Review of phosphorus pollution in Anglian River Basin District

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Miranda Kavanagh

Director of Evidence

Executive summary

Introduction

The Environment Agency and National Farmers Union have been working together to find ways to improve the quality of water in catchments, through a new initiative to address phosphorus (P) pollution from farming. Phosphorus enriched waters can lead to eutrophication, which may adversely affect the ecology and quality of water bodies as a result of excess plant and algal growth. Phosphorus pollution is one of the reasons water bodies commonly fail to achieve “good status” under the Water Framework Directive (WFD). Measures to reduce the level of nutrients in waters need to be targeted where river ecology has deteriorated due to elevated nutrients, taking into account costs and benefits.

Before embarking on a catchment-scale pilot project involving farmers changing their practices, the NFU and the Environment Agency needed to agree on the precise role agriculture plays in eutrophication. This report reviewed the evidence on sources of P, transport pathways to rivers and streams, the resulting ecological impacts and the likely cost-effectiveness of mitigation measures.

The evidence collected is being fed into two pilot catchment studies which aim to:

- identify the nature of phosphorus pollution from farming;
- engage with farmers in the catchments and encourage them to participate in the project to alleviate this source of pollution;
- monitor the success of the initiative in terms of water quality improvements (the pilots will need to run for four to five years to evaluate their success).

Sources of phosphorus

The release of phosphorus to rivers and lakes is being driven by the rising human population which demands more food, greater agricultural production, and for human waste to be disposed of. Agricultural sources are livestock waste and the use of artificial fertilisers, which release P that travels along impervious (farmyards, tracks) and pervious (farmed land) surfaces. Farmyards on livestock farms have high levels of both soluble and particulate P. Rainwater interacts with only a shallow layer of soil so P loads in surface runoff depend on the readily-available (soluble load) and total P (particulate load) in this layer. Fresh applications of fertilizer, and manure and crop residues also contribute to this runoff. Sub-surface P losses are increased by preferential flow through surface cracks. Soil removal by rainsplash, runoff and erosion is the main cause of P export from arable land. As little as 0.1 kg P ha^{-1} yr^{-1} can enrich P concentrations in drainage water sufficiently to cause eutrophication in Eastern England.

Natural background sources of P include atmospheric deposition, soil weathering, river bank erosion, riparian vegetation and migratory fish. Phosphorus concentrations from these sources are generally low, but may be significant where other inputs are limited. Anthropogenic inputs come from point and diffuse sources. Point sources of P include wastewater, septic tank and industrial effluents, and generally contain a high proportion of soluble, bioavailable P. Diffuse sources include urban and road runoff, and contain a higher proportion of particulate P. Sewage treatment works (STW) contribute up to 90 per cent of the non-agricultural P load.

England & Wales

National estimates suggest industrial, human and household sources (point and diffuse) account for around 65-76 per cent of total P in rivers, mostly from STW, while agriculture accounts for 18-28 per cent, and background sources make up about 4.5-6.5 per cent (Environment Agency 2010a). Agricultural sources are thought to be particularly significant for lakes, protected habitats and headwaters. Other diffuse sources, such as septic tanks and urban runoff, may contribute up to 40 per cent locally, although at the national scale, these contributions are much lower (Dudley and May, 2007). The timing and form of diffuse versus point source loads are significantly different. Diffuse P loads vary with rainfall, while point source inputs are consistent throughout the year. Most agricultural P is in particulate form, which may be less damaging to riverine ecology as it is not immediately bioavailable. Soluble P losses from agricultural sources can also be high, especially when rainfall follows fertiliser or manure applications.

Eutrophication remains a serious and widespread problem in England and Wales. In 2008, monitoring indicated that half of English rivers and nearly a tenth of Welsh rivers (by length) had 'high' (>0.10 mg P l^{-1}) soluble P concentrations. Under the UWWTD and Nitrates Directive, five per cent of rivers, two per cent of lakes and 15 transitional and coastal waters are designated as "eutrophic". Many water bodies are currently being investigated under the WFD to establish whether they have a eutrophication problem.

Measures are already being implemented to improve water quality under the WFD and other legislation, although they are not expected to have enough impact to meet WFD objectives in 2015 (Environment Agency, 2010a).

Anglian River Basin District (RBD)

The Anglian River Basin District (RBD) is a predominantly rural area, with more than 70 per cent of the land area used for agriculture. The impact of phosphorus from agricultural land on river concentrations is highly dependent on the flow of water across the land and hence its dilution potential; flows are often low in the Anglian RBD due to low rainfall, meaning small amounts of P loss can lead to high P concentrations. Groundwater shows some of the highest P concentrations in the UK, above 0.12 mg l^{-1} in some areas. Soil P levels are generally greater in East Anglia than in the rest of the country due to high fertilizer and manure use in intensive arable farming.

Recent WFD assessments of the Anglian RBD indicate that nearly a fifth of waterbodies attain 'good' ecological status, while a third have 'high' or 'good' biological status. In terms of P compliance, under a third have 'high' or 'good' status. The River Basin Management Plan objectives are to improve the classification of P in 772 km of rivers and seven lakes by 2015. The estimated cost of reducing phosphate pollution is higher in the Anglian RBD than in any other RBD, at £158 per kg.

Pilot catchments

Phase 2 of this project is to identify two pilot catchments where the NFU and the Environment Agency can work in partnership to deliver improvements in riverine phosphate concentrations through working with the farming community to modify current management practices. The catchments were selected because they (a) failed WFD targets for P, (b) were representative of rivers in the RBD, (c) did not have any initiatives aimed at reducing diffuse pollution, and (d) were deemed to have relatively high agricultural contributions . The catchments selected were Harpers Brook (Northamptonshire) and upper Bourn Brook (Cambridgeshire). Walkovers of the catchments were conducted in 2010/11 to confirm their suitability.

Harpers Brook – For this water body, WFD standards are 0.05 mg soluble reactive phosphorus (SRP) l^{-1} (high) and 0.12 mg SRP l^{-1} (good); the mean concentration was 0.18 mg SRP l^{-1} (2000-2010) measured at the monitoring point at the catchment outlet. Modelled P concentrations show that point source contributions to the catchment are high compared with diffuse sources, with two STW contributing around half of the load at the nearest monitoring point. The remaining half comes from ‘diffuse’ sources. Agricultural losses of P come from three main sources: areas of hard-standing (largely beef cattle), and manure and fertilisers. Losses from fertiliser are mainly in the upper part of the catchment, whilst P loss from manures and areas of hard standing is spread more evenly across the catchment. The walkover revealed that sub-surface drainflow is the dominant pathway of P loss from farmed land and land drains run turbid during heavy rain. Concentrations of P in groundwater are potentially significant.

Bourn Brook - For this water body, WFD standards are 0.05 mg SRP l^{-1} (high) and 0.12 mg SRP l^{-1} (good); the mean concentration was 0.62 mg SRP l^{-1} (1999-2010) measured at the monitoring point at the catchment outlet. Modelled P concentrations are similar to those seen in Harpers Brook, although diffuse loads are thought to be slightly lower. P losses from the three agricultural sources (fertilisers, manures and areas of hard standing) are similar, and drainflow is the main pathway for P transport. There is thought to be little connection between surface and groundwater systems, and groundwater P concentrations are lower than in Harpers Brook.

Lessons learnt from this project regarding on-the-ground catchment delivery included:

- Difficulty in identifying pilot catchments that :
  - failed WFD status objectives on phosphorus only;
  - were representative of rivers in the Anglian RBD;
  - currently did not have any on-going agricultural initiatives aimed at reducing diffuse phosphorus pollution;
  - were deemed to have relatively high agricultural contributions
- Importance of catchment walking to supplement and ground-truth catchment characterisation information derived from monitoring and modelling.
- Need for long-term monitoring programmes due to lag times.
- Importance of establishing a baseline to enable the effectiveness of catchment interventions to be assessed.
- Benefits of early engagement with farmers through established and trusted advice networks.

Recommendations for further research

This report recommends further research on the following subjects:

- The relative importance of runoff from hard surfaces and septic tanks as sources of P to surface waters and groundwater.
- The contribution of sewer misconnections to urban diffuse sources of P, and the relative contribution of urban diffuse sources to surface waters.
- The adequacy of current routine monitoring of P for determining water body P fluxes and cycling.
- The feasibility of using seasonal (summer/winter) P standards.
- The role of river headwaters in buffering P loadings.
- Understanding the recovery of rivers and lakes from decreases in P loadings.
- The effectiveness of measures to reduce diffuse sources of P at the catchment scale, to supplement evidence available at the field and farm scale.
- The sustainability of emerging treatment processes to remove P from wastewater, and the role of recovery and reuse of P from wastewater.

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1 Introduction

1.1 Background

The Environment Agency and the National Farmers Union (NFU) are working together to find ways to improve the quality of water in catchments, through a new initiative to address phosphorus (P) pollution from farming. Two pilot catchments in Anglian Region have been chosen to examine the contribution of agriculture to P pollution. In the pilot catchment areas, the NFU is engaging with its membership to address the problem through voluntary action.

The consequence of phosphorus enrichment of freshwater is the risk of eutrophication which adversely affects the ecology, quality and uses of water bodies through excess growth of plants and algae. Control measures (in this case, voluntary action) need to be targeted at waters with elevated nutrient levels, taking into account costs and benefits.

Phosphorus pollution is one of the most common reasons for failing to achieve “good status” for water bodies under the Water Framework Directive (WFD). Non-compliance with phosphorus standards in England is considerable – over half of the river length in England and Wales falls below ‘good’ status. In the Anglian River Basin District (RBD), P is the cause of failure to achieve ‘good’ status in 381 out of a total of 757 river water bodies (50%). In a fifth of these water bodies, P is the sole reason for failure. The Environment Agency is carrying out a number of investigations to identify the reasons for these failures. These investigations will use a weight-of-evidence approach, using information about P chemical levels and about the biological impact, to assess the need for measures and to support voluntary actions by farmers and land managers.

Phosphorus loads need to be correctly apportioned to different sources for control measures to be appropriately targeted. White and Hammond (2009) suggest that in England, households account for roughly 78%, agriculture for 17%, industry for 1% and background sources for 3% of the total P load. Looking at the UK as a whole, the relative contributions to total P are: households 72%, agriculture 20%, industry 3% and background 5%. For the Anglian River Basin District (RBD), of the estimated total P load 14% is from agriculture and about 80% from households. The agricultural contribution is around one-third lower if the calculations are based on soluble reactive phosphorus (SRP) (White and Hammond, 2009). The relative contribution of P from different sources changes with time in response to land management, climatic variation, technological advances, national and international regulations and voluntary practices. The relative contribution also changes significantly within and between catchments and larger RBDs.

1.2 Rationale for project

A variety of policy tools, from incentives to advice, are applied nationally and regionally to combat P loss from agriculture. The England Catchment Sensitive Farming Delivery Initiative (ECSFDI) is attempting to address diffuse pollution from agriculture, including P pollution, in priority areas.

The rationale for this project was to establish an agreed evidence base on the sources, effects and measures to combat P pollution primarily within the pilot catchments but set in the wider Anglian RBD and National context.

This project carried out an independent review to collect evidence to support discussions with the agricultural industry on a strategic approach to tackling aquatic eutrophication. The pilot studies will use the evidence presented in this report to shape farmer-led measures coupled with a monitoring programme to determine the effectiveness of the measures over time.

1.3 Aims and objectives

Before introducing voluntary agricultural mitigation measures, the Environment Agency and the NFU need sufficient confidence in the evidence on a range of issues including: current chemical and ecological status of water bodies; nature and sources of the pressure causing problems in these water bodies; and the cost-effectiveness and proportionality of the measures proposed to tackle the problems. The overall objective of the project was to independently review evidence for freshwater (rivers and lakes) on phosphorus sources, pathways and ecological impacts, and the likely cost-effectiveness and proportionality of agricultural mitigation measures. This was done using a tiered approach to gather evidence at three spatial scales: national; Anglian RBD; and two pilot catchments within Anglian RBD.

The findings provide an evidence base for the Environment Agency, and for the NFU to promote voluntary measures in the East of England to improve the ecological condition of inland and coastal waters in terms of levels of P. The report also identifies gaps in the evidence base and where further work is required.

The study looked for evidence of how likely current and proposed measures to control P will result in: (i) reduced P loadings to watercourses, (ii) reduced P concentrations in receiving waters, and (iii) significant changes in aquatic ecology.

1.4 Scope and approach

The project employed a two-stage approach:

1. A review of scientific literature to describe cause-effect relationships between farming activities and the environment. A Driver-Pressure-State-Impact-Response (DPSIR) approach was used to assess pressures and impacts on watercourses, exploring the sources and pathways of agricultural and non-agricultural pollution and the receiving environment.

2. Based on scientific knowledge and case studies developed over the last 15 years, the report aimed to fully or partially answer the following questions at (i) national, (ii) Anglian RBD, and (iii) pilot catchment scales:

- How much P from different agricultural activities gets into surface waters? How does this compare to non-agricultural sources?
- How, when and in what form is P mobilised and transferred from farmland to surface waters, and then within surface waters? What is the potential for re-release and what are the mechanisms and likely timing?
- What is the contribution of agriculture to P concentrations in groundwater and hence to groundwater-dependent ecosystems and in-stream loads/concentrations?
- Where does P from agricultural sources contribute to excess, ecologically significant, levels of P, and what is the effective contribution? What impact on surface water body status (P and ecological) could reductions in the agricultural load have?

- For catchments within Anglian RBD with P removal at STW, do agricultural P sources contribute to excess, ecologically significant levels of P? What is their current P and ecological status, and that forecast for 2015?
- What voluntary measures can be used by those who manage land and water to reduce the level of phosphate reaching and mobilising within surface water bodies?
- What will be achieved in terms of annual and seasonal P load reductions, P concentrations in water bodies and ecological status, if these voluntary measures are adopted? How quickly may benefits be realised?
- Where and to which farming sectors should these voluntary measures be targeted to be most cost-effective?

1.5 Structure of report

The report is structured as follows:

- Section 2 uses a DPSIR approach to explore and summarise our understanding of the causes of P emissions to water, the mechanism by which eutrophication occurs and interventions used to address P enrichment.
- Sections 3, 4 and 5 use the evidence base from Section 2 to address questions at national (Section 3), Anglian RBD (Section 4) and pilot catchment (Section 5) scales.
- Section 6 summarises the findings of this report in a discussion and sets out the conclusions.

2 DPSIR assessment

2.1 What is DPSIR?

Driver-Pressure-State-Impact-Response (DPSIR) is a conceptual framework for organising information about the state of the environment. The framework assumes cause-effect relationships between interacting components of social, economic, and environmental systems, which are:

- **driving forces** of environmental change (such as economic and social policies of governments, large-scale environmental processes such as climate change);
- **pressures** on the environment (such as phosphate emissions to water);
- **state** of the environment (such as water quality in rivers and lakes);
- **impacts** on population, economy, ecosystems (such as eutrophication);
- **response** of the society (for example, catchment management, nutrient stripping).

This section uses the DPSIR approach to explore and summarise scientific understanding of the causes of P emissions to water, the mechanisms by which eutrophication occurs and the management interventions used to address excessive P enrichment. This knowledge base is then used to answer the questions posed in Sections 3, 4 and 5.

2.2 Drivers and pressures

2.2.1 Agricultural sources of phosphorus

2.2.1.1 Drivers

In England and Wales, there are currently no direct regulatory controls on agricultural P inputs (to limit the accumulation of surplus P in soils), or the prevention of P loss from agricultural activities.

