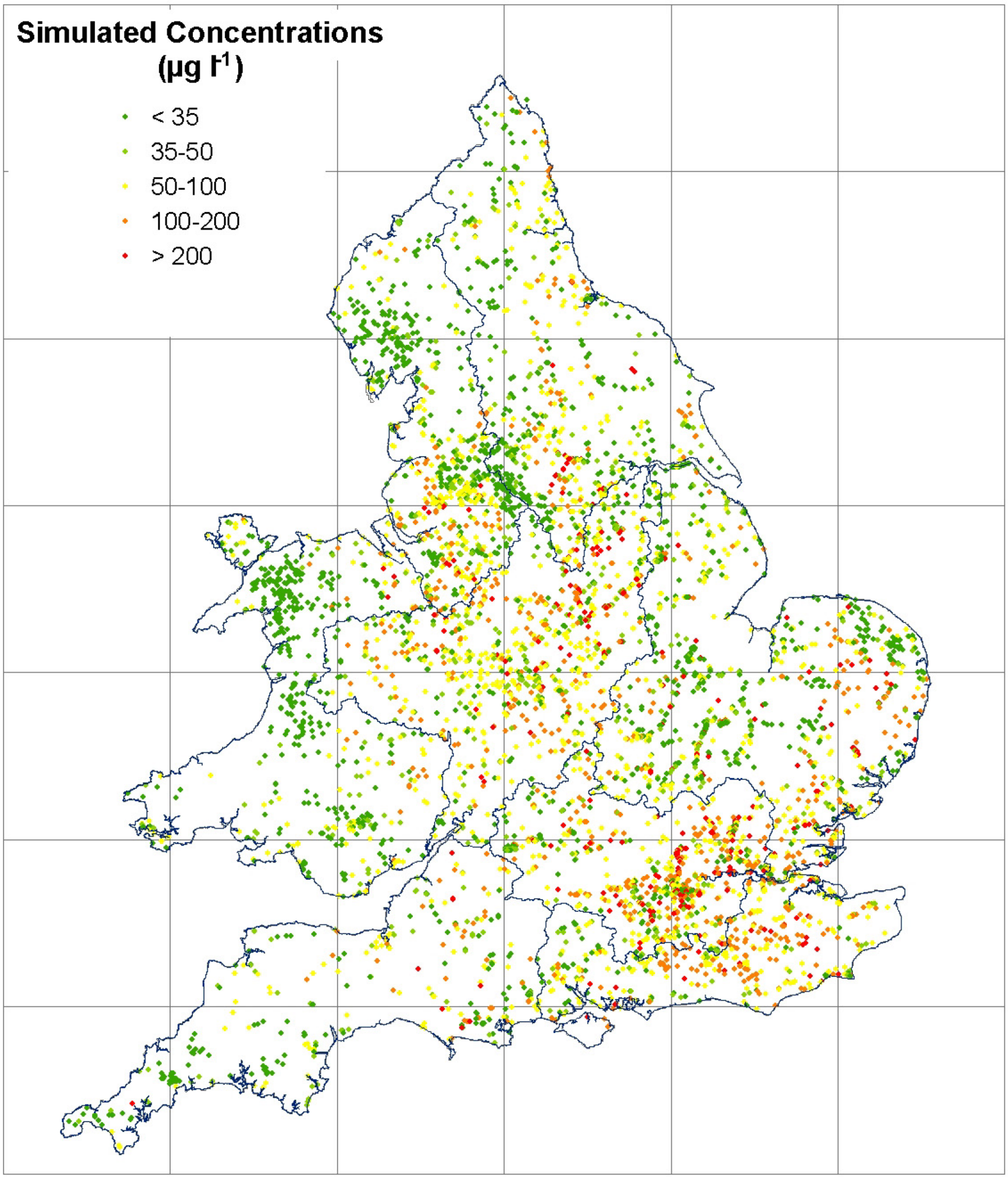

Figure 5.4 Distribution of modelled annual average total phosphorus concentrations in lake systems, under 2004 agricultural census and implementation of mitigation methods.

Type	Count	High	Good	Moderate	Poor	Bad	Fail GES
LAVS	386	3	7	25	33	32	90
LAS	397	10	22	37	24	6	68
LAD	17	12	29	47	12	0	59
MAVS	330	4	9	26	31	30	87
MAS	181	6	14	35	23	22	80
MAD	4	0	0	75	25	0	100
HAVS	3276	26	14	33	19	9	61
HAS	777	12	10	28	36	15	78
HA unclass.	1	0	0	100	0	0	100
MarIVS	43	0	21	40	23	16	79
MarIS	124	6	27	23	20	24	68

Table 5.2 Percentage of all modelled lake systems by type and status class, and the percentage failing good ecological status standards, under 2004 agricultural census and practices.

Type	Count	High	Good	Moderate	Poor	Bad	Fail GES
LAVS	386	3	7	26	32	32	90
LAS	397	11	22	37	24	6	67
LAD	17	12	41	35	12	0	47
MAVS	330	4	9	27	30	29	86
MAS	181	6	15	38	20	20	79
MAD	4	0	0	75	25	0	100
HAVS	3276	27	14	33	18	8	59
HAS	777	13	11	27	36	13	76
HA unclass.	1	0	0	100	0	0	100
MarIVS	43	0	23	40	21	16	77
MarIS	124	6	30	23	22	19	65

Table 5.3 Percentage of all modelled lake systems by type and status class, and the percentage failing good ecological status standards, under 2015 agricultural census and practices.

5.1.3 Ecological Status

Modelled concentrations were compared with the relevant phosphorus thresholds for good ecological status (Section 2.2). Table 5.2 summarises the percentage distribution of modelled lake systems achieving phosphorus standards for good ecological status under present day agricultural census and practices (year 2004). Overall 68% of lake systems were predicted to fail phosphorus standards for good ecological status. Figure 5.5 maps the distribution of modelled lake status. In general low alkalinity lakes were less at risk of failing good ecological status than medium and high alkalinity lakes.

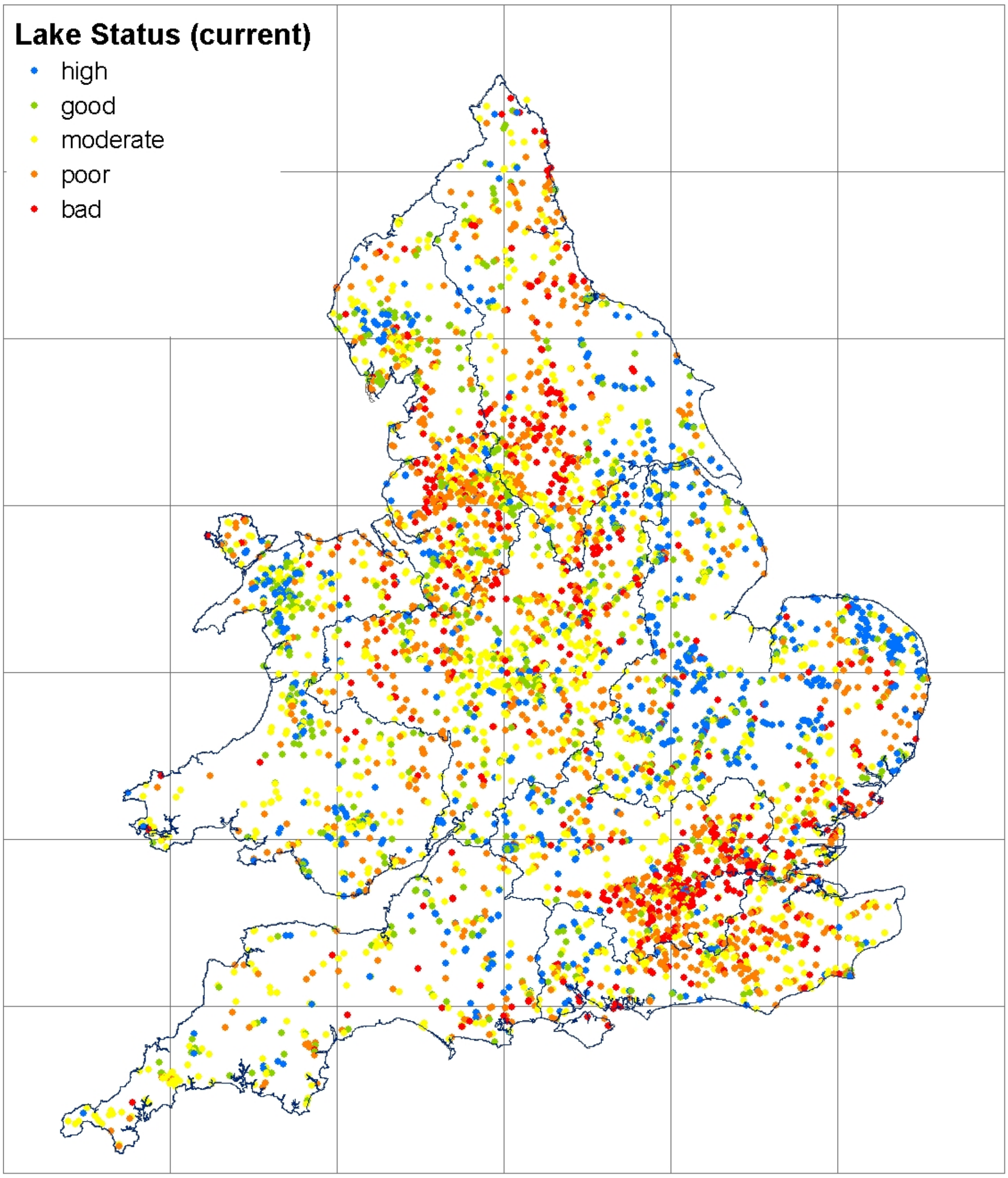

Figure 5.5 Distribution of modelled lake status, under 2004 agricultural census and implementation of mitigation practices.

5.2 Scenario Results

The scenario results are for the year 2015. The scenario results for ecological status are summarised by country and for each River Basin District, and are also reported separately for the lakes with a surface area greater than 50 ha. The Water Framework Directive requires that lake status need only be reported for these larger water bodies.

5.2.1 Ecological Status

Modelled concentrations were compared with the relevant phosphorus thresholds for good ecological status (Section 2.1). Table 5.3 summarises the percentage distribution of modelled lake systems achieving phosphorus standards for good ecological status under forecast agricultural census and practices (year 2015). Although a 10% reduction in agricultural phosphorus losses was forecast between 2004 and 2015 this has had a negligible impact on overall lake status. Overall 67% of all modelled lake systems were predicted to fail phosphorus standards for good ecological status. This highlights a very poor response of the lake population in England and Wales to reductions in agricultural phosphorus load.