Export of phosphorus (P) from agriculture to water occurs as a result of the interactions between the way water moves through the landscape (climate, topography, soil type), the annual inputs (fertilizers, feed and imported wastes) and stores (soil, slurry stores) of P at the land surface and the way the land is managed (land use, soil, crop, livestock and nutrient management). By their very nature such interactions show high spatial and temporal heterogeneity and the intensification, specialization and regionalization of agriculture since the Second World War has resulted in farming systems that have greatly modified these interactions. Changes in farming include: (a) a move away from mixed farming to specialized arable and livestock systems that are geographically separated; (b) a reliance on substantial inputs of water-soluble P fertilizers and feeds as farming systems have intensified; (c) a switch to continuous cultivation on vulnerable soils (including marginal land) and introduction of tramlines; (d) the removal of hedgerows which increases the length of slope for runoff generation; (e) increased stocking densities and adoption of slurry-based manure recycling; and (f) the expansion of field underdrainage systems to remove soil waterlogging, due to increased government grants since 1940 (Green,1980; ADAS, 2002a). These practices have left a legacy of phosphorus surpluses and a widespread increase in soil P fertility, organic matter-depleted soils that are more vulnerable to soil erosion, increased runoff due to cultivation methods and soil compaction, greater opportunity for P release from livestock manure to runoff, and greater hydrological connectivity between the field and the watercourse. These changes are considered to have increased runoff, erosion and P transfer from agricultural land in the UK in recent decades (Withers and Lord, 2002; Johnston and Dawson, 2005) increasing the risk of eutrophication in inland and coastal waters. The amounts exported are small in agricultural terms (0.3-6 kg ha^{-1} yr^{-1}) but are sufficient to trigger eutrophication (Ulén *et al.* 2007). Johnston and Dawson (2005) calculated that as little as 0.1 kg P ha^{-1} yr^{-1} would enrich P concentrations in drainage water sufficiently in Eastern England to cause eutrophication.

Two main drivers determine current and future P export from agriculture. The first relates to inputs of P in fertilizers and manures to the land surface and to what extent these are in surplus relative to crop P uptake. The greater the P surplus the greater the *potential* for P loss in runoff because of the link between surplus P accumulation in soil and P concentrations in runoff (Maguire *et al.* 2005), and the risk of P release to runoff from fertilizers, livestock excreta and manures deposited or recycled around the farm (Withers *et al.* 2003; Hart *et al.* 2004). The second relates to land use patterns and farming practices that increase the vulnerability of land to runoff, erosion and enrichment of runoff with recently-applied or deposited P. Examples of land management pressures on P loss risk are the proportion of land under cultivation in arable catchments and stocking densities in grassland catchments (Foy and Withers, 1995). The natural storage of P in the landscape and the way sources are managed affect the amount, form and timing of P export from agriculture and the risk of eutrophication. In principle, a small surplus managed badly may lead to as much, or greater, P export than a large surplus managed well.

Phosphorus surplus and soil P fertility

Trends in the national P surplus in UK agriculture since 1935 are shown in Figure 2.1. The largest P surpluses arose during the 1970s and 80s when livestock numbers and imports of highly P-soluble feeds and fertilisers were at their peak. In recent years, the P surplus has dropped to around 5 kg P ha^{-1} largely due to a steep reduction (more than 50 per cent) in P fertiliser use on arable and managed grassland in recent years (AIC, 2009). Current consumption of P fertilizer is around 96,000 tonnes as P. Manure P inputs are a reflection of animal numbers since about 70 per cent of dietary P intake is excreted. Although animal numbers have declined by five per cent for poultry, 11% for cattle, 27% for sheep and 35% for pigs over the last 10 years, consumption of imported feedstuffs (concentrate and non-concentrate) has increased by five per cent (Defra, 2010a). To what extent the P content of imported feeds has changed over this time period is unclear. Manure P inputs (excreted during grazing and spread) are now estimated at around 130,000 tonnes as P, a reduction of 40 per cent since 1993 largely due to the drop in animal numbers (Withers *et al.* 2001). The reductions in fertilizer and manure P inputs reflect the general (but fluctuating) unprofitability of farming, the large increase in the cost of fertilizers and imported feeds and the impacts of foot and mouth and bluetongue on animal numbers. The national P surplus is now lower than that calculated for 1935 and further downward trends can be expected if the price of fertilizers and feeds continues to increase.

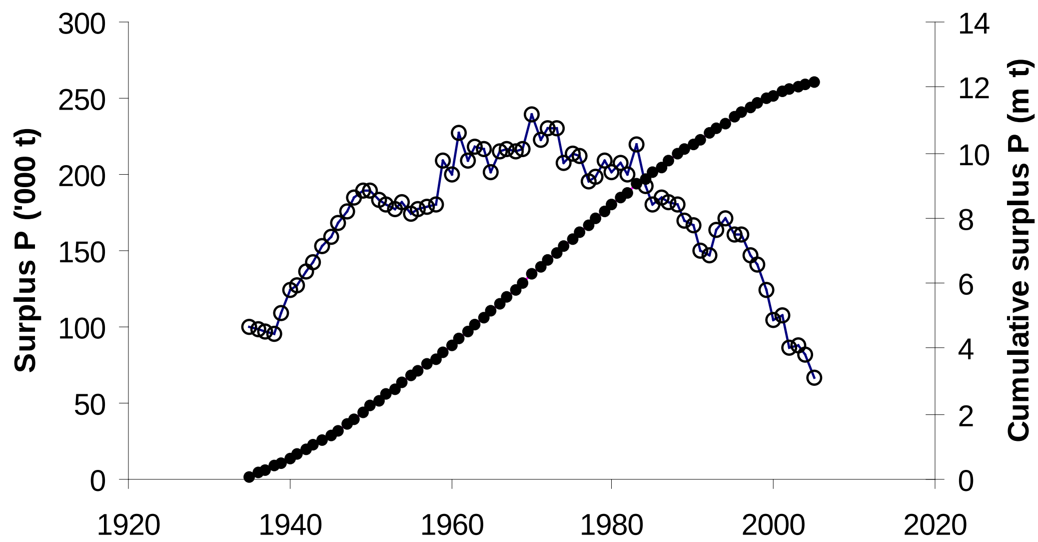


Figure 2.1 Changes in the annual surplus of phosphorus for the UK between 1935 and 2008 and the cumulative trend (in million tonnes) (adapted from Withers *et al.* 2001)

The cumulative P surplus since 1935 amounts to 12 million tonnes and is still increasing whilst a P surplus exists. This equates to an average P loading to UK soils of over 1,000 kg P ha^{-1} over the last 70 years, which has doubled the mean 'background' concentration of total P in soils (Withers *et al.* 2001). Similar trends in P surpluses and soil P status are found in a number of countries such as the Netherlands (Reijneveld *et al.* 2010) and Ireland (Foy *et al.* 2002).

The impact of cumulative P surpluses on soil total P (TP) and soil test P (STP: the soil test (usually Olsen) routinely used to measure readily-available P for calculating fertilizer recommendations, Defra, 2010b) is site-specific depending on soil type, the farming system and the depth to which P has accumulated. The spatial variability in surplus P is therefore large at the field and farm scale; for example, Domburg *et al.* (2000) calculated surpluses of up to 160 kg P ha^{-1} for farms in one Scottish catchment. This has resulted in a wide range of soil total and STP concentrations in UK soils (Withers *et al.* 2001; Emmett *et al.* 2010). As arable and horticultural soils generally receive larger amounts of water-soluble P fertiliser than grassland soils, they maintain greater average STP concentrations. Soils regularly receiving both manure and fertiliser often exhibit high STP concentrations, for example those in intensive horticulture, in rotations involving responsive crops such as potatoes and sugar beet, or receiving pig and poultry manure and on dairy farms with high stocking rates and forage maize (Skinner and Todd, 1998). Grassland and minimally cultivated soils typically show a steeper STP gradient with soil depth than ploughed arable soils due to the lack of regular soil inversion and cultivation. Current recommendations as stated in the Fertiliser Manual (RB209, Defra, 2010b) suggest that soils should not be maintained above the target Index for plant-available P in soil, that is, P Index 2 for arable crops and grassland, and P Index 3 for vegetables and potatoes. Whilst this recommendation is based on sound agronomic principles, it does not necessarily mean that exceedance of Index 3 will lead to sufficiently high P concentrations in land runoff to cause eutrophication (Withers, 2012).

Land use and farming practices

The areas of cultivated land (4.5 M ha), temporary grassland (1.3 M ha) and managed permanent grassland (6.1 M ha) in the UK have remained relatively stable over the last 10 years (Defra, 2010a). Set-aside land increased temporarily at the turn of the century in response to the Common Agricultural Policy (CAP) in Europe but has since declined as this part of CAP has been phased out. Statistics indicate that set-aside land declined by two-thirds to 159,000 hectares between 2007 and 2008 (Langton, 2009). Areas of individual crops have fluctuated more in response to demand and economic conditions with notable increases in forage maize, potatoes in some areas and in outdoor pig production (Defra, 2010a). Considerable variation in land use and management exists between and within farms. This variation relates to the suitability of the farming system to the prevailing climate and the practices adopted within different farm types. Since P use is largely a function of farm type, the regional distribution of farming systems is an important factor influencing soil P accumulation rates and potential for P loss in relation to climate and landform. For example, in the UK, there is a clear geographical separation between upland farms which occur above 300 m OD, lowland arable farms in the east and lowland grass farms in the west. Arable soils in Eastern England have high soil P fertility (Baxter *et al.* 2006). Grassland occupies about 70 per cent of the UK land area and predominates on heavier-textured soils, whilst arable crops, particularly rotations that include high P input crops (such as potatoes and horticultural crops) tend to be concentrated on well-drained and more workable soils.

Farm practices that influence P transfer in runoff are shown in Table 2.1. These relate to soil, crop, livestock and nutrient management on the farm. Other land management factors are largely historic or reflect previous decisions on farm infrastructure and are difficult to alter; for example, the type of manure handling system and underdrainage systems which increase the connectivity between the field and the watercourse. About 6.4 M ha of agricultural land in the UK is estimated to have been underdrained, of which 2.4 M ha was under grant aid between 1950 and 1993 and therefore considered to be ‘modern’ and reasonably effective (ADAS, 2002a).

Future changes in land use and farming systems are likely to affect the risk of P loss through their impact on P inputs, soil P fertility and the way land is managed. Land use change will be governed by fluctuations in the demand for different commodities (including biofuels) as influenced by global markets and CAP reform, the need to adapt to climate change and comply with international policies to protect resources and ecosystems and the introduction of new technologies that will improve farming systems and food security for an increasing population (Foresight Land Use Futures Project. 2010). Future shortages of rock phosphate are forecast to threaten the supply and increase the costs of P fertilizers (Cordell *et al.* 2009). Rounsevell *et al.* (2005) suggest that traditional farmed land may reduce considerably if technological advances in food production continue at their present rate, with any surplus being used for bioenergy crops. Defra Farm Business Indicator B111 shows that for arable and dairy farming, the trend has been expansion and intensification to increase yields over the last few decades. Crops show a more volatile pattern, their yields being more influenced by the weather.

Table 2.1 Farming practices that influence P transfer in runoff from agricultural land (from Withers, 2008)

Farming operation	Farm type	Timing	Environmental impact	Contributing factors
Land use decisions (including pasture improvement)	All	Variable	Loss of soil and nutrients in rapid runoff and during floods	Siting of high-risk crops close to watercourses
Soil cultivation up and down slope	Arable	Aug-Nov Feb-May	Erosion of soil before crop established	Over-cultivation Late sowing Planting of ridged crops
Establishment of tramlines after sowing	Arable	Sep-Mar	Channelling of runoff and erosion of soil	Trafficking on wet soils leaves deeper channels
Harvesting of root and forage crops	Arable, dairy	Oct-Dec	Erosion of bare compacted soils	Harvesting in wet soil conditions
Fertiliser and manure application	All	Variable	Build-up of P in soil P release from rainfall after application	Over-application No allowance for manure P Broadcast rather than incorporated
Grazing of pastures	Dairy, beef and sheep	Mar-Oct	P release from dung pats Erosion of soil from poached areas	Livestock access to streams High stocking densities Grazing wet soils
Grazing of fodder crops/stubbles	Outdoor pigs Dairy and sheep	All year Nov-Feb	Erosion of bare and compacted soils Build-up of soil P where animals congregate	Grazing during wet soil conditions High stocking densities
Storage of manure on-farm	All livestock	Nov-Mar	Runoff from manure heaps Overflow from slurry stores	Siting manure heaps close to stream Inadequate storage facilities
Moving livestock between fields and between fields and farm	All livestock	All year	Runoff from dung on tracks and roads	Proximity of tracks and roads to stream
Washing out milking pipes and hosing down yards	Dairy, pigs and poultry	All year	Runoff from farmyard	No separate storage facilities

Climate change in the UK is likely to raise average temperatures, winter rainfall, and the intensity of storms and reduce water availability in summer (Hulme *et al.* 2002). Higher temperatures will increase soluble P losses in runoff due to greater mineralization rates of organic manure P in soils and the temptation to keep livestock out over winter due to increased availability of forage. Intense storms will increase surface runoff volumes, erosion rates on fields without crop cover and transport of fertilizer granules and manure particles left on the land surface. Increased flood risk may lead to abandonment of farming in some areas (Orr *et al.* 2008). Farming systems may intensify in northern and western areas where water availability is greater, whilst southern and eastern areas may need to adapt cropping systems to the risk of drought. Hydrological connectivity with the watercourse will also increase during intense storms, increasing the area on the farm from which P maybe mobilized (Heathwaite, 2010). However, these processes will change gradually, over a long period.

2.2.1.2 Sources

Much research has been conducted over the last 15 years to improve our understanding of the sources and pathways of P export from agricultural land, the principal processes by which P is mobilized and delivered to the watercourse and the main factors affecting the amounts and concentrations transported. Phosphorus sources derived from agriculture are widely scattered across catchments; that is, they are diffuse. They have recently been re-defined in terms of their mode of transport during storm events to better establish their ecological relevance (Edwards and Withers, 2008; Withers and Jarvie, 2008). Unlike point sources, agricultural sources of P are largely storm-dependent and occur only when it rains. The exceptions to this are the use of irrigation water on high-value crops which may preferentially find its way back to the watercourse and the hosing down of concrete areas on the farm that might discharge directly to the watercourse via a drain (Edwards and Hooda, 2007). As water flows across and through the landscape, it mobilises P from any inorganic and organic material on the land surface, producing large spatial and temporal variability in the concentrations and loads being transported.

Sources of P on farms can be separated into those originating from impervious surfaces (farmyards, tracks and roads) and those originating from pervious surfaces (farmed land). Impervious surfaces by definition have no, or limited, infiltration capacity and usually rapidly transfer P to the watercourse directly. Pervious surfaces allow infiltration of water and therefore show much greater variation in surface and sub-surface runoff both spatially and temporally depending on storm intensity and soil infiltration rates. It is common for runoff water to contain a mixture from different sources (Withers *et al.* 2009a). Particulate (>0.45 µm) and soluble (<0.45 µµm) forms of P are mobilized through detachment, dissolution and desorption. Dissolution and desorption are sometimes collectively referred to as solubilisation (Haygarth and Jarvis, 1999).

Impervious surfaces

Sources of agricultural P contributing to runoff from impervious surfaces include yard scrapings, livestock faeces, crop residues, roof runoff containing bird droppings, sediment from neighbouring fields and roadside verge soil. Runoff and piped discharges from farmyards where animals congregate have been shown to be highly concentrated in soluble and particulate P (Dils and Heathwaite, 1996; Edwards *et al.* 2007). Some recent data are given in Table 2.2. Farmyards appear particularly important because source material is readily available and high P concentrations are maintained throughout the storm. Edwards and Hooda (2007) measured up to 51 mg TP l^{-1} in flow from two farmyard drains and estimated that the farmyard area contributed 25-30 per cent of annual downstream P loads. Hively *et al*. (2005) concluded that relatively impervious surfaces on farms such as cow paths and farmyards may cover a small area, but the highly concentrated runoff generated from these may be much more significant to summertime P loadings. For example, mean TP concentrations in runoff from cow paths and farmyards were 0.99 and 13.2 mg l^{-1}, respectively.