The Water Framework Directive only requires reporting of status for lakes with a surface area greater than 50 ha. There are 130 lakes in England and Wales meeting these criteria in the lake database (Table 5.4). There are regional patterns in the percentage of all lakes failing good ecological status. The lowest rates are in the Anglian (46%) and Western Wales (55%) River Basin Districts, and the highest in Northumbria (77%) and Thames (77%). The results for all lakes are generally matched by the results for the smaller set of WFD reporting lakes (Table 5.4). Overall, it was calculated that 68% of WFD reporting lakes (> 50 ha) in England failed good ecological status and 49% of the lakes in Wales.

Table 5.4 Number of lakes in each River Basin District and by surface area class across England and Wales, and the percentage that are forecast to fail phosphorus concentration standards for good ecological status in 2015.

River Basin District	Surface Area < 50 ha			Surface Area >= 50 ha		
	Pass	Fail	Fail Rate	Pass	Fail	Fail Rate
Anglian	545	470	46.3	11	6	35.3
Dee	18	51	73.9	0	4	100.0
Humber	322	717	69.0	4	23	85.2
North West	158	478	75.2	6	13	68.4
Northumbria	40	136	77.3	4	5	55.6
Severn	169	363	68.2	9	8	47.1
Solway Tweed	19	28	59.6	1	1	50.0
South East	85	225	72.6	0	2	100.0
South West	78	188	70.7	2	7	77.8
Thames	220	775	77.9	2	10	83.3
Western Wales	144	176	55.0	5	7	58.3

The proportion of lake systems that failed to achieve good ecological status also varied with the dominant source within the catchment (Table 5.5). The fail rate was greatest in catchments dominated by sewage effluent (99%) and septic tanks (79%) and least in the catchments dominated by ground water (11%) and atmospheric deposition (24%). It is significant to note that phosphorus losses in catchments dominated by diffuse urban sources were sufficient to result in a high percentage of fails. The calculated average phosphorus loss in diffuse urban runoff 0.7 kg ha $^{-1}$ P (Section 3.2.2) is comparable to measured losses from agricultural land. This estimate has excluded losses from minor roads in rural areas, and so does not risk double accounting for losses from agricultural fields that are routed to water bodies via roads. The value is subject to local uncertainty, as the estimation of the connectivity of impervious areas to water bodies will depend on the specific design of the storm and sewerage system. However, it is evidence for the importance of controlling urban losses with Sustainable Urban Drainage Systems (SUDS; see for example, Jefferies, 2004) in addition to mitigation of agricultural contributions.

Table 5.5 Number of lakes in which each phosphorus source is dominant, and the percentage of the lakes that are forecast to fail the relevant standards for good ecological status in 2015.

Source	Count	Fail Rate
Sewage Effluent	536	99.4
Septic Tanks	439	79.7
Diffuse Rural	2,879	68.5
Diffuse Urban	785	76.8
Ground Water	77	11.7
Bank Erosion	91	52.7
Atmos Deposition	729	24.4

The percentage phosphorus load reduction required to achieve good ecological status was calculated for each of the failing lake systems. This calculation assumed either a) that only the diffuse agricultural (including forestry) load was reduced (Figure 5.6); or b) that the load from all sectors was reduced equally (Figure 5.7). The analysis reveals that if the agricultural load were completely removed, then *c*. 50% of lakes would continue to fail good ecological status. The remaining lakes are most likely located in urban and semi-urban areas with a high diffuse urban runoff component to the total load. More modest and feasible agricultural load reductions of up to 25% would result in a maximum 10% improvement in the proportion of lakes that are forecast to achieve good ecological status in 2015. It is shown that the WFD reporting lakes are more likely to benefit from reductions in the agricultural sector load than the typical lake.

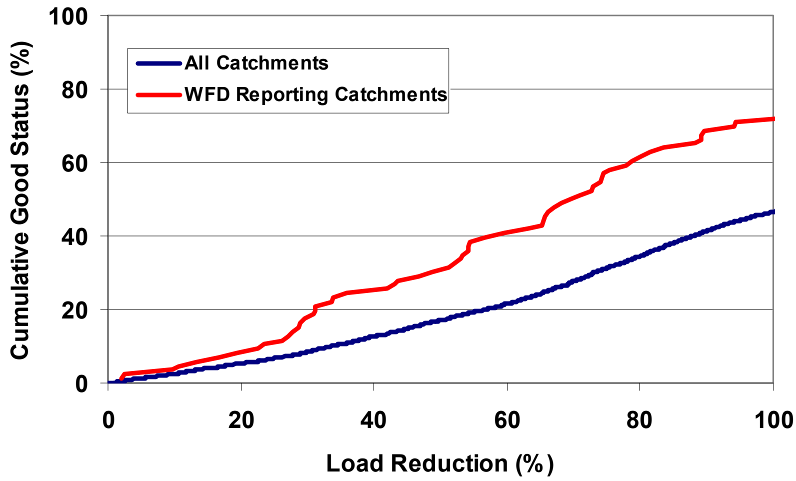

Figure 5.6 Percent of lake catchments forecast to fail good ecological status in 2015 that subsequently pass as a result of a reduction in the phosphorus load from the agricultural sector, summarised across England and Wales.

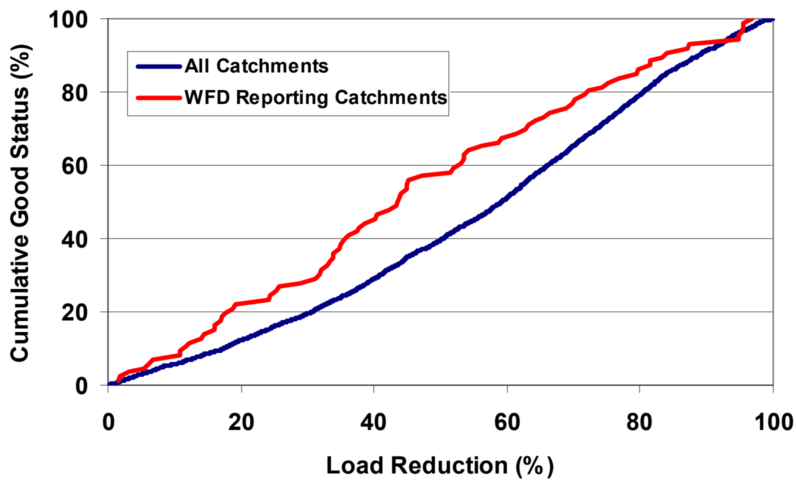

Figure 5.7 Percent of lake catchments forecast to fail good ecological status in 2015 that subsequently pass as a result of a reduction in the phosphorus load from all sectors, summarised across England and Wales.

6. Discussion

A key outcome of this analysis has been a status evaluation for 5,536 lakes in England and Wales. This could not have been achieved based on available monitoring data. The Environment Agency presently monitors 109 lakes in England and Wales as part of a pilot lake monitoring network. The UKTAG (TAG/LTT 120; 2008) has also reported that current Environment Agency analytical detection limits are too high for low alkalinity lake types. Measurements for some lakes in Cumbria have significantly higher reported total phosphorus values than those reported in the literature, and it is likely that compliance for lakes in Wales and north-west England is not as low as reported from the available monitoring data. The results of this analysis have generally confirmed previous model assessments (Carvalho *et al*., 2003) and analyses of available Environment Agency monitoring data. Overall, 66% of all lakes are forecast to fail standards for good ecological status in 2015. This assessment has taken account of forecast changes in the structure of agriculture, and the uptake of mitigation methods by landowners in response to existing policy instruments.