Table 2.2 Range, mean and median concentrations (mg l^{-1}) of soluble reactive P (SRP) and total P (TP) in different storm runoff sources in three rural catchments (from Withers *et al*. 2009c)

Runoff type	n	SRP				TP		
		Range	Mean (s.e.)	Median		Range	Mean (s.e.)	Median
Field surface	35	0.01-1.6	0.17 (0.05)	0.09		0.17-6.8	1.29 (0.22)	0.94
Field drain	56	0.01-0.5	0.10 (0.02)	0.05		0.02-6.2	0.76 (0.14)	0.39
Tracks	13	0.02-0.9	0.16 (0.07)	0.08		0.24-7.3	2.69 (0.74)	1.46
Roads	51	0.01-1.3	0.29 (0.05)	0.16		0.11-16	2.31 (0.35)	1.70
Farmyard	26	0.01-5.7	1.04 (0.27)	0.51		0.08-15	2.74 (0.68)	1.47
Septic tank	22	0.01-1.4	0.55 (0.09)	0.48		0.17-6.5	1.62 (0.32)	1.23

Sources of P mobilised in road/track runoff include atmospheric deposition, leaf fall, industrial wastes, urban litter, residential activities (such as car washing), livestock excreta, domestic and industrial fertilisers, soil particles deposited by vehicles and eroded from roadside verges, detergents and lubricants (Bannerman *et al.* 1993). Average event mean concentrations of TP are typically 0.2-0.4 mg l^{-1} with about 40-50 per cent in dissolved P form (see Kayhanian *et al.* 2007). However, concentrations can vary by an order of magnitude due to variation in source materials and the speed and routing of flow paths. Concentrations of TP appear to be slightly greater in runoff from residential areas, whilst proximity to agricultural land has a large influence on both TP and soluble P concentrations in rural areas. Recent data from three rural agricultural catchments showed median TP concentrations in road and track runoff of over 1 mg l^{-1} (Table 2.2).

Pervious surfaces

Ahuja (1986) showed that rainwater interacts with only a shallow layer of soil (2-5 cm) and any P enrichment of surface runoff will therefore depend on the sources of available P in or on this shallow layer. Soil, fresh applications of fertilizer and manure and crop residues therefore contribute P to runoff from pervious surfaces. Sources present at the surface also increase sub-surface P losses due to preferential flow down macropores, cracks and fissures (Chapman *et al.* 2005; Schelde *et al.* 2006). In the absence of preferential pathways, inorganic P migrating in solution downwards through the soil profile would be readily adsorbed by the subsoil. Shepherd and Withers (1999) concluded that exceptionally large amounts of cumulative surplus P were needed to cause subsoil enrichment and leaching of inorganic P from uniformly textured soils. However, leaching of P has clearly occurred in certain areas with very sandy or peaty soils of low P retention capacity, such as in Europe's low-lying deltas (Chardon and Schoumans, 2007). Organic P compounds in solution have been shown to migrate through the soil more quickly than inorganic P due to their size-charge characteristics and weaker adsorption strengths. Chardon *et al.* (2007) recently found evidence of such P migration through the soil below large dung pats in grazed fields.

Entrainment of soil particles (particulate P) dispersed by rainsplash, runoff and erosion is the main source of P mobilized from arable land, although particulate P may also be transported from grassland where grazing is high, especially under wet soil conditions (Hodgkinson and Withers, 2007; Billotta *et al.* 2007). As applied P preferentially accumulates on the finer soil particles, it is the transport of fines (fine silt and clay) in agricultural runoff that is of most concern with respect to eutrophication. Release of soil soluble P to runoff depends on the degree of soil P saturation and soil microbial activity (Maguire *et al.* 2005). Soluble inorganic (reactive - SRP) P is usually the dominant form released from well-fertilized soils, whilst soluble organic forms are also significant in runoff from organic or peaty soils and those regularly receiving organic manures. Release of SRP to runoff is governed by diffusion rates between soil and solution and is therefore soil type-specific. A number of studies have shown that rates of SRP release from soils increase more rapidly once soil P saturation reaches about 25-30 per cent (Nair *et al.* 2004), but this is not routinely measured. Various linear and non-linear relationships between runoff P and STP have been obtained and research on some soils suggest there may a considerable difference between the STP concentration (Olsen P) in soil required for optimum yield and that required to cause a sharp increase in P in runoff water (see Heckrath *et al.* 1995). However, STP is not a suitable indicator of P release to runoff across soil types because it does not take account of the soil P buffering capacity (Kleinman *et al*. 2000). The potential for P export from soil is therefore strongly influenced by the residual P content of the soil (particulate P) and the percentage P saturation of the soil P sorption capacity (soluble P). Soluble P in runoff increases at a greater rate than the particulate P fraction as soil P increases (Withers *et al.* 2009b).

When fertilizers and manures are applied to the surface of the soil, there is a risk of direct P release (soluble P) or entrainment (particulate P) in storm runoff before the P they contain can be adsorbed into the soil; this is termed *incidental* P loss (Withers *et al.* 2003). This source of P generally dominates runoff concentrations and loads where up to 25 per cent of total P applied is transported, ddepending on: the amount of P applied; the properties of the materials applied (percentage of P extractable in water); the timing of storm events after application; soil conditions and the amounts of runoff generated. Large P applications left on the surface of wet, frozen, compacted and intensively underdrained soils in high rainfall areas are particularly vulnerable to incidental P loss. Concentrations of P in runoff are often greatest during the first storm following P application, but can remain high for several weeks or even months after application. In reviewing work for model development, McDowell *et al.* (2009) found that fertilizers/manures could remain a source for up to 60 days, whilst the average was 28 days. The potential for incidental P loss strongly depends on day-to-day management and can, in theory, be more easily avoided.

Decaying vegetation including crop residues can also release P to runoff, although the relative contribution of this source under UK conditions has not been evaluated. In other countries with frequent freeze/thaw cycles, soluble P is released from damaged leaf surfaces (Bechmann *et al.* 2005). Desiccating vegetation in riparian buffer strips may also be a source of soluble P transport to streams (Stutter *et al.* 2009). McDowell *et al.* (2007) found that defoliation by grazing caused substantial release of soluble P from pasture plants, accounting for a fifth of the phosphorus exported from grazed pasture.

2.2.1.3 Pathways

Once mobilised, P is delivered to the watercourse rapidly by overland flow and bypass flow in field drainage systems, or more slowly by sub-surface lateral flow and soil matrix flow via the groundwater. Groundwater contributes a significant proportion of river flow in many areas (NERC, 1998). During transit to, and within, the watercourse and any interconnecting ditches, soluble and particulate forms of P undergo a further series of reactions and transformations; sorption/desorption, retention/remobilization, precipitation/dissolution (summarized by Withers and Jarvie, 2008). Further complexity is introduced as ‘new’ sources continually become mobilized and transported from different areas within the field, farm and catchment and at different times of the year. This complexity often produces a disconnection between the P exported from field plots to that measured at the catchment outlet (McDowell *et al.* 2004; Sharpley *et al.* 2009). Opportunities for retention of P during the delivery process are dictated by the speed of transfer, degree of connectivity and type of landscape (for example, presence of ditches, ponds).

Surface runoff (overland flow) occurs when rainfall intensity exceeds the infiltration capacity of the soil (infiltration excess or Hortonian runoff) or due to a slowly rising water-table or perched water-table (saturated excess runoff). Surface runoff is generated in only small and variable areas (termed partial or variable source areas, VSA) during storm events. When linked to P release, these VSA become critical for P delivery (Pionki *et al.* 2000). This critical source area (CSA) concept has revolutionised catchment management, since P loss mitigation actions needed to be targeted at CSA rather than across the whole catchment.

Infiltration excess runoff can be generated anywhere in the catchment but invariably does not reach the watercourse unless linked by an impervious surface. Modern farming practices have increased infiltration excess runoff by reducing soil infiltration capacity through greater soil compaction caused by tramlines, cultivation and livestock poaching (see Deasey *et al.* 2009; McDowell *et al.* 2003). Soil compaction is a particular risk in grassland areas grazed by livestock; the level of compaction in grassland areas (mostly in western England and Wales) was assessed by a Defra project (BD2304). This study found that around two-thirds of grassland soils in the UK are at moderate or high risk of compaction, based on soil vulnerability and stress (Cranfield University, 2007). Infiltration excess runoff is therefore highly sensitive to land use and land management. Impervious surfaces around farm buildings and along farm tracks and roads have little or no infiltration capacity and therefore contribute the bulk of infiltration excess runoff in catchments.

Saturation excess flow typically occurs near streams or in hollows where water tables are closer to the surface and on heavier-textured soils where impermeable clay subsoils restrict water percolation, causing the soil surface to become saturated even under mild storms. Near-stream areas showing saturation excess runoff expand outwards from the stream in a dynamic spatial and temporal pattern depending on hillslope topography and storm duration. The distribution of slowly permeable soils rather than land use is the primary factor affecting the extent of saturation excess runoff within catchments, which is thus a more important pathway of P loss than infiltration excess runoff (Needleman *et al.* 2004). Although surface runoff and erosion can occur along hillslopes within individual fields, especially where the soil is compacted or where tramlines run up and down the slope, this is not considered a ubiquitous problem in the UK and often any runoff/erosion generated is retained within the field. The exception is where roads intercept field runoff and transport it directly to the watercourse.

Sub-surface runoff refers to the component of rainfall that infiltrates the soil and moves to the stream above the main groundwater level in timescales ranging from minutes to days. In many agricultural soils, artificial drainage systems (tile, plastic and mole drains) intercept sub-surface runoff, greatly reducing these transit times. Where underdrainage systems are present, most of the stormwater is directed through the field drains and most of the P is therefore delivered with it. Numerous studies have indicated that concentrations of soluble and particulate P are lower in drainflow than in surface runoff due to the entrapment of particulates and the adsorption of soluble P to the subsoil during passage through the soil. The reduction in soluble P is about one-third (Haygarth *et al.* 1998; Withers *et al.* 2009a). What remains is delivered quickly to the watercourse with little opportunity for further P retention depending on the network of ditches and drainage design. Slower percolation of P through the soil matrix can reach shallow groundwater in some areas with P-saturated soils (see Chardon and Schoumans, 2007), but this is not the major transport pathway in most catchments in the UK.

Typically, most of the phosphorus is transported during a few major events and seasonality is largely dominated by the pattern of storm events during the year. For example, Sharpley *et al.* (2008) found that over a 10-year period, storm events accounted for a third of flow, two-thirds of soluble P export and four-fifths of TP export, with the majority of storms having a return period of less than a year. Rainfall distribution is therefore an important factor governing whether runoff P concentrations are greatest during summer (see Donohue *et al.* 2005), or during winter (see McDowell and Wilcock, 2007; Withers and Hodgkinson, 2009).

2.2.2 Non-agricultural sources of phosphorus

2.2.2.1 Drivers

Phosphorus export to rivers from non-agricultural sources comes from both natural and anthropogenic sources, and losses show significant spatial and temporal variations. One of the largest drivers for anthropogenic P losses is increases in the human population and the greater degree of connectivity to the sewerage network. Increases are countered by a greater extent of sewage treatment to remove P from sewage effluents discharged to surface waters.

Sewage generated by the water industry is thought to be the largest single source of bioavailable P with approximately 40 million tonnes of domestic sewage (organic waste prior to dilution with tap or rain water) produced annually in the UK (CEFIC/CEEP, 1998). These wastes are estimated to contain around 45,000 tonnes of phosphorus (as P), largely derived from human wastes (average of 0.44 kg P per capita) and other household activities, including contributions from household detergents (a further 0.48-0.68 kg P per capita) (Dils *et al.* 2001).

The population of the UK has shown a steady increase in the last 10 years, from 59.1 million in 2001-2 to 61.7 million in mid-2009. The rate of increase has grown in the past few years, to 0.6 per cent per year (394,000 people) between 2001 and 2009 compared with 0.3 per cent per year between 1991 and 2001. These increases have been influenced by changes in birth and death rates, and international migration (Office for National Statistics, 2010).

This increase in population not only entails the need for greater food production, but also boosts demand for sewage services. An extra 394,000 people corresponds to an extra 173 tonnes of P in sewage per year. The rising population could also boost levels of P in wastewater from household detergents by a similar amount. This increases the pressure on wastewater treatment works (WwTW) and STW, and could push up treatment costs. In rural areas, greater numbers of septic tanks as new houses are built could also have an impact on water quality.

Legislation and cost savings or cost recovery from P recycling can help to improve P levels. Global drivers include changes in the market for recycled P, and new requirements or targets for businesses and governments. Concerns over sustainability and the environment may also boost P removal and efficiencies in P use.

Regulation is helping to reduce P in non-agricultural sources. The Urban Waste Water Treatment Directive (UWWTD) imposes concentration limits of 1 or 2 mg Pl^{-1} in effluent discharges at STW which serve more than 10,000 people. Section 2.5.1 provides further detail on the different legislative and policy drivers.

Reducing the amount of P entering sewers is a priority for Defra. It Five to ten per cent of P pollution in England is estimated to come from detergent use, and Defra is working with the UK cleaning products industry to phase out the use of phosphates in laundry detergents by 2015 (Defra, 2008a, 2008b).

Environmental concerns relating to levels of P are wider-reaching than just eutrophication. Current extraction and processing of phosphorus is not sustainable, and produces hazardous by-products. Raw phosphorus sources are diminishing, and processing will become more difficult and expensive as quality drops (Rogner, 2009). This is likely to increase pressure to develop ways of recycling P from sewage waste.

2.2.2.2 Sources

Background or natural sources of P are generally low, but can be significant where anthropogenic P inputs are low. Sources of background P can include atmospheric deposition, soil weathering, river bank erosion, riparian vegetation and migratory fish returning to spawning grounds. The concentrations of P from these sources are generally low (<0.1 kg P ha^{-1} yr^{-1}), and loads are generally in particulate forms, but they may still provide substrates for microbial synthesis in upland, forested and oligotrophic streams (Withers and Jarvie, 2008).

Anthropogenic P inputs come from both point and diffuse sources. Point sources of P include wastewater, septic tank and industrial effluents, and generally contain a high proportion of soluble, more bioavailable P. Diffuse sources include runoff from roads and urban areas, and generally contain a high proportion of particulate forms of P.

Point sources

1) Domestic and industrial wastewaters

Domestic wastewaters are discharged from STW after being treated to remove contaminants. In some STW this includes tertiary treatment to remove P, down to levels as low as 1 or 2 mg l^{-1}. Industrial discharges may also be subject to quality consents (Environment Agency, 2009a). The levels of P in these discharges vary depending on the population served and the extent of industrial activity, and effluent P concentrations can range from <1 to 20 mg l^{-1}.

Neal *et al.* (2005) studied six STW in a rural part of the Upper Thames, finding a mean P concentration of 6.6 mg l^{-1}, with a range of 0.2-17.1 mg l^{-1}. Of this P, around 91 per cent was in soluble reactive form, and three per cent in particulate form. Total loads in the catchment were between 0.1 and 2.9 t yr^{-1}, around 0.5 kg per person per year. These loads were low compared with those found by Bowes *et al.* (2005) in the Warwickshire Avon catchment, where STW loads were between 0.1 and 548 t yr^{-1}. These loads were estimated to represent between nine and 93 per cent of the total P load.

A further issue with domestic wastewater discharges is the presence of misconnections. In the UK, foul water is collected and taken for treatment, while surface water from rainfall is collected and allowed to discharge directly to a receiving watercourse. A misconnection is defined as “any direct discharge of foul wastewater to a separate surface water sewer, or of surface water or groundwater to a separate foul sewer”. At present, the extent of misconnections is not fully known, so their impact on river waters cannot be determined. However, it has been estimated that between 0.6 and two per cent of UK households are currently misconnected, equivalent to around 300,000 misconnections (Dawe, 2010).

Detergents are discharged in both household and industrial effluents. Of the 78 per cent of the total P load to the water environment from households in England (White and Hammond, 2009), around 16 per cent comes from dishwasher and laundry detergents (Defra, 2008b). Detergents contribute up to four per cent of the P load from non-agricultural sources (see Table 2.3).