The analysis was model based and dependent on national agricultural and environment datasets. It is normal to expect uncertainty in the input data and model predictions. We have calculated that the agricultural sector is the dominant source of phosphorus in 70% of the WFD reporting catchments. The analysis is, therefore, most sensitive to the agricultural sector estimates.

The small catchment areas typical of the lakes in the GB Lakes inventory can result in uncertainty in the mapping of agricultural land use and especially livestock numbers. Previous analyses by ADAS of the mapped agricultural census data against ground truth have indicated uncertainties of the order $\pm$10% for crop areas and $\pm$25% for livestock numbers in catchment areas of 5 by 5 km^2. In comparison, the median catchment area for the WFD reporting lakes is 28 km^2. However, these uncertainties are much less important for the calculation of population statistics than they are for making predictions for individual lakes. Providing that there is no error in the phosphorus loss models, it is believed that the estimates of the percentage of lakes at risk are robust due to the very large number of lakes modelled (n = 5,536).

This analysis has used the PSYCHIC model of diffuse agricultural sources (Davison *et al*., 2008). Its use ensured consistency with previous gap analyses for Defra (Anthony, 2006; Anthony and Lyons, 2006), and allows for sensitivity to environmental factors such as rainfall, soil type and drainage density. The model is subject to on-going development and validation at farm, catchment and national scales (Anthony and Lyons, 2006; Collins *et al*., 2007). Phosphorus losses predicted by the model are lower than previous published model estimates.

The diffuse source loss of phosphorus to rivers has previously been estimated as 1.24 kg ha^{-1} P from the arable and improved grassland area using the AERC Export Coefficient Model (Haygarth *et al*., 2003). White and Hammond (2007) used a variant of the same model to calculate average losses from agricultural land including rough grazing of 0.7 and 1.1 kg ha^{-1} P for England and Wales respectively. A model developed by the SSLRC predicted a national average loss of 1.25 kg ha^{-1} P to the edge of field (Defra project NT1003). A simple inventory model developed by ADAS estimated phosphorus losses to the field edge as 1.4 kg ha^{-1} P (Defra project NT1022). Allowing for an average landscape delivery factor of 40% gives a national average loss of 0.60 kg ha^{-1} P from the arable and improved grassland area (Defra project PE0105).

These results are generally higher than the baseline PSYCHIC model prediction of 0.44 kg ha^{-1} P from all agricultural land (including rough grazing and woodland) in England and Wales, excluding the calculated loss to groundwater. The PSYCHIC model prediction is equivalent to 0.62 kg ha^{-1} P when expressed per hectare of arable and improved grassland only. The additional load from losses to groundwater may be equivalent to *c*. 0.1 kg ha^{-1} P locally.

Despite these differences, the model framework used here was demonstrated to predict measured concentrations for a validation lakes dataset (Section 4). The phosphorus losses predicted by the PSYCHIC model are also based on recent evidence regarding the importance of hydrological pathways, especially of sub-surface drain flow, and have been supported by field evidence. It is also suggested that the model is better able to represent the impact of spatially varying environmental factors than the export coefficient model results quoted for comparison. When integrated with inventory estimates of phosphorus losses from other sectors, the source apportionment is also comparable to recent independent calculations. White and Hammond (2007) calculated the agricultural contribution to all phosphorus losses across England and Wales to be 30% in comparison to this project estimate of 18% (Section 5).

Overall, it was calculated that 32% of all lakes in England and Wales achieved standards for good ecological status in 2004. This was forecast to increase slightly to 33% in 2015. It was calculated that 31% of the 130 lakes with a surface area greater than 50 ha achieved standards for good ecological status in 2004 and 34% in 2015. The failing WFD reporting lake systems required an average 46% reduction in total phosphorus loads from all sectors to achieve good ecological status, compared to an average reduction of 55% for lake systems of all sizes (Section 5). On this basis, the WFD reporting lakes appear to be representative of the total lake population. However, we must also consider the uncertainty associated with the MEI model used to predict reference values for total phosphorus concentrations. The percentage of the variance explained by the model is modest (50%) and results in a four-fold variation in the reference concentration for an individual lake. This factor may be more significant for the smaller number of WFD reporting lakes.

Mitigation methods for the control of phosphorus losses from agricultural land were reviewed by Cuttle *et al*. (2007). No single mitigation method can achieve the load reductions required to achieve good ecological status in the failing lake catchments, with the exception of exit strategies such as arable land reversion and large reductions in livestock numbers. Preliminary calculations using the model framework developed by Anthony (2006) indicate that the maximum additional national phosphorus reductions that can be achieved beyond 2015 are *c*. 15% from arable land and 30% from improved grassland. This would be at a significant annual cost to the agricultural sector. However, this analysis assumes 100% uptake of the mitigation options available to landowners and it is reasonable to believe the implementation will fall short of the theoretical potential due to a range of financial and social constraints and individual value judgements. Furthermore, there are few mitigation methods relevant to areas of rough grazing that are important in many lake catchments. The potential reductions therefore fall considerably short of the large reductions required for all lakes to achieve good ecological status (Section 5).

However, it is possible that the reductions that can be achieved will have a greater ecological impact than inferred from the reductions in the total phosphorus loss. Particulate phosphorus losses make up 30 to 50% of total losses from rough grazing and improved grassland (Haygarth *et al*., 1998; Hooda *et al*., 1999; Withers *et al*., 2007). The available mitigation methods are generally focussed on the control of soluble phosphorus losses from fertiliser and manures. These losses are more bio-available than the particulate losses from the soil. Hence, control of these sources may result in a greater ecological response than control of particulate phosphorus losses. The impact of the differing bio-availability of the phosphorus inputs from the different agricultural sources is uncertain. In a similar vein, one particular source of uncertainty in rural catchments is the contribution of bio-available phosphorus from septic tanks and small sewage treatment works. All septic tanks contribute, to some extent, to the pollution of nearby surface and underground water bodies 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. Very little is also known about how effective septic tank systems are at removing phosphorus from sewage effluent, and how much phosphorus they release to surface waters. It is important that this is now quantified so that it can be placed into context in relation to other sources of phosphorus in rural areas, such as agriculture. The availability of the additional information necessary for improving estimates from rural sewerage was recently investigated through a case study of the Loch Leven catchment (Dudley and May, 2007)

Based on the modelling results and available monitoring data that show a high frequency of impacted lakes, it could be argued that the nutrient standards are too stringent, either with too low reference conditions, or with a good/moderate boundary reflecting less than a 'slight' deviation from undisturbed. However, in terms of reference conditions it is difficult to justify different levels than those agreed by UK TAG. These were based on an analysis of a large dataset of European reference lakes (>500 lakes) and have been validated in the UK with independent approaches, such as the use of diatom fossils in lake sediments to reconstruct baseline phosphorus concentrations (Bennion *et al*., 2004). It is worth noting that application of the morphoedaphic model to provide reference phosphorus concentrations for the lake types assumes that lake alkalinity has not changed from the pristine state. A history of liming of agricultural land may invalidate this assumption, although this would lead to even more stringent standards applicable to lower alkalinity lakes. In terms of the good/moderate boundary, the phosphorus standards were based on current understanding of ecological impacts. There are clearly uncertainties in these empirical ecological models, but even taking these into account would not alter these standards to any significant extent.

This study assumes that a reduction in external load will result in an instantaneous reduction in in-lake phosphorus concentration. A wide range of case studies have, however, observed a lag effect following a reduction of external nutrient inputs from a catchment to a lake, demonstrating that lake recovery can take many years, if not decades to occur (Sas, 1989; Jeppesen *et al.* 2005). This is especially true of shallow lakes where the sediment to lake volume ratio is lower than deeper lakes (Sas, 1989). This transient recovery period is caused by gradual re-equilibration of phosphorus stored in the lake bed sediments with the overlying water column following a reduction in external loading (Spears *et al.,* 2006 Søndergaard *et al.,* 2005). Sediment phosphorus concentrations, and therefore the expected length of the transient period, are regulated largely by the pollution history (i.e. intensity and time period) and a lake's flushing rate (Spears *et al.,* 2007). A low flushing rate and extended pollution history will result in high sediment phosphorus content and, therefore, long recovery period, and vice versa. Eventually, a new equilibrium state (post-transient state) will establish at which the resultant ecological status of the lake will be reached, assuming no alteration in external nutrient loading during the transient period.

As Vollenweider type models do not include a sediment release correction, the predicted absolute in-lake total phosphorus concentrations reported here for 2015 represent potential future equilibrium conditions (post-transient state), but may greatly underestimate the timing of the lake response. As such, the results presented are likely to represent the best-case scenario of what is likely to be achieved. The actual time required for many lakes (especially shallow lakes) to respond to reduced nutrient loading may extend beyond the current timelines of the WFD (i.e. 2015 and 2021).