2) Septic tanks

Statistics on the proportion of septic tanks across the UK have been obtained by integrating water company service data with population census data. Using this approach, about 1.2 million properties in England and Wales are estimated to have septic tanks, which equates to about five per cent of the population (May *et al*. 2010). These homes are generally located in rural areas, and subsequently use 'stand-alone' private sewerage systems. This term encompasses different types of systems including pre-fabricated package plants, cesspools, reed beds and most commonly, septic tanks.

Septic tanks are the preferred method of wastewater treatment for single dwellings. Such systems are designed primarily for the settlement of solids and removal of biological oxygen demand (BOD) and nitrogen via anaerobic digestion inside the tank. The quality of the effluent may be further improved where septic tank effluent is discharged to a soakaway or reed bed. As a result, the preferred method of discharge from septic tanks is to groundwater via a soakaway or reed bed system. Discharges from septic tanks directly to surface waters are prohibited.

When the Environmental Permitting Regulations 2010 were implemented in England, a registration scheme for small sewage discharges, such as from septic tanks, was thought to be the least burdensome way of controlling such discharges. Registration exempts the need for an environmental permit where the risk of pollution is low. Deadlines for registration were set out and for discharges to groundwater the registration deadline was 1 January 2012. However, Defra is currently working with the Environment Agency to ensure that this approach is the most appropriate and whether it could be further simplified. Whilst the review is on-going the Environment Agency will not pursue registration for a small sewage discharge in England where the:

- discharge is to ground and is two cubic metres per day or less via a septic tank and infiltration system (soakaway) and is outside a Source Protection Zone 1 - this is approximately equivalent to nine people occupying a single property;
- discharge is to surface water and is five cubic metres per day or less via a package sewage treatment plant - this is approximately equivalent to 31 people occupying a single property (for example, a small school, residential home and so on);
- sewage is only domestic;
- sewage system is maintained in accordance with the manufacturer's instructions and the owner keeps a record of all maintenance - in the case of septic tanks this includes regular emptying;
- discharge does not cause pollution of surface water or groundwater.

Despite the registration scheme, the number, whereabouts and condition of many small unconsented discharges are unknown. At present the Environment Agency holds 55,000 consents for this type of discharge. However, an additional 500,000 unconsented discharges (mostly to ground) may exist in England and Wales (Environment Agency, 2009b). In rural areas the proportion of households not connected to the main sewerage network is thought to be significantly higher than at national level.

Many older systems discharge directly into streams or do not have adequate soakaway facilities, and can make significant contributions to in-stream P concentrations. In addition, awareness of the need to regularly empty such systems is limited, which can further increase P loads. Concentrations of TP from septic tanks have been measured at 1-22 mg l^{-1}, with a mean value of 10.2 mg l^{-1} (Edwards and Withers, 2008). Concentrations in septic tank effluent measured by Withers *et al.* (2011) had a mean value of 3.1 mg TP l^{-1}, with a range of 0.2-20.6 mg l^{-1}. Of this, around 80 per cent was soluble reactive P, and nine per cent was particulate P. These concentrations are similar to those in STW effluent, although the volume of effluent is much smaller. Withers *et al.* (2011) found that a septic tank input had a statistically significant impact on total and soluble reactive concentrations of P in the catchment studied. May *et al.* (2010) concluded that septic tanks may contribute up to a fifth of catchment P loads in some areas.

Diffuse sources - Urban/highway runoff

Impervious surfaces, including roads and urban areas, can act as significant sources of phosphorus. Studies have shown that ecosystems are affected when as little as 10-15 per cent of the catchment area is covered by impervious surfaces (Withers and Jarvie, 2008). Potential sources of P in urban and highway runoff include: atmospheric deposition, leaf fall, industrial debris, urban litter, chemicals such as de-icers and rust inhibitors (US Dept. of Transportation, 1999), detergents and lubricants in surface drains, and excreta from birds and domestic animals such as dogs.

P concentrations have been shown to vary significantly between urban and highway runoff in urban and rural areas. Withers and Jarvie (2008) noted average concentrations of 2.39 mg l^{-1} on rural roads and 0.29 mg l^{-1} on urban highways, with both highly dependent on rainfall. In rural areas, around 90 per cent of this P was in particulate form. Mitchell (2001) measured event mean concentrations (EMC) for TP in 765 catchments, finding results between 0.2 and 0.4 mg l^{-1}, around half of which was soluble reactive P. Leaf fall and roadside gullies are significant sources of soluble forms of P, while motorways and roads deliver lower proportions of soluble P than road runoff in residential areas.

Withers *et al.* (2009a) found wide ranging concentrations of P in road runoff, with a range of 0.1-16 mg l^{-1} and a mean of 2 mg l^{-1}. This range was apparent in all forms of P, with the highest proportion (up to 90 per cent) of the total P load in particulate form. Concentrations of suspended sediment were also high, indicating that this runoff can be turbid and reflecting the rapid response associated with impervious surface runoff. Significant differences were found between the two sites investigated.

Several studies have shown that concentrations of contaminants in highway runoff are well correlated with the number of vehicles travelling on highways during storms, although this has not been investigated for P specifically. Runoff may also be influenced by the amount of impervious area, frequency, intensity and duration of storms, and any measures to remove P (drainage ditches, grass swales and so on) (Michigan Dept. of Transport, 1998).

Summary

As shown above, concentrations of P in STW discharges can be very high, reaching 20 mg l^{-1} at some works. Concentrations of P in urban runoff show similar levels, but in general the total P load is lower. WRc (2007) found that in the River Ribble catchment, in the north-west of the UK, P loads from urban and highway runoff were small, contributing around three per cent of the total P load, compared with 88 per cent from STW discharges and nine per cent from agricultural sources.

A further estimate of the proportion of P loads from non-agricultural sources in England and Wales in 2005 is given in Table 2.3. These figures were derived for the WFD Preliminary cost-effectiveness analysis for the WFD.

Table 2.3 Phosphorus losses in England and Wales from non-agricultural sources in 2005 (from ICF International, 2007)

Source	**P loss t yr^{-1}**	**Percentage contribution**
STW	39,970	91.9
Misconnections	1,438	3.3
Industrial discharges	434	1.0
Detergents	1,660	3.8
Total loss	43,502	100.0

The impact of P sources on eutrophication varies between sources. Runoff from roads and urban areas is closely related to rainfall, and is likely to be highest in the winter months when river levels are at their peak and dilution levels are increased. As such, the non-agricultural sources of P which are likely to have the most significant impact on ecological status are those which discharge at a constant rate, STW and industrial discharges, septic tanks and misconnections. Misconnections may be particularly significant, as they contain the same pollutants as STW, but have not undergone any form of treatment.

2.2.2.3 Pathways

Domestic and industrial wastewater sources most commonly enter water sources through direct discharges. In the case of misconnections, this means the water has not been treated, and water quality is likely to be poor. While STW and industrial discharges have been treated, any direct effluent input can have a significant impact on the ecology of a river.

Septic tank discharges may reach watercourses through three main pathways: soakaways, field drains and direct discharges. Soakaways, also known as infiltration systems, are designed to disperse the septic tank effluent into the soil surrounding them. When this happens, a proportion of the P is retained and adsorbed onto the surrounding soils, reducing the amount which reaches the watercourse. Soakaway design depends on the surrounding soil, and a percolation test must be undertaken before installation. Where percolation is too slow, alternative methods of disposal must be used. Gradual deterioration of the functionality of soakaway areas on impermeable soils due to build-up of effluent may also lead to failure.

Direct discharges to watercourses are unusual, but as septic tanks provide only a basic level of treatment in most cases, these can be significant. In such cases, some form of secondary treatment pathway, such as a reed bed, may be necessary to improve water quality.

Urban stormwater can contain P from runoff from roofs, roads, driveways, car parks, construction sites and gardens. Runoff from urban surfaces is generally connected to rivers in one of two ways: either directly, via surface water outfalls, or indirectly via the sewage system and combined sewer overflows.

Like runoff from agricultural land (Section 2.2.1.3), patterns of runoff from urban areas and highways are storm-dependent, with the highest P inputs occurring during storm events. However, these sources still represent a semi-continuous supply of P throughout the year, as impervious surfaces have no infiltration capacity (Withers and Jarvie, 2008)

2.2.3 Quantifying source pressures

Research suggests that rates of P loss from agricultural land are highly sensitive to land use and the management of crops, livestock and P inputs at the field scale where flow is initiated. At the catchment scale, flow accumulates in larger quantities and the effects of individual practices at the field scale become greatly diluted unless a critical density of these farming operations occurs within the catchment. A source only becomes a pressure when that source is exposed to storm events (the critical source area concept); for example, fertilizer and manure applications. Page *et al.* (2005) found that flow pathways in one catchment did not coincide with areas of high soil P.

The large spatial and temporal variability in P mobilization and transport across a catchment makes accurate source apportionment difficult. A range of methods have been developed to overcome this problem, from simple export coefficients related to land use and livestock numbers (Johnes, 1996) to risk-based analysis of the multiple source and transport pressures operating at a site (Buczko and Kuchenbuch, 2007), to process-based modelling of source pressure variables (Davison *et al.* 2008), empirical modelling of flow-concentration and flow-load relationships (Bowes *et al.* 2008; Jarvie *et al.* 2010) through to uncertainty analysis based on expert opinion and on-site investigation (McDowell *et al.* 2009; Schärer *et al.* 2006). Since source pressure is so heavily site-specific, desk-based approaches, while offering some quantitative estimate of *potential* source contributions, can only indicate potential sources and areas to investigate further. Identification of actual source pressure requires a site visit during wet weather to pinpoint ‘hotspots’. However, such site visits do not enable any quantification of source contribution. A combined approach is therefore required (Mainstone *et al.* 2008).

2.3 State

2.3.1 Phosphorus processes and behaviour in freshwaters

Understanding and assessing the nutrient status of freshwaters is complicated by the numerous chemical forms that phosphorus can take and the complex biogeochemical transformations that take place in rivers and lakes.

Phosphorus can exist in many forms, including inorganic soluble forms, organic bioavailable and non-available forms, and attached to particulates. This latter condition includes ‘labile’ phosphorus that is only loosely attached to particles, and firmly bound phosphorus that has been incorporated into the matrix of the particles. Phosphorus immediately available for plant growth (bioavailable phosphorus) essentially consists of inorganic soluble forms, organic forms of low molecular weight, and labile phosphorus that can rapidly desorb from particulates under certain conditions. Phosphorus in complex organic compounds will be released as the compounds are broken down by biological processes, whilst the long-term availability of phosphorus firmly bound to particulates is unclear (Mainstone *et al*. 2000).

Once in the river, phosphorus is chemically and biologically active, undergoing numerous transformations and moving between the particulate and dissolved phases, between the sediment and water column, and between the biota and abiotic environment (Withers and Jarvie, 2008). Figure 2.2 illustrates the main processes driving P cycling in freshwaters.

Phosphorus can move between the water column and bed sediment in three ways: by physical deposition/resuspension of particulates, by sorption/desorption to/from particulates, and by diffusion between pore water and the main water column (Mainstone *et al.* 2000). In the water column, labile phosphorus attached to suspended particulates can rapidly desorb into the water column and become bioavailable. This dissolved phosphorus can be incorporated into inorganic phosphate minerals by precipitation, particularly in association with calcium (in high alkalinity waters) and iron and aluminium (in low alkalinity waters) (Mainstone *et al*. 2000; Withers and Jarvie, 2008).

Biological processes also play an important role in phosphorus cycling (Withers and Jarvie, 2008). Rooted aquatic plants are capable of deriving much of their phosphorus requirements from the sediment, but can also take up soluble phosphorus directly from the water column. Filamentous (such as *Cladophora*), epiphytic, epilithic and planktonic algae generally remove phosphorus directly from the water column, whereas benthic algae may use both sources (Hilton *et al*. 2006). Microbial uptake from the water column and particularly within the sediment can also be substantial. Phosphorus is released back into sediment pore waters and the water column by decay of plant material and the mineralisation of organic matter by the microbial community (Mainstone *et al*. 2000).

The balance of these physico-chemical and biological processes and their relative contribution to phosphorus flux and downstream delivery varies in time and space according to stream size, catchment geology and channel hydromorphology (reviewed by Withers and Jarvie 2008). In small headwater streams loss of SRP through a combination of biological assimilation into organic forms and sediment uptake can significantly restrict downstream phosphorus supply, whereas in larger rivers uptake of SRP from the water column by phytoplankton and retention of phosphorus in bed sediments, on floodplains and in reservoirs are more significant. Significant uptake of largely soluble P discharged from a STW or septic tank by bed sediments can occur within a relatively short distance and be re-released to the water column under low flow conditions (see Palmer-Felgate *et al.* 2010). Catchment geology influences the quantity and composition of suspended and bed sediments. Fine clay or silt sediments with high concentrations of iron and aluminium oxides have a particularly high capacity for phosphorus sorption. River hydrology and geomorphology have a major influence on P cycling because they control flow velocity and the contact time between the streamwater and P exchange surfaces. For example, higher water velocities promote resuspension and entrainment of particulate P and reduce water residence times for uptake of P by bed sediments and biota.

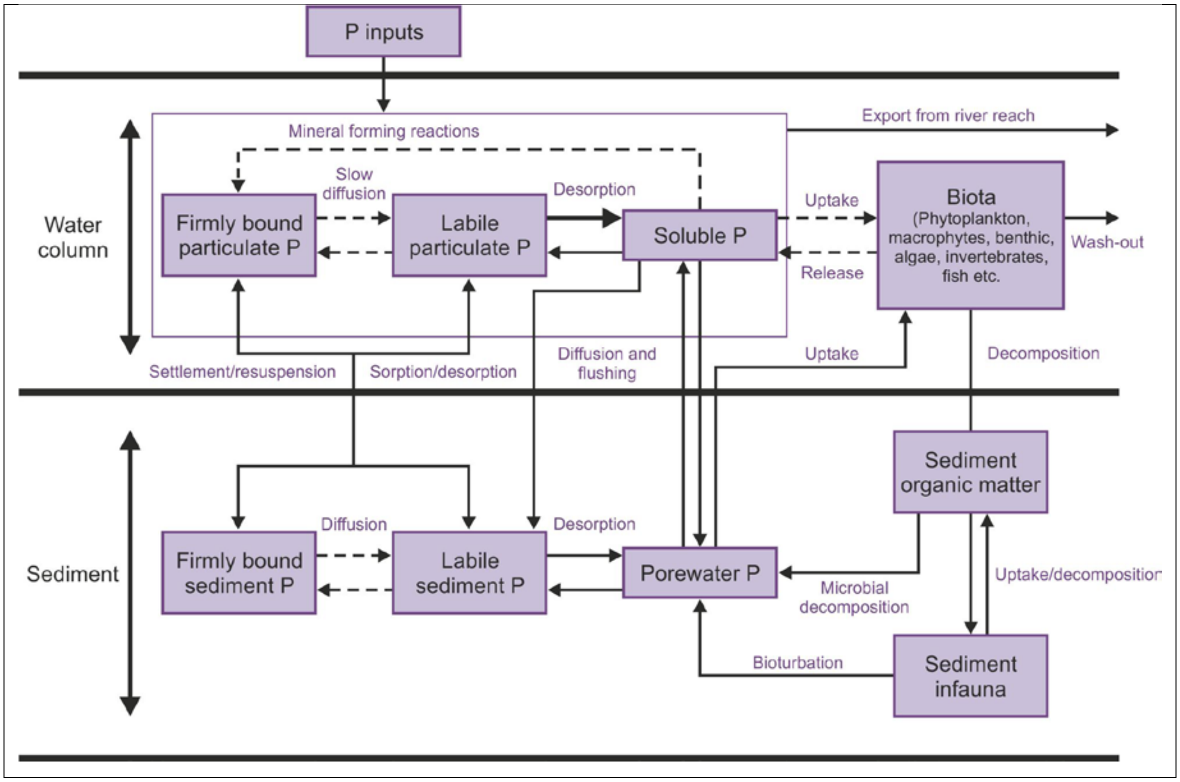


Figure 2.2 Phosphorus behaviour in rivers (after Mainstone *et al.* 2000). (Width of line indicates importance of pathway, dotted lines indicate pathays that are usually minor or are of unknown importance)

2.3.2 Measuring status

The nutrient status of rivers and lakes is assessed by routine monitoring of phosphorus concentrations in the water column. Various analytical methods separate out different fractions of the phosphorus present in the soil or river water sample collected. The procedures for determining the more common fractions of phosphorus are indicated below. The dissolved/soluble fraction consists of inorganic orthophosphate which is considered to be immediately available for biological uptake, and some organic compounds and complexes which are not immediately bioavailable. Although particulate P (PP) is not immediately biologically available, it can provide a long-term source of phosphorus for aquatic plant growth. The particulate fraction is worked out indirectly by difference: that is, total phosphorus (TP) minus total dissolved phosphorus (TDP).