A range of predictive models have been developed to assess recovery time in lakes. These models generally estimate the period taken for re-equilibration of phosphorus between sediment and overlying water column given a known reduction in external phosphorus load to the lake, sediment total phosphorus concentration, and flushing rate (Jensen *et al.,* 2006). Unfortunately, few data exist for UK lakes (or lakes in general) detailing absolute sediment total phosphorus concentrations, with which such predictions may be made. A general sediment typology was presented by Sas (1989) for shallow lakes designed to predict the period of time over which re-equilibration will occur following a reduction in external phosphorus load. This ranged from near instantaneous response (sediment total phosphorus concentrations <1 mg P g^{-1} dry weight) to a transient period of greater than 5 years (sediment total phosphorus concentrations >2.5 mg P g^{-1} dry weight). These predicted transient periods, however, seem conservative when compared to those observed using long-term lake monitoring data at UK sites recovering from external nutrient load reductions (e.g. Loch Leven (Carvalho amd Kirika, 2002; Spears *et al.,* 2006), Lough Neagh (Foy *et al*., 1995) and Llangorse Lake (May *et al*., 2008). This is further supported by a synthesis of 35 case-studies from around the world, which demonstrated that internal loading delayed recovery, and in most lakes a new equilibrium for total phosphorus was only reached after 10 to 15 years (Jeppesen *et al.,* 2005). A more comprehensive survey of sediment total phosphorus concentrations from a population of UK lakes is really needed to more accurately predict lake recovery times from loading reductions.

The discussion above only considers lags in the response of in-lake phosphorus concentrations to external phosphorus loading. Ecological responses, such as phytoplankton chlorophyll concentrations, cyanobacteria blooms or changes in aquatic plant or fish communities are even less predictable. These are all measures to be used in future WFD ecological quality classifications for UK lakes. Of the limited ecological studies available, most have focused on responses of chlorophyll concentrations to nutrient load reductions. These case-studies (Carvalho and Kirika, 2002, Moss *et al*., 2005, Phillips *et al*., 2005; Jeppesen *et al*., 2005) all indicate strong ecological resilience to reduced in-lake phosphorus concentrations, with limited improvements observed in chlorophyll concentrations or algal blooms even following large reductions in both external loads and in-lake phosphorus concentrations. It is certainly apparent that recovery of chlorophyll concentrations does not follow predictions derived from well-recognised phosphorus-chlorophyll relationships (e.g. Vollenweider and Kerekes, 1982), although the uncertainty in these models is often great and increases with increasing phosphorus concentrations.

One of the main reasons for ecological resilience is that even following reductions of in-lake phosphorus concentrations, phosphorus may still be in excess of plant and algal requirements, and other factors, such as nitrogen or light availability may now limit production. Ecological resilience to change in in-lake nutrient concentrations is particularly great under very high phosphorus concentrations. This resilience is weakened at intermediate concentrations where a lake can switch between alternative ecological states (e.g. turbid, phytoplankton-dominated waters to a clear-water, macrophyte dominated state) regardless of nutrient concentration (Scheffer, 2004). Thus, in order to achieve the desired ecological response, it is important to ensure that any external load reduction results in sufficient reduction in in-lake phosphorus concentration below intermediate concentrations. From a WFD perspective, phosphorus standards for UK lakes have been established to reflect these thresholds in ecological change. The moderate/poor status class boundary representing a high likelihood of poor water quality (high chlorophyll concentrations, cyanobacterial blooms, poor submerged macrophyte growth), whilst the good/moderate boundary represents a low likelihood of poor water quality. The moderate status class, therefore, represents the phosphorus gradient over which ecological change is most likely to be observed. Sites in poor status represent a doubling of phosphorus concentrations from moderate status, and sites in bad status a further doubling from poor status. Given the large number of sites in poor or bad status it, is therefore, unsurprising that the reductions in phosphorus loadings modelled in this study have little effect on the proportion of sites failing good status. In many lakes, particularly in lowland England, greater than 50% reductions in phosphorus concentrations may be required before phosphorus becomes the predominant limiting nutrient again. Lake algal biomass and diversity of macrophytes may also be limited by nitrogen rather than phosphorus loads (James *et al*., 2005). A focus on reducing both nitrogen and phosphorus loadings from the catchment is, therefore, likely to be more effective.

Finally, it is important to note that phosphorus eutrophication risk is only one of a number of indices for measuring the ecological status of lake systems under the Water Framework Directive. Kernan *et al*. (2004), for example, estimated that 3% of standing waters in England and 20% in Wales were at risk of acidification due to atmospheric deposition. The areas most at risk were the Lake District, upland Wales and the Pennines. The assessment was based on the same GB Lakes inventory (Hughes *et al*., 2004).