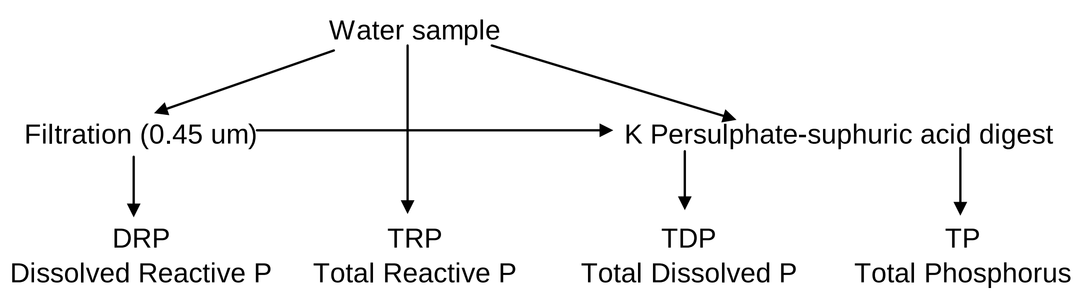


Figure 2.3 Separation of phosphorus fractions in a water sample

The Environment Agency routinely measures orthophosphate which is often referred to as **total reactive phosphate** (TRP). The sample is not filtered. Due to the chemicals used to produce the colorimetric reaction, a small fraction of condensed phosphates and phosphates bound to complex inorganic and organic compounds are hydrolysed (broken down) during the reaction. The result for total reactive phosphate is thus an overestimation of the orthophosphate in a water sample.

Less frequently, the Environment Agency monitors filtered orthophosphate (sometimes referred to as **soluble or dissolved reactive phosphorus (SRP/DRP**). DRP is a measure of the filterable (soluble, inorganic) fraction of phosphorus, the form directly taken up by plant cells. Filtered orthophosphate is derived by the same procedure as total orthophosphate, except that the sample is passed through a 0.45 µm filter before analysis to remove some of the labile particulate and organic phosphorus.

Total phosphorus is a measure of the total amount of phosphorus in a water sample, including suspended and dissolved inorganic and organic fractions. This is useful for characterising the water body and assessing catchment condition. This measure can be used in conjunction with flow rates to provide load-based estimates of phosphorus being transported downstream but does not indicate the fraction of phosphorous that is available for direct biological uptake, thus limiting its value as an indicator for ecological risk. The sample must be digested to measure total phosphorus.

Total dissolved phosphate is a measure of the filterable (soluble) fraction of phosphorus. TDP is derived by passing the sample through a 0.45 µm filter and digested before analysis.

Since 1990, the Environment Agency has routinely monitored concentrations of phosphate, measured as total reactive phosphate (TRP), and nitrate in rivers in England and Wales as part of its General Quality Assessment (GQA) scheme. The headline indicator is the proportion of rivers (by length) with annual mean concentrations greater than 0.1 mg l^{-1} for phosphate and 30 mg l^{-1} for nitrate (Environment Agency, 2010a). TRP is derived by the same analytical procedure as soluble reactive phosphate (SRP) but without first filtering the sample, thereby incorporating any labile phosphorus attached to particulates that desorbs during analysis (Mainstone *et al*., 2000). Work undertaken for the ECSFDI project and to support the development of WFD standards has examined the differences between SRP and TRP. The conclusions of this work are that the differences are small, at least until the concentration is significantly below 0.1 mg l^{-1}. The main reason seems to be that when the unfiltered samples are analysed in the laboratory, they are not shaken, so little particulate P is included in the analysis. The differences between SRP and TRP are in reality small, especially in catchments such as the Bourn Brook where soluble P is above 0.5 mg l^{-1}. A recommendation of this project is that the Environment Agency reconsiders the need to filter samples in the field and perform a more defensible analysis for soluble P. This can be done by analysing samples taken in the Bourn Brook and Harpers Brook for both TRP and SRP.

More recently, the UK Technical Advisory Group (UKTAG) has developed new WFD standards for SRP in rivers (UKTAG, 2008a). These new standards are designed to prevent adverse changes to the most sensitive biological element, diatoms, and are expressed as the annual mean concentration of SRP in the water column. Unlike GQA, which sets a single, uniform standard for all rivers, WFD standards are set according to the sensitivity of the river to eutrophication, based on altitude and alkalinity. The four river types and related threshold values for annual mean SRP for 'high' and 'good' status are set out in Table 2.4. Problems have been found with the data used for Type 3n, so at present the same standards are used for Type 3n as for Type 4n. UK river and lake P standards are to be reviewed by spring 2012, for use in the second River Basin Management Plan (RBMP) cycle. Reviews of standards are in accordance with the "checking procedures" in EU WFD classification guidance, whereby type-specific nutrient standards can be relaxed or tightened to ensure that they are no more or less stringent than needed to support the biology.

Although nitrogen also plays a role in eutrophication in some freshwaters, UKTAG considered the current understanding of this to be insufficient for it to be used as a basis for setting standards or conditions. No standards for nitrogen in rivers are set at present (UKTAG, 2008a).

In addition to the WFD standards, the Environment Agency, Countryside Council for Wales and Natural England have developed phosphorus standards for rivers (Table 2.5) as part of the process of reviewing permit conditions in order to meet the requirements of the Habitats Directive (UKTAG, 2008a). In contrast to WFD standards, Habitats Directive standards are expressed as annual mean concentration of total reactive phosphorus (TRP). The Habitats Directive standards are set according to river size, with tighter standards for smaller headwater streams. Although it is not possible to directly compare the WFD and Habitats Directive standards, the threshold concentrations are broadly similar and provide a similar level of protection. If an area has more than one EU designation, the more stringent standard applies.

Table 2.4 WFD standards for phosphorus in rivers (UKTAG, 2008a)

River typology	Altitude	Annual mean alkalinity (mg l^{-1} $CaCO_3$)	SRP annual mean (mg l^{-1})	
			High	Good
Type 1n	Under 80 m	<50	0.03	0.05
Type 2n	Over 80 m	<50	0.02	0.04
Type 3n	Under 80 m	>50	0.05	0.12
Type 4n	Over 80 m	>50	0.05	0.12

Table 2.5 Habitats Directive guideline phosphorus standards for rivers (UKTAG 2008a)

River typology	TRP annual mean (mg l^{-1})		
	Natural (mid-high)	Guideline (mid-good)	Threshold (just into moderate)
Headwaters	0.00-0.02	0.02-0.06	0.04-0.10
Most rivers	0.02-0.03	0.04-0.10	0.06-0.20
Large rivers	0.02-0.03	0.06-0.10	0.10-0.20

For lakes, UKTAG proposed a standard expressed as the mean annual concentration of total phosphorus (TP). TP includes bioavailable phosphorus, phosphorus bound to particulate material and that contained in phytoplankton. TP was selected in preference to SRP because it is an established indicator for eutrophication in lakes and is easy to measure (UKTAG, 2008b).

In lakes, phytoplankton biomass responds most closely with P levels, as opposed to diatoms in rivers. This measure is most closely represented by total P, rather than SRP which is used in rivers and has a strong relationship with diatoms (Stutter and Demars, no date). Total P is also a less variable measure than SRP, which shows significant seasonal variation, with levels falling in spring when up to half of the SRP load could be removed by biological uptake (House and Casey, 1989). Dissolved P may also be taken up by sediment, converting to less bioavailable PP. As such, TP is a better measure of P inputs, as it is less affected by internal partitioning.

Lakes tend to become more eutrophic with (geological) time, as they become shallower due to build-up of silt, which can contain significant levels of P, and naturally show a wide variation in trophic status and sensitivity to nutrient enrichment (Hilton *et al.* 2006). This natural variation makes assessment of the extent and intensity of eutrophication difficult and high phosphorus concentrations alone are not necessarily indicative of anthropogenic eutrophication. The standards proposed are therefore specific to individual lakes and are set on a lake-by-lake basis using a model that takes into account the alkalinity, depth and altitude of the lake (UKTAG, 2008b). Due to uncertainty in the links between ecological change and phosphorus in lakes, UKTAG advised that the standards should be interpreted with caution and that actions to improve lakes should be taken where ecological data from biologically sensitive elements confirm that there is, or that there is likely to be, a problem (see Section 2.4.3 for methods of assessing biological impacts).

Under the WFD, monitoring of P is carried out under three programmes, namely surveillance monitoring, operational monitoring and investigative monitoring. Surveillance monitoring is used to supplement the impact assessment procedure and assess long-term changes in conditions from both natural and anthropogenic causes. Operational monitoring is used for classification of water bodies, identification of sites likely to fail objectives and assessment of any changes due to programmes of measures (PoM). Investigative monitoring is used where the reason for exceedance is unknown, where surveillance monitoring indicates that objectives are unlikely to be met, to ascertain the magnitude and impact of accidental pollution or to inform the establishment of a PoM to combat the effects of such pollution.

The methods used to routinely monitor P in rivers and lakes in England and Wales provide a standardised and simple assessment of nutrient status but have significant limitations. First, measurements focus on just a single P fraction (SRP or TP), which provides limited information about fluxes or cycling of phosphorus within these systems. Second, concentrations of phosphorus are measured only in the water column, and not in the sediment, so it is not known to what extent water column concentrations may be buffered by phosphorus in the bed sediments. Finally, nutrient conditions are expressed as an annual average concentration, which ignores seasonal variation in phosphorus loading and sensitivity to nutrient enrichment. In particular, diatoms and macrophytes have a restricted growing season and are most responsive to nutrient conditions during the spring and summer months.

2.4 Impact

2.4.1 Response of receptors to elevated P

Phosphorus occurs naturally in freshwaters and is an important nutrient for the growth of algae and higher plants (macrophytes). In the absence of anthropogenic impacts, the concentration of SRP in rivers is naturally very low (<0.03 mg l^{-1}, often <0.01 mg l^{-1}; Mainstone *et al*. 2000) and depends upon the ability of the system to retain nutrients as they are continually taken up and released by the nutrient spiralling processes described in Section 2.3.1. Rivers and lakes in England and Wales are naturally deficient in biologically available phosphorus (Reynolds and Davies, 2001) and phosphorus is widely acknowledged to be the major limiting nutrient in such systems (Hecky and Kilham, 1988; Mainstone *et al.* 2000). The type of limiting nutrient in a system is also likely to vary depending upon the plant community and trophic state of the river. In addition, it is thought that P only limits the growth of algae until it reaches a concentration of around 0.1 mg SRP l^{-1}, or less in some areas (Bowes *et al*., 2011). Above this level, additional P has no extra effect, and other factors become limiting to growth. All requirements for plant growth, such as light level and trace nutrient concentrations, must be in excess for plants to achieve their full growth potential (Hilton *et al.* 2006).

Elevated levels of this key limiting nutrient above natural background levels can therefore trigger a series of changes in the ecology of these systems. The process by which elevated nutrients impact ecosystems is known as eutrophication.

Although eutrophication is generally perceived to have a negative effect on ecosystem structure and function, the response of stream biota to elevated P is expected to be non-linear, as described by subsidy-stress theory (Odum *et al*. 1979). Increased P loading may therefore initially have a benign effect ('subsidy') on the productivity and diversity of the ecosystem, whilst longer-term exposure to excessive P supply will have undesirable consequences for ecosystem services ('stress'). In some special cases the maintenance of a nutrient subsidy may be justified, for example to maintain a productive fishery, but in most cases the subsidy-stress threshold is exceeded and the effects of eutrophication are undesirable.

Algae and macrophytes are the primary receptors of P and increased exposure to bioavailable forms of phosphorus, particularly SRP, accelerates their growth and enhances productivity. This alters the competitive balance between species, which can have knock-on consequences for the composition and diversity of these communities and associated fauna and adversely affect water quality (Smith *et al.* 1999; Mainstone *et al.* 2000). From a mechanistic point of view, elevated phosphorus concentrations directly:

- accelerate the growth of macrophytes, leading to an increase in overall biomass;
- accelerate the growth of epiphytic, epibenthic, filamentous and planktonic algae (increased phytoplankton biomass/abundance and blooms of blue-green algae are some of the most apparent impacts in lakes);
- alter the composition of algal and macrophyte communities by favouring species with high nutrient requirements and tolerance;
- reduce the rooting depth of macrophytes, thereby making them more susceptible to being ripped out of the substrate under high river flows.

These impacts can have important knock-on effects. Changes in the plant community and excessive plant respiration leads to severe nocturnal sags in dissolved oxygen levels that stress oxygen-sensitive invertebrate and fish species. In upland streams and rivers, excessive growths of epilithic algae traps fine particles, leading to siltation and spawning difficulties for species such as brown trout and Atlantic salmon (Mainstone *et al.* 2000). Enhanced growth of algae can have complex and far-reaching impacts on ecosystem structure. For example, excessive growth of epiphytic algae on macrophytes restricts carbon uptake, shades leaves and stems and reduces photosynthesis. Positive feedback loops then cause lakes to flip between alternative stable states, from a natural clear-water state dominated by submerged higher plants to a disturbed, turbid state dominated by phytoplankton (Hilton *et al*. 2006). The competitive balance between higher plants and algae is highly complex but the likelihood of a switch from one state to another is fundamentally dictated by phosphorus availability (Mainstone *et al.* 2000).

Conceptual understanding of the mechanisms controlling the rate and severity of eutrophication is much further advanced for lakes than for rivers. Indeed, the processes by which increases in a limiting nutrient can increase the biomass of phytoplankton in lakes are sufficiently well understood that a number of reliable models have been developed, capable of predicting changes in phytoplankton biomass and community structure over time (for example PROTECH, Reynolds *et al.* 2001). By contrast, Hilton *et al.* (2006) argue that there is currently no conceptual understanding of eutrophication in rivers and propose a model that emphasises the importance of hydraulic flushing in controlling algal growth. They hypothesise that rivers with short retention times respond to increasing nutrients by increased growth of benthic, filamentous and epiphytic algae, whereas long retention time, impounded rivers operate more like lakes, moving from macrophyte domination to phytoplankton domination. These, and related, predictions have yet to be fully tested.

2.4.2 Factors affecting sensitivity to elevated P

Phosphorus is widely acknowledged to be the main cause of eutrophication in freshwaters but in some systems the additional phosphorus can raise the biological supportive capacity to the level of the next limiting factor (carbon, light, hydraulic retention or another nutrient). At high phosphate loadings, the linear stoichiometric yield relationship between biomass yield and phosphorus availability breaks down (Reynolds and Davies, 2001), suggesting that the effects of phosphorus enrichment can sometimes be constrained by other factors such as light availability (tree shading), flow or nitrate availability (Hilton *et al.* 2006; Bowes *et al.* 2007). Nitrate limitation or nitrate-phosphate co-limitation can occur in some oligotrophic, acidic upland lakes (Maberly *et al.* 2002) and some hypertrophic lakes with large phosphorus inputs, such as the Shropshire and Cheshire meres (James *et al.* 2003). However, a significant proportion of the UK's rivers have high nitrate levels, so nitrate limiting is less likely in these areas.