6.1 Summary and Conclusions

This project has evaluated the total phosphorus loads input to lake systems by combining computer models and inventories of diffuse and point sources of pollution. Phosphorus loads from agricultural land were calculated using the PSYCHIC model (Davison *et al*., 2008), and incorporated forecasts of crop areas, livestock numbers and agricultural practices for 2015 under a Business as Usual scenario (Shepherd *et al*., 2007). Diffuse losses from roads and urban areas were calculated using the Expected Mean Concentration methodology of Mitchell (2005). Losses from sewage effluent works were calculated using an Environment Agency inventory of consented discharges, and from septic tanks were calculated based on property counts and water service company data on the numbers of connected properties. Lesser contributions from atmospheric deposition, regional groundwater flows and riverbank erosion were calculated based on a combination of monitoring and model data. The calculated loads were then input to the OECD retention model (Vollenweider and Kerekes, 1982) to calculate annual average total phosphorus concentrations that were compared against standards for good ecological status defined under the Water Framework Directive (Cardoso *et al*., 2007; UK TAG, 2007). Load calculations and estimates of ecological status were made for 5,536 individual lakes with a surface area greater than 1 ha defined in the GB Lakes inventory (Hughes *et al*., 2003). Summary results relevant to the reporting requirements under the Water Framework

Directive (WFD) were also provided for 130 lakes with a surface area greater than 50 ha.

The total calculated phosphorus loss from all sources to all fresh waters in England and Wales in 2004 was 27.3 kt P of which 18% was due to agriculture and 73% was due to sewage effluent discharges. The source apportionment was consistent with recent independent estimates of the total phosphorus budget for the United Kingdom (White and Hammond, 2006). The national total phosphorus loss from the agricultural sector was forecast to decrease from 5.0 to 4.6 kt P by 2015 in response to changes in livestock numbers and implementation of mitigation practices under Business as Usual (Shepherd *et al.*, 2007). The reductions in phosphorus loss varied spatially and were greatest in areas of relatively high livestock density. Surface runoff and drainage from agricultural land was the dominant source of phosphorus in 54% of all modelled lake catchments, and in 70% of the WFD reporting catchments with a lake surface area greater than 50 ha. Overall, agriculture was responsible for an average 43% of the total phosphorus load input to lake catchments. Urban runoff was the next most dominant source in all lake catchments of all sizes but was dominant in only 2% of the WFD reporting catchments. Sewage effluent discharges were the dominant source in only 6% of all catchments and 16% of the generally larger WFD reporting catchments. Atmospheric deposition was dominant in 10% of the WFD reporting lake catchments.

It was calculated that 32% of all lakes in England and Wales achieved standards for good ecological status in 2004. This was forecast to increase slightly to 33% in 2015. It was calculated that 31% of the 130 lakes with a surface area greater than 50 ha achieved standards for good ecological status in 2004 and 34% in 2015. The failing WFD reporting lake systems required an average 46% reduction in total phosphorus loads from all sectors to achieve good ecological status, compared to an average reduction of 55% for lake systems of all sizes. It was calculated that a 100% reduction in only the agricultural sector contribution would achieve a maximum of 65% of all lakes achieving good ecological status. A more modest and feasible 25% reduction would result in 38% of all lakes achieving good ecological status. A 100% reduction in only the agricultural contribution would achieve a maximum 80% of the WFD reporting lakes achieving good ecological status, and a 25% reduction would result in 41% achievement.

The percentage of all lakes that were calculated to achieve good ecological status was generally consistent with previous model based calculations. Carvalho *et al.* (2003) calculated that 88% of lakes in England and 56% of lakes in Wales would fail to achieve good ecological status under present day conditions. Our model estimates for the WFD reporting lakes in 2015 were 68% and 49% respectively. The Environment Agency has also reported that 82% of 109 monitored lakes in England and Wales have failed standards for good ecological status (UK TAG, 2008). Uncertainties in the calculation of the sector pollutant loads are not thought to be significant in estimating population statistics for good ecological status. These calculations therefore forecast that a high percentage (66%) of both WFD reporting lakes and the wider population will fail phosphorus standards for good ecological status in 2015. Additional large (25%) reductions in the phosphorus load from agricultural sources would result in only modest improvement in the population statistics. Even if these reductions occurred then the internal phosphorus loading of the lakes may delay response and compliance by 10 to 15 years (Jeppesen *et al.*, 2005) and possibly longer. Robust estimation of the response time will require a characterisation of phosphorus concentrations in lake sediments and a more sophisticated process based modelling methodology (see, for example, Dahl and Pers, 2004).

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