The health of diatom communities is significantly affected by the availability of silicon, which is a controlling factor in cell growth. To maximise diatom growth and photosynthesis, diatoms need nutrient levels close to the Redfield ratio: 15 silicon (Si) to 16 nitrogen (N) to one phosphorus (P) (Lutz, 2010). In transitional and coastal waters the N:P ratio is usually less than 16:1, making nitrogen the limiting nutrient.

Although concentrations of dissolved phosphorus exceeding 0.1 mg l^{-1} are generally considered to be indicative of eutrophication, the sensitivity of rivers and lakes to eutrophication is influenced strongly by water chemistry and hydrology (Bowes *et al.* 2007). Non-calcareous (soft water) catchments are more sensitive to phosphorus enrichment than calcareous (hard water) catchments. In addition, dissolved inorganic carbon (DIC) can influence community structure in shallow lakes, altering competitive interactions between epiphytic algae and macrophytes and rendering low DIC lakes more prone to loss of macrophytes when nutrient loading increases (Jones *et al.* 2002).

Impounded rivers and the lower reaches of large rivers are at greater risk of eutrophication than smaller, faster-flowing headwater streams for several reasons: (i) deposition of sediment leads to the accumulation of a reservoir of in-stream particulate phosphorus, (ii) a longer retention time facilitates uptake of phosphorus by algae and macrophytes, (iii) an abundant inoculum of algae is usually available from upstream sources, and (iv) low velocities permit the epiphytic algae to accumulate and smother macrophytes (Hilton *et al*. 2006).

In lakes, grazing by zooplankton may reduce the sensitivity of the lake to enrichment (UKTAG 2008b) as reduced turbidity allows macrophytes to use more light and nutrients and attain dominance in a system. However, Jones and Sayer (2003) showed that, at low nutrient levels, algal biomass is not related to zooplankton numbers or nutrient concentrations, but to macroinvertebrate numbers. Macroinvertebrates graze on epiphytes, allowing macrophytes more light, which results in faster growth. This increased macrophyte biomass increases uptake of water-column nutrients (Hilton *et al.* 2006).

The close coupling of sediment-water phosphorus dynamics means that other anthropogenic impacts can exacerbate the effects of high phosphorus loading. For example:

- Abstraction of water reduces the volume of water available to dilute phosphorus inputs and increases the retention time of water in the river, giving receptors more time to take up phosphorus from the water column. Whereas high velocities can inhibit the colonisation of macrophytes by epiphytic algae, low velocities under reduced flow conditions can enhance rates of periphyton growth and increase the risk of epiphyte blooms developing (Franklin *et al*. 2008; Withers and Jarvie, 2008).

- Flow management that reduces the frequency or intensity of flooding can contribute to the build-up of bed sediment in rivers and lead to increased retention of particulate phosphorus.

- Channelisation reduces connectivity between the channel and its floodplain and reduces the capacity for in-channel and floodplain nutrient storage (Withers and Jarvie, 2008).

- High organic loadings affect redox conditions within deposited sediments, promoting anoxia which can cause release of SRP from bed sediments (Jarvie *et al*. 2008).

Thus, the interaction of multiple stressors makes it difficult to predict how an individual river or lake will respond to elevated phosphorus loading and means that reducing phosphorus inputs alone may not be sufficient to reverse ecological changes resulting from eutrophication.

As discussed in Section 2.2.2.2, the impact of P inputs depends heavily on the timing of contributions. Point source loads of P from STW, industrial discharges and misconnections are continuous or semi-continuous, while inputs from urban and agricultural runoff are storm-dependent and contribute more in the wet winter months (Withers and Jarvie, 2008). During these times, dilution in rivers is increased by high flow levels, and the ecological impact of the P inputs is likely to be low. During the summer months when flow levels are low, continuous P inputs from STW discharges can have a major impact on ecological quality.

In addition to the impact of timing of discharges, the form of P discharges also has an effect on ecological quality. Point source discharges have been shown to discharge the majority of their P in SRP form, which is highly bioavailable, while agricultural runoff contains higher levels of PP (Withers and Jarvie, 2008).

2.4.3 Measuring impacts of elevated P

2.4.3.1 Measuring impacts of elevated P in rivers

The primary receptors for elevated phosphorus concentrations in rivers are macrophytes and diatoms and the composition of these communities is used to quantify the impact of nutrient enrichment in freshwater streams and rivers. Prior to the advent of the WFD, macrophytes were monitored using a metric called the Mean Trophic Rank (MTR) and diatoms were assessed using a similar metric called the Trophic Diatom Index (TDI). To comply with the requirements of the WFD, these tools have since been adapted and extended and are now termed River LEAFPACS (for macrophytes) and River DARLEQ (for diatoms).

The MTR index involves a detailed botanical survey and assessment of the physical character of a section of river bank (usually 100 m, occasionally 500 m). Each macrophyte taxon present in the survey stretch is identified and recorded along with an estimate of its abundance (Species Cover Value, SCV) on a scale from one to nine, where 1 = <0.1%, 5 = 5-10% and 9 = >75% cover. Each taxon is also assigned a number between one and 10 on the basis of its sensitivity to nutrient enrichment (Species Trophic Rank, STR; 1= highly insensitive, 10 = highly sensitive). The MTR score is the average of the STR scores, weighted by the SCV for each taxon, and multiplied by 10 to give a score between zero and 100. Scores for sites are then interpreted on the basis that scores greater than 65 are unlikely to be eutrophic, scores from 25 to 65 are eutrophic or at risk of becoming eutrophic, and scores less than 25 are badly damaged by nutrient enrichment, toxicity or physical degradation. The MTR score is influenced by habitat type, and therefore it is most useful to interpret the score alongside physically similar sites of known high water quality (Holmes *et al.*1999).

The MTR approach has been adapted by River LEAFPACS, a multi-metric tool designed to detect the impact on macrophytes of nutrient enrichment, alterations to river flows and modifications to morphological conditions (Willby *et al.* 2009). One of its four sub-metrics is the River Macrophyte Nutrient Index (RMNI), which yields a score comparable to the MTR score. The other three sub-metrics measure taxonomic and functional diversity and alterations to hydromorphology. The key difference between LEAFPACS and MTR is that a reference score is produced for each site based on its alkalinity, altitude, gradient and distance from source. The reference score factors out natural spatial variation in macrophyte community composition and provides a benchmark representing macrophyte conditions under minimally impacted conditions. Dividing the observed score by the reference score yields an Ecological Quality Ratio (EQR) ranging from zero (bad status) to one (high status).

The TDI was developed to assist in the monitoring of rivers in response to the requirements of the Urban Waste Water Treatment Directive (Kelly and Whitton,1995). The method uses benthic diatom communities (also termed phytobenthos or periphyton) to assess water quality, with particular emphasis on nutrient enrichment. Trophic status is derived on the basis of the taxa present within a sample, with certain taxa being more sensitive than others to nutrient enrichment. Strictly TDI does not define trophic status, but Kelly and Whitton (1995) found a statistically significant relationship between TDI and SRP, measured at 70 sites in the UK, and the method has been widely used to assess spatial differences and temporal trends in trophic status.

Benthic diatom films are collected from natural or artificial substrates within the sample site, and slides of the diatom sample are analysed in the laboratory to identify the taxa present and estimate their relative abundance. The TDI uses 86 taxa that are sensitive to nutrient status.

Each taxon is assigned a value between one (favoured by low nutrient concentrations) and five (favoured by very high nutrient concentrations) and a weighted average is again used to yield a TDI score ranging from zero (indicating very low nutrient concentrations) to 100 (indicating very high nutrient concentrations).

Just as with MTR, the TDI methodology has been adapted to produce a WFD compliant diatom assessment tool called River DARLEQ (Kelly *et al.* 2008). Like LEAFPACS, the observed TDI score is divided by a reference score, predicted from a statistical model incorporating alkalinity and season as predictor variables, to yield an EQR ranging from zero (bad status) to one (high status).

2.4.3.2 Measuring impacts of elevated P in lakes

Unlike rivers, there was no standardised method for assessing nutrient enrichment in lakes prior to the WFD. Recent tool development has produced Lakes LEAFPACS for macrophytes and DARLEQ for diatoms.

Lakes LEAFPACS has five metrics:

- Lake Macrophyte Nutrient Index (LMNI);
- number of functional groups of macrophyte taxa (NFG);
- number of macrophyte taxa (NTAXA);
- mean percent cover of hydrophytes (COV);
- relative percent cover of filamentous algae (ALG).

Each metric is scored using data collected from four sectors, each comprising 100 m of shoreline and extended to the centre of the lake. Reference conditions are predicted using statistical models incorporating alkalinity, mean depth, altitude, surface area, conductivity, and geology as predictors.

The DARLEQ methodology assesses lake ecological quality by using diatom indicators to detect impacts from nutrient enrichment. The parameter used is the Lake Trophic Diatom Index. This parameter is indicative of the impact of nutrient enrichment on the quality element. It is calculated using information on benthic diatom species and groups of such species (UKTAG 2008c).

Samples of benthic diatom species should be collected by brushing or scraping the upper surface of cobbles or small boulders obtained from the littoral zones of lakes in order to remove the biofilm. The number of taxon present and the abundance of each are used to calculate an EQR. When the method is used to classify the ecological status or potential of a water body, the annual mean value of the EQR for the parameter should be used.

2.4.4 Response of receptors to reduced P

Eutrophication is caused by elevated concentrations of nutrients and strategies to reverse eutrophication focus on reducing the loading of nutrients to the system. In highly eutrophic waters, typical of many lowland rivers in England, factors other than phosphorus may limit macrophyte and algal growth, which means that reducing phosphorus concentrations will not necessarily reduce the growth of epiphytic algae, which is thought to be a pre-requisite for the recovery of macrophyte communities in rivers (Hilton *et al.* 2006). Hilton *et al.* (2006, p79) emphasise this point well:

> "*… it is highly likely that the initial stages of P removal will not show any observable effects, because the P concentrations are so high that P is not limiting. Continued reductions in P concentrations will eventually bring them into the range where they begin to be limiting. At this stage, further reductions in phosphorus load would reduce the epiphytic biofilm on the plants … and even more P must be removed before a positive effect on the macrophytes will be observable. As a result, it is likely that in severely eutrophic rivers, the removal of phosphorus from a large number of sources may be required before the macrophytes are seen to begin to recover*."

Contrary to this prediction, Kinniburgh and Barnett (2010) reported that phosphorus removal at 36 STW on the River Thames reduced concentrations of orthophosphate from 0.5-2.0 mg l^{-1} to 0.2-0.4 mg l^{-1}, and that chlorophyll-a concentrations decreased over the same period despite orthophosphate exceeding the concentrations thought to be limiting for algal growth.

Determining the actual concentration at which phosphorus no longer limits algal or plant growth can be achieved by nutrient enrichment experiments, but only where phosphorus is still limiting. In highly eutrophic rivers where phosphorus is not limiting, it is often not clear to what level phosphorus needs to be reduced to elicit a biological response. Novel nutrient stripping experiments by Bowes *et al.* (2007) on the River Frome (a chalk stream in Dorset) indicated that SRP concentrations need to be reduced below 0.1 mg l^{-1} in order to limit epilithic algal growth and reduce algal biomass, which is lower than the 0.12 mg l^{-1} WFD standard for good status in lowland high alkalinity rivers. Bowes *et al.* (2007), Neal *et al.* (2010) and Bowes *et al*. (2010) note that the limiting concentration of SRP varies considerably among catchments.

Even if orthophosphate concentrations are reduced to natural background levels, this will not necessary trigger an immediate biological response. In lakes, high nutrient loading can cause a shift from a clear water state dominated by submerged higher plants to an alternative turbid state dominated by phytoplankton. This change may be maintained by a positive feedback loop whereby a combination of high turbidity, benthic algal mats and changes in sediment composition inhibit the return to a macrophyte-dominated state (Mainstone *et al.* 2000; Hilton *et al.* 2006).

Even in the absence of positive feedback loops, recovery may be delayed if a large reservoir of phosphorus has accumulated in river or lake sediments. Phosphorus can be released back into the water column, either by resuspension or desorption, buffering against reductions in dissolved P load achieved, for example, by P removal at STW (Hilton *et al.* 2006). This process may be particularly important in lowland rivers and shallow lakes which have silty bed sediments that trap phosphorus-rich particles and a large capacity for adsorbing soluble phosphorus from the water column (Mainstone *et al.* 2000). For example, Phillips *et al.* (2005) reported that phytoplankton biomass in a shallow lake took more than 15 years to respond to reduced phosphorus loading because of sustained phosphorus remobilisation from the sediment. Hilton *et al.* (2006) concluded that as a general rule, deep lakes, that is, those which stratify permanently throughout the summer season, typically respond rapidly to reductions in nutrient loads, whereas shallow lakes can take decades to recover. Jeppesen *et al.* (2005), reviewing recovery trajectories in 35 lakes following loading reduction, found that internal loading delayed recovery and that a new equilibrium between phosphorus in the sediment and water column was typically reached after 10-15 years, but no clear effect of lake depth on recovery was detected. In contrast to lakes, it is not known to what extent internal phosphorus loading may impede recovery in rivers (Hilton *et al.* 2006).

2.5 Response

2.5.1 Legislative and policy drivers

2.5.1.1 Water Framework Directive (WFD)

The EU Water Framework Directive (WFD) introduced in 2000 is now the main driver for actions to control emissions of P from agriculture, with a requirement to protect, maintain and improve the ecological quality of surface and groundwaters. The WFD provides a more co-ordinated approach to tackling eutrophication and is being implemented through river basin management planning that will develop a programme of measures in each River Basin District (RBD) to achieve the necessary improvements in water quality at catchment scale with due regard to costs. Additional historic and evolving national controls exist to further protect habitats and wildlife, which may include more specific management actions to reduce P where these are causing habitat degradation (Mainstone *et al.* 2008).

The WFD requires all inland and coastal waters to reach 'good' chemical and ecological status by 2015, indicating only a slight deviation from reference conditions. Some Directives will be fully integrated into the WFD, allowing them to be repealed or phased out by 2013, but the Urban Waste Water Treatment Directive (UWWTD) and Habitats Directive will remain in force and supplement the WFD. Areas designated under other EU Directives, such as Special Areas of Conservation (SAC) and Special Protection Areas (SPA), are designated as Protected Areas under the WFD.

Each RBD has drawn up a programme of measures which sets out the processes to understand, monitor and combat pollutants in order to meet environmental objectives. Both voluntary and mandatory mechanisms are used to implement the programme of measures, which are divided into 'basic' and 'supplementary' measures. Basic measures are those governed by legislation and consents, while supplementary measures include voluntary initiatives, codes of conduct, and projects.

Table 2.6 shows that just over 31% of rivers in England and Wales were found to be at risk of phosphorus enrichment from point sources, with an extra 4.6% probably at risk (WRc, 2009). Article 5 risk assessments undertaken for the second River Basin Characterisation estimated that 50.8% of (5,868) river water bodies (57.7% by length), and 29.6% (out of 432) lake water bodies (37.9% by area) were at risk from diffuse phosphorus inputs. Other diffuse sources such as septic tanks were not considered.

Table 2.6 River water bodies at risk from point sources of phosphorus in England and Wales

Risk category	Number of water bodies	Percentage (%)
At risk	2,434	31.3
Probably at risk	354	4.6
Probably not at risk	59	0.8
Not at risk	4,919	63.3

Under the WFD, a range of measures can be used to help reach the required phosphorus standards. These include point source controls, such as P stripping at STW, and bans on P in domestic laundry detergents, but may also include measures relating to diffuse source control, such as restrictions on manure spreading under the Nitrates Directive, which may have an indirect impact on P losses.

2.5.1.2 Birds and Habitats Directives (BHD)

The Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (EC Habitats Directive) aims to maintain biodiversity by requiring measures to preserve or restore natural habitats and wild species at ‘favourable conservation status’ and new measures to protect habitats and species of European importance. Some 189 habitats and 788 species have been deemed to be of European importance (WRc, 2009) and these are listed in Annexes I and II of the Directive. The Directive provides for the designation of Special Areas of Conservation (SAC) to protect qualifying habitat and species interest features drawn from the Annexes of the Directive.

Directive 2009/147/EC of 30 November 2009 on the conservation of wild birds (EC Birds Directive) establishes a comprehensive scheme of protection for all wild bird species naturally occurring in the EU. The directive recognises that habitat loss and degradation are the most serious threats to the conservation of wild birds. It therefore places great emphasis on the protection of habitats for endangered as well as migratory species (listed in Annex I), especially through the establishment of a coherent network of Special Protection Areas (SPA) comprising all the most suitable territories for these species.

Together, SAC and SPA designated under the BHD form an integral part of the Natura 2000 ecological network.

Any Natura 2000 site with water-dependent (ground- and/or surface water) Annex I habitat types or Annex II species under the Habitats Directive or with water-dependent bird species of Annex I of the Birds Directive, and where the presence of these species or habitats has been the reason for the designation of that protected area, has to be considered for the register of protected areas under the WFD (European Commission, 2010). These areas are summarised as "water-dependent Natura 2000 sites". For these Natura 2000 sites, the objectives of BHD and WFD apply. Within the context of the BHD, each Natura 2000 site has a set of defined conservation objectives summarised in the form of a Favourable Condition table that sets out the conditions for biotic and abiotic factors that will allow the site to be maintained or restored to favourable condition. For many species, the absence of eutrophication is key to attaining favourable condition and consequently a set of P standards for rivers and lakes within Natura 2000 sites has been derived (see Table 2.5). These standards are generally more stringent than those applied elsewhere in rivers and lakes and measures must ensure that these more stringent standards are met within Natura 2000 sites.

The Conservation of Habitats and Species Regulations 2010 (SI No. 2010/490) consolidate and replace the Conservation (Natural Habitats, &c.) Regulations 1994 in England and Wales. Under Regulation 61, it is a requirement that certain plans which are likely to have a significant effect on European sites or European offshore marine sites must be subject to an "Appropriate Assessment". The responsibility falls to the Environment Agency to implement these regulations and this will include compliance with the appropriate P standard.

2.5.1.3 Sites of Special Scientific Interest (SSSI)

There are over 4,000 Sites of Special Scientific Interest (SSSI) in England, covering around seven per cent of the country's land area (Natural England 2009) and more than 1,000 in Wales, covering around 12 per cent of the country (CCW 2009). SSSI are designated to protect some of the country's best wildlife and geological sites. They include a wide range of habitats and are home to numerous significant wildlife species.

More than 70 per cent of the UK's sites are internationally important for their wildlife, and are also designated as SAC, SPA or Ramsar Sites. Of the sites in England, three per cent are inland aquatic SSSI consisting of rivers, streams and lakes (OFWAT 2004).

Protection of SSSI is provided for under the Countryside and Rights of Way (CROW) Act 2000 where authorities (such as the Environment Agency) should further the conservation and enhancement of SSSI and consult Natural England (or CCW in Wales) before consenting to operations likely to damage a SSSI. This would include the requirement to ensure that the P standard would not be breached and could result in the introduction of P removal.

The UK government set a target to ensure 95 per cent of SSSI would be in 'favourable condition' by 2010. A Natural England review of SSSI in 2006 concluded that rivers and streams were the SSSI habitat in the worst condition, with only around a quarter reaching favourable status (Natural England 2006). Threats to aquatic SSSI included: groundwater and surface water abstraction; physical modifications which alter channel morphology and flow regimes; recreational activities, boating, swimming, bathing; introduction of invasive foreign plant and animal species; and pollution, especially eutrophication and acidification as a result of urban, industrial and agricultural development.

2.5.1.4 Integrated Pollution Prevention and Control (IPPC) Directive

The IPPC Directive (2008/1/EC codified from 96/61/EC) aims to minimise pollution from industrial sources throughout the EU. Industrial production accounts for a significant share of overall pollution in Europe, including greenhouse gases, acidifying substances, wastewater emissions and waste. The directive was introduced in 1996, and contains a set of rules for controlling industrial installations.

The directive covers around 52,000 installations, and requires the operators of these installations to obtain environmental permits. New installations and existing ones where changes have been made have been required to comply with the directive since October 1999, with other directives requiring compliance by October 2007.

Permits are based on an integrated approach taking into account the whole environmental performance. Permits include emission limits, based on best available techniques. However, permits have some flexibility, depending on the technical characteristics of the installation, its location, and the local environmental conditions. The directive also includes the right to public participation in the decision making process, with public access to permit applications, permits and emissions data.

Phosphorus is listed in a number of permits, which include those regulating chemical sites that produce organic and inorganic chemicals, as well as those producing P fertilisers. Farms may require an environmental permit (England and Wales) if they have more than 2,000 places for production pigs that weigh more than 30 kg; 750 places for sows; or 40,000 places for poultry, including ducks and turkeys.

2.5.1.5 Urban Waste Water Treatment Directive (UWWTD)

The Urban Waste Water Treatment Directive (91/271/EEC as amended by Directive 98/15/EC) aims to protect the freshwater, estuarine and coastal environment from adverse impacts due to domestic sewage, industrial waste water and surface water run-off. It sets requirements for the collection, treatment and discharge of urban waste water and establishes timetables for the achievement of standards according to the sensitivity of the waters.

In the Directive, eutrophication is defined as: "*the enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned.*"

Firstly, the Directive requires a water body to be designated as a Sensitive Area (SA) if it is eutrophic or may become eutrophic in the near future if no action is taken. A range of symptoms to judge whether a water body is affected by eutrophication has been drawn up and nationally agreed.

Waters are reviewed every four years in order to confirm that existing SA are still sensitive and to establish any new areas which should be designated. The first 33 SA were designated in 1994, with further additions in 1998, 2002 and 2006. The current total number of SA is 144.

Under the UWWTD, all STW with a population equivalent (PE) greater than 2,000 that discharge into a SA must have secondary treatment facilities to meet standards for biological oxygen demand (BOD), chemical oxygen demand (COD) and total suspended solids. STW which have a PE of more than 10,000 that discharge directly or indirectly to SA must have tertiary treatment to meet standards for phosphorus and/or nitrogen. For STW with a PE of between 10,000 and 100,000, the annual mean concentration of TP must be ≤ 2 mg l^{-1}, and for STW larger than 100,000 it must be below ≤1 mg l^{-1}. In addition, the annual mean TP concentration at all works with treatment must be reduced by at least 80 per cent. Nutrient stripping within tertiary treatment must be active within seven years of the SA designation.

Although biological water quality criteria are used to designate SA, UWWTD objectives are set only in effluent quality terms; no assessment is made of the level of load reduction required to achieve a given standard of biological water quality (WRc 2009).

2.5.1.6 Common Agricultural Policy

Strategic reform of the EU Common Agricultural Policy (CAP) now rewards farmers for protecting national land resources and environmental assets rather than maximising agricultural production. As part of CAP reform, a programme of 'cross-compliance' has been introduced that links Single Income Payment (SIP) to compliance with relevant legislation and the maintenance of a basic standard of environmental protection, termed the Good Agricultural and Environmental Condition (GAEC). GAEC relates to soil erosion, organic matter and structure, avoiding habitat deterioration and protection and management of water.This includes recommendations such as avoiding the use of fertilisers within two metres of a watercourse, and growth of green cover close to watercourses, which will help to prevent the runoff of agricultural nutrients like P into watercourses (Schmit and Rounsevell 2010).

2.5.2 Control measures for agricultural emissions

2.5.2.1 Overview

A number of measures can help reduce the transfer of P from agricultural land to water and recent reviews highlight the range of options available and their cost-effectiveness and potential for pollution swapping (Withers and Jarvis, 1998; Kronvang *et al.* 2005; Sims and Kleinman, 2005; Stephens and Quinton, 2010). In the UK, mitigation options are separated according to their principal mode of action in terms of whether they are targeted at reducing the inputs of P, the mobilization of P in soluble and particulate forms in runoff or the delivery of P to the watercourse (RPA, 2003; Haygarth *et al.* 2009). Further controls may be implemented in the water body itself as engineered solutions (for example, making water potable) or through more general manipulation of stream habitats (Beechie *et al*. 2008). Measures to reduce the inputs and mobilization of P are often collectively termed source controls, whilst those aimed at the delivery of P have been termed transport controls (Sharpley *et al.* 2000), but this distinction is not clear. Makarewicz *et al.* (2009) usefully differentiated between cultural measures (those taken by farmers on a day-to-day basis) and structural measures (those that require longer-term alterations to the landscape or farming infrastructure).

The ethos behind the WFD requires that measures to control diffuse pollution in catchments should be targeted at their source and a wide range of practical source-oriented P mitigation options have been itemised in a diffuse pollution inventory (DPI) user manual (Cuttle *et al.* 2006). This manual has been updated to include 83 measures (Newell-Price *et al*., 2011). In many cases these options have effects on more than one pollutant (such as sediment, nitrogen and pathogens), in a positive or negative direction (Stephens and Quinton, 2010). Not all measures are applicable to all farming systems and their implementation involves adapting nutrient (fertiliser and manure), soil, crop, livestock and landscape management on the farm; typical measures relevant to P are summarised in Table 2.7. Many of these measures have been included in agricultural stewardship schemes. In general, more measures are applicable to arable cropping systems than to livestock systems. For example, using a simple P loss accounting model based on export coefficients, Haygarth *et al.* (2009) found that P mitigation measures collectively were able to reduce P loss by up to 2.2 kg ha^{-1} on a model arable farm, 0.9 kg ha^{-1} on a dairy farm, 0.6 kg ha^{-1} on a pig farm and up to 0.2 kg ha^{-1} on an upland farm. In practice the effects of individual and collective measures on farms is extremely variable and may not always lead to P loss reductions.

Estimating the effects of mitigation measures has been through direct experiment, usually at the field or small catchment scale, through modelling, or by a combination of approaches. Typically, the effects of individual measures have been investigated at the field scale and their practical implementation on farms has not always been successful due to variability in local management control, implementation over an insufficient area to have the desired effect or extreme hydrological variability (Sharpley *et al.* 2009; Ranganath *et al.* 2009). Spatial influences affecting the likely impact of mitigation options on river chemistry and/or ecology at catchment scale include in-field processing, in-river processing and the contributions of P from a number of sources other than agriculture (Edwards and Withers, 2007, 2008). Most studies have quantified effects on water chemistry rather than ecology. Moss (2008) calls for a more holistic approach to habitat management that requires adoption of a combination of methods to overcome the complex and poorly understood nature of catchment processing of nutrients. The work of Makarewicz and colleagues (Makarewicz *et al.* 2009) is a major advance in this area and similar to the philosophy behind the Defra Demonstration Test Catchments in England and Wales. Sharpley *et al.* (2009) suggest that if mitigation options are to be successful they must be (a) iterative, (b) involve stakeholder-driven programmes and (c) address the inherent complexity of all P sources within catchments.

Individual mitigation methods are fully described in the User Manual (Cuttle *et al.* 2006) and are only briefly summarised below in terms of their rationale, effectiveness, applicability, practicalities and limitations.

2.5.2.2 Reducing inputs of P (nutrient management)

Rationale, effectiveness and applicability

As described in Section 2.2, the amount of surplus P present on farms governs the potential for P transfer to water. Farming practices which increase surplus P accumulation in soils include underestimating the nutrient value of P in organic manures, uneven distribution of manures around the farm area and not matching total P inputs to crop and livestock demand. Adoption of N-based fertilizer recommendations can automatically lead to surplus P accumulation in soil because of the low N:P ratio in some manures such as biosolids. Guidelines and tools are available to better judge the value of manure fertilizer and allow farmers (and their advisers) to calculate their farm annual P surplus to avoid unnecessary wastage; for example, the agricultural industry's Tried and Tested Nutrient Management Plan or the government's software package PLANET and supporting recommendations (Defra, 2010b). The store of P in the soil can also be easily quantified by regular soil analysis to maintain Olsen-P index 2 or 3 depending on crop type. Maintaining soil P at these target indices will help to minimize soluble P transfer from soils.

Cultural approaches to reducing P surplus are based on targeting inputs of P in fertilisers, feed and any imported wastes more accurately to the needs of crops and livestock, and invariably lead to considerable cost savings. On all farms, fertilizer applications can be withheld, or considerably reduced, to allow soil P fertility to decrease where it is in excess. One way to achieve this is by making full allowance of the P nutrient content of livestock manures and biosolids when deciding on crop fertilizer requirements, or through adoption of P-based fertilizer planning. Sharpley *et al.* (2009) reports a reduction of 60 per cent in soluble and total P loss by using a P-based rather than an N-based fertilizer management strategy on one USA farm. On livestock farms, the P content of excreta/manure can be reduced by altering dietary P intake; for example by reducing the P content of the diet, more accurate prediction of P availability in feeds, feed additives such as phytase (that can increase the digestibility of P feed), increased use of phase feeding, and novel feed ingredients (such as high available P grains) that decrease the need for supplementation. Maguire *et al.* (2007) suggest reductions of 25-50 per cent in manure P contents can be achieved; Dou *et al.* (2002) achieved a reduction of 60 per cent by halving the P content of diets for dairy cows. Precision farming further extends the principle of matching inputs to need by targeting individual field areas, or groups of livestock, according to soil nutrient status, crop yield potential or livestock output, but requires investment in time, soil analysis and equipment. More strategic but high cost structural approaches to reducing farm P surpluses include reducing animal numbers and recovery of P from animal wastes as struvite. Makarewicz *et al.* (2009) found that removing dairy animals achieved by far the biggest reduction in P export in a study of catchment-based approaches to P mitigation. Technologies for struvite recovery from livestock manures are in their infancy and struvite (ammonium magnesium phosphate) production is at present only an option for very large livestock enterprises.

Shepard (2005) showed that nutrient planning reduces P inputs on farms, and is an essential component of P mitigation at farm and catchment scale (Sims and Kleinman, 2005). Van der Salm *et al.* (2009) found that withholding P fertilizer reduced the easily available P pools in soil (such as water-extractable P) more quickly than other forms of soil P. In the UK, the recent Countryside Survey soil report (Emmett *et al.* 2010) and recent analysis of more historic data from the Representative Soil Sampling Scheme (Oliver *et al.* 2006) suggest a significant reduction in soil Olsen-P in managed grassland and arable land in recent years. The authors argue this is due to the general decline in P fertilizer usage and better nutrient management on farms. In the Netherlands, despite increasingly tight restrictions on P applications in fertilizers and manures since 1984, these effects are not yet reflected in soil P status (Reijneveld *et al.* 2010). Watson *et al.* (2007) omitted P fertilizer on grazed plots and found only small reductions in soil P over a five-year period. Using a model based on soil P decline in eight field experiments, Schulte *et al.* (2010) predicted that it would take on average seven to 15 years for soil test P to decline from high to minimum agronomic requirements, but the uncertainty in these estimates suggested the variation could be from three to over20 years depending on soil type and the amount of surplus reduction. Data from Withers *et al.* (2009b) suggest that losses of particulate P will persist long after any reductions in soluble P have been achieved through reductions in STP. Using a simple modelling approach, Haygarth *et al.* (2009) calculated that reducing high soil P to agronomically required levels would reduce P losses by only 0.2 kg ha^{-1}. Similar data are reported by Monaghan *et al.* (2007) who found P loss reductions of 0.1 kg ha^{-1} (<10% of total P loss) on dairy farms. In Sweden, Ulén *et al.* (2007) showed that nutrient management in agricultural catchments over 21 years resulted in only small changes in river P concentrations.

Practicalities and limitations

Farmers have historically applied P in excess to build up soil fertility, insure against poor crop or livestock performance and because of uncertainties over P release rates from manures and unevenness in spreading or incorporation. If inputs of fertilizers and feeds are reduced too much, crop yields and animal health will suffer; confidence in reducing P inputs clearly needs to be demonstrated. In Ireland, reductions in fertilizer P use on livestock farms have been compensated by an increase in feed P inputs (Maguire *et al.* 2009). However, manures have been shown to be an effective alternative to fertilizers, especially if analysed regularly to assess their nutrient value. Nutrient management therefore requires revised attitudes, greater attention to detail and additional management skills, for example in using farm planning tools such as PLANET (Withers *et al.* 2003).

Reductions in surplus P inputs of fertilizer and feed will not automatically lead to lower P emissions from agricultural systems, especially in the short term because of the large stores of P already present in soils and rivers (for four national examples, see Maguire *et al.* 2009). The large reserves of total P in the landscape will decline only slowly. Standard soil test methods are not a reliable indicator of soil P loss to water, but are nevertheless a very useful tool for guiding fertilizer and manure P inputs for optimal crop production. Farm P surpluses have declined in recent years due to natural market forces. Low farm commodity prices and spiralling costs of P fertilisers have forced many farmers to take P fertilizer holidays. Future shortages of rock phosphate may limit supplies of P fertilizer and increase costs further. Use of phytase has become more widespread and feed companies have already taken steps to reduce the P content of ruminant diets. The scope for further decreases in feed P inputs is probably limited because of concerns over animal health.

2.5.2.3 Minimizing P mobilization (soil, crop and livestock management)

Methods to minimize the mobilization of P in runoff include reducing vulnerability to soil erosion and preventing the enrichment of runoff due to P release from sources present on the land surface (such as recently-applied fertilizers and manures).

Reducing vulnerability to soil erosion

Rationale, effectiveness and applicability

The detachment of soil particles during rainsplash and subsequent runoff generation, via surface runoff or drainflow, is often the major contributor to P export from agricultural land; options to decrease the risk of detachment involve combinations of soil, crop and livestock management. Particle entrainment in runoff is ubiquitous; it occurs in the absence of typical signs of erosion (rills and gullies) and can be just as much of a problem on poorly managed grassland as on arable land. Options to mitigate against soil particle entrainment are largely cultural and include those that help to protect the soil (by providing vegetative cover, restricting livestock access and so on), those that improve the structural stability of the soil (. reduced cultivations, incorporation of organic matter and so on) and/or those that hinder runoff generation (such as contour cultivation, alleviation of soil compaction, tramline management, in-field buffer strips).

On arable farms, soil and crop management have a large impact on P export. Establishment of cover crops during periods when the soil would otherwise be bare (for example, during autumn and winter in the UK after maize or before a spring crop), and incorporation of crop residues, to help protect the soil surface have been shown to be effective in reducing particulate P transfer, but transport of dissolved P may increase due to release of P from the crop material (Bechmann *et al.* 2005; Ulén *et al.* 2010). Sharpley *et al.* (2009) report P loss reductions in surface runoff of up to 63 per cent and Stephens and Quinton (2010) report up to 95 per cent reduction in total P by establishing cover crops. Torbert *et al.* (1999) found that incorporation of residues increased the time before runoff was generated. Cover cropping is widely and effectively practised in Scandinavian countries at modest cost.

Minimal cultivation (non-inversion tillage) has been shown to reduce sediment and P transport by up to 80 per cent in a number of studies by providing crop residue cover, improving soil structural integrity and reducing runoff volumes (Carter 1998; Withers *et al.* 2007; Zhang *et al.* 2007; Ulén *et al.* 2010). However, a number of studies have shown that P losses can be increased by lack of soil inversion due to structural deterioration and accumulation of applied and crop residue P at the surface (Carter, 1998; Stephens and Quinton, 2010). Sharpley *et al.* (2009) report overall positive effects of 35-70 per cent, but a fundamental problem with soil conservation measures is that they boost the loss of soluble P. In the UK, much research has focused on tramline management. Absence of tramlines, or simply disrupting tramlines by tines after trafficking, has been shown to reduce P losses by 72-99 per cent at field scale (Deasy *et al.* 2009). Effectiveness at catchment scale will depend heavily on the extent to which tramline runoff actually enters the watercourse. There is less information on contour cultivation and in-field buffer strips which are not widely practised in the UK. Stephens *et al.* (2009) found only marginal benefit from contour cultivations but P loss reductions of up 30-75 per cent are summarised by Sharpley *et al.* (2009).

Restricting access to wet soils or vulnerable areas (river banks) by livestock is another measure to reduce soil erosion from grassland areas. Hotspots for livestock trampling of soil are around gateways, water troughs and farm buildings, whilst more general soil compaction can be caused through poaching when stocking rates are high and the soil conditions are not suitable. Little information exists in the UK on the effects on particulate P loss by improving livestock management in this way. Sharpley *et al.* (2009) report P loss savings of between zero and 78 per cent by improving livestock management (Table 2.7).

Practicalities and limitations

The effectiveness of soil and crop management techniques in reducing erosion heavily depends on site conditions (soil type, slope, climatic conditions and management). Adoption of some techniques has had no effect on P loss (such as minimal cultivations, Stephens *et al.* 2009) or has exacerbated the loss due to an increase in the mobilization of soluble P from crop residues and applied P at the surface (Sharpley *et al.* 2009). The literature suggests that while positive effects are highly likely, adoption of soil and cropping techniques should be judged according to the site conditions. Cover crops may lead to yield reductions in existing or subsequent crops (Aronsson *et al.* 2011).

Preventing runoff enrichment

Rationale, effectiveness and applicability

A significant source of P emissions from agriculture is the release of P, especially soluble P, from inorganic fertilisers, livestock excreta and organic manures that are surface-applied to agricultural land. Livestock excreta and manure residues around farm buildings, yards, housing and farm tracks are also a significant source of P to runoff. As discussed earlier, the amount of P released from these fresh applications is largely dependent on the water-solubility of P in the amendments applied (Kleinman *et al.* 2007; McDowell and Catto, 2005), and the risk of P transfer is greatly increased where amendments are applied to areas which are hydrologically active and directly connected to the water body during storm events (Withers *et al.* 2003; Monaghan and Smith, 2004). Mitigation options to reduce the enrichment of runoff with P therefore aim to reduce this risk in two fundamental ways: (a) minimize the risk of exposure of the source material to storm events by removing the source and/or adapting the distribution, timing and method of amendment application, and (b) minimizing the release of P when it rains by reducing the water solubility of the material applied.

A number of low cost methods can reduce the risk of exposure of the source material. Principal amongst these is the careful timing of fertilizer/manure applications to high risk areas (critical source areas) during hydrologically active periods. Experiments have shown that the highest release rates of P occur soon after application, so allowing sufficient time for fertilizer and manure P to equilibrate with the soil before it rains will reduce the risk exponentially. For example, McDowell and Catto (2005) found that whilst incidental P loss after fertilizer application in wet conditions represented 34 per cent of total P loss in one catchment, incidental P loss application under dry conditions represented less than 10 per cent of the total P loss, a saving of 0.2 kg P ha^{-1} yr^{-1}. Torbert *et al.* (1999) found similar effects. Reducing the area that is fertilized or manured at the same time may also help to spread the risk (Withers and Bailey, 2003) and reduce the overall increase in P concentrations at the farm scale. Similarly, the risk of runoff is much greater when soils are wet up to field capacity; avoiding P application to, and livestock grazing on, wet soils and during the critical winter (and in some cases autumn) periods will reduce the risk of P transfer. Monaghan and Smith (2004) found that substantial P losses in mole-drained pasture soils occurred when the volume of dairy effluent applied exceeded the soil moisture deficit in the top 45 cm of soil. Careful timing of slurry/dirty water applications to avoid wet soil conditions was estimated by Monaghan *et al.* (2007) to reduce P loss by 30 per cent on a New Zealand dairy farm. These authors suggest that lowering the irrigation rate as well as deferring dirty water/slurry application to periods when soils are drier could reduce P losses in sub-surface drains by 95 per cent.

Experiments have shown that the risk of exposure can be greatly reduced by removing the source material from the land surface by injection (grassland) or incorporation (arable) into the soil. Withers and Bailey (2003) found that incorporating manure immediately after application reduced P loss by 60 per cent compared to surface application at the field scale. Sharpley *et al.* (2009) reports that placement of P fertilizer instead of broadcasting might reduce P losses by between eight and 92 per cent. On livestock farms, limiting livestock access to streams effectively removes a potential source of direct excretal contamination and siting of manure heaps well away from watercourses and tile drains in fields will also reduce the risk of runoff enrichment when it rains. Sharpley *et al.* (2009) report that livestock exclusion from streams might reduce P losses by 32-76 per cent, although Raganath *et al.* (2009) found little effect on stream P biota because the stream length that was fenced was too short. Similarly, roofing of livestock yard areas lessens the interaction of excreta and rainfall in farmyards.

Reducing the water-solubility of P in manures can also minimize P release to runoff water; for example, McDowell *et al.* (2010) were able to reduce TP and dissolved P export from a small field catchment in New Zealand by 38 and 58 per cent respectively by using a low-water soluble P fertiliser (rock phosphate) instead of a highly-soluble P fertilizer (superphosphate). Solid manures generally have a lower water-soluble P content than dilute slurries and a switch from slurry to solid-based manure handling systems would be a more strategic but costly means of achieving this. Smith *et al.* (2001) found that P losses could be reduced by 75 per cent by using slurry of five per cent dry solids rather than slurry of 2.5 per cent dry solids and twice as much P in farm yard manure than slurry could be applied and still show less P loss than slurry.

Other studies have examined the use of various P-binding agents (natural, manmade or byproduct waste materials containing Fe, Al or Ca) to reduce water-extractable P in manures and slurries and also in P-rich soils. Dayton and Basta (2005) found that water treatment residuals (clean water sludges) containing Al could reduce soluble P release from soil to surface runoff by 60-96 per cent. Similarly, regular application of municipal biosolids and other waste materials which contain P binding elements (Fe/Al/Ca) have been shown to reduce the release of soil inorganic P to runoff water (Elliot *et al.* 2002; Leader *et al.* 2008).

Practicalities and limitations

Reducing the risk of runoff enrichment with P clearly requires that areas and pathways of rapid hydrological connectivity during storm events are identified on the farm so that these areas can be managed more sensitively in terms of P applications and livestock access. Key zones on the farm are riparian areas that maybe subject to rising water tables, heavy-textured soils with low infiltration rates that remain wet for long periods and valley floor features where surface runoff pathways congregate to form preferential flow pathways. Underdrainage systems also represent an important pathway for incidental P transfer and this needs to be recognised at the outset. If these 'high risk' areas are taken out of farming, this might have a large effect on overall farm productivity depending on the total area involved. If just P inputs to these areas are omitted, this may also reduce yields in the longer term. Restricting livestock access to these areas is generally achievable.

Adapting the timing of fertilizer and manure applications will require more attention to weather forecasting and more specialized injection equipment will be required to place slurries below the surface on livestock farms. Slurry treatment systems may have to be adapted to ensure the slurry is sufficiently dilute to prevent blockage during application. As with soil management, slurry/manure management must take account of site conditions; for example, in some studies P release and export in runoff has increased after injection. A switch from slurry-based systems to solid systems requires a major alteration of the farming system and an availability of straw or other bedding material.

2.5.2.4 Containment

Rationale, effectiveness and applicability

Controls over the delivery of P to the watercourse involve different types of containment that capture particulate and soluble P before it enters the watercourse via surface and sub-surface pathways. Many of these options require structural changes on the farm. Containment is applicable to all farmland but is particularly important to control P transfer from impervious surfaces (farmyards, tracks and roads) where flow rates are instantly generated, P concentrations are high and there is less opportunity to control mobilization (Withers *et al.* 2009a). As discussed in Section 2.2, tracks and roads often connect fields hydrologically to watercourses. Field underdrainage systems similarly have more limited options to retain P than surface runoff.

Methods for containment include constructed wetlands, riparian and in-field buffer strips, relocation of gateways, sediment traps and the deployment of P-binding agents to remove soluble P. Constructed wetlands and riparian buffer strips have been extensively studied, and there is much interest in the use of P-binding materials to capture soluble P in runoff. Constructed wetlands and buffer zones show highly variable effectiveness depending on their design, site conditions (hydraulic loading in particular) and the type of runoff they receive (Uussi-Kamppa et al., 2000). Positive effects (20-90 per cent P removal) of wetlands have been demonstrated in a number of studies (Babatunde *et al.* 2008; Koskiaho *et al.* 2003) but variability between seasons can be high and in some instances P exports have increased. For example, Tanner *et al.* (2005) found P export from a constructed wetland increased by 100 per cent in its first year. Dunne *et al.* (2005) found P removal rates of five per cent in winter and 84 per cent in summer. Failure of constructed wetlands is primarily due to a lack of sufficient residence time and the gradual saturation of the system such that it becomes a source of P (especially dissolved P) rather than a sink during storm events. Wetlands that integrate physical, chemical and biological removal mechanisms and/or use a larger retention area, or a series of retention areas, have been shown to be effective (Sholz *et al.* 2007; Foy *et al.* personal communication).

The usefulness of buffer strips was recently reviewed by Taylor, Lovell and Sullivan (2006) and Dorioz *et al.* (2006). As with constructed wetlands, many experiments have shown riparian buffers to be extremely effective (>90%) in reducing sediment and P loss in surface runoff (Kay *et al.* 2008; Sharpley *et al.* 2009; Stephens and Quinton, 2010), but these may fail due to inadequate width or design and may become sources of soluble P release as they become more saturated. As pointed out by Kay *et al.* (2008), the maximum delivery period of P (winter) overlaps with the least efficient period for buffer zone operation due to a combination of high water tables, reduced infiltration capacities and poor vegetative cover.

A number of phosphorus-binding materials have been used to remove P from water before it enters streams. Many of these materials have been deployed in constructed wetlands to treat wastewater. McDowell *et al.* (2008) report reductions in export of P in tile drains from P-rich pasture soils of 50-60 per cent when Fe-rich smelter/basic slag was used as backfill material, with a projected lifespan of 25 years. Effectiveness of these materials placed in ‘socks’ in streams was lower (30 per cent) due to lack of contact time. Ulén and Jakobsson (2005) found that adding lime to backfill soil in Sweden improved runoff infiltration rates. The use of these P-binding materials to reduce the mobilization of P in runoff has not been investigated in the UK.

Practicalities and limitations

As with other types of P mitigation methods, the effectiveness of containment options is highly variable depending on site conditions and measures need to be carefully designed. Clearly, riparian buffer strips are not effective at reducing P export in tile drains and their effectiveness in controlling P in surface runoff is highly dependent on how they are placed and managed (Dorioz *et al.* 2006). All containment options become saturated at some point if they are working and this needs to be recognized in the design and day-to-day management. The use of P-binding agents can be expensive and the longer-term effects of waste-product materials containing metals on soil quality have not been investigated.

Table 2.7 Effectiveness (expressed as percentage change or amount (kg ha^{-1})) and costs (cost per kg P saved) of mitigation options to control total P export in runoff from agricultural land according to recent reviews; a plus sign means an increase in P export

Mitigation option	Sharpley et al. (2009)	Kay et al. (2008)	Stephens and Quinton (2010)	Haygarth et al. (2009)	
	% change	% change	% change	kg ha^{-1}	£ kg^{-1} P saved
Source measures					
Matching P input to crop need (nutrient planning)	0-47			0.01-0.21	0
Reduce animal numbers					700-12,700
Mobilization measures					
Placement/incorporation of P	8-92	80-95		0.14-0.17	14-153
Timing/rate/type of fert/manures	0-45			0.08-0.16	16-415
Cover crops	7-63		40-65	1.12-1.18	58-61
Reduced cultivation (no till)	35-70		+100 - 100	1.10	45
Crop management + residues	25-88		1-40	1.11-1.12	18-222
Tramline management				1.11	8
Conversion to grass	75-95				129
Contour ploughing	30-75			1.11	9
Livestock exclusion	32-76			0.28-1.11	37-185
Managed grazing/compaction	0-78			0.62	40-75
Containment					
Buffer strips	0--93	+27- 98	40-80	1.62-2.17	4-5
Constructed wetlands	0-79	+6 - 72	10-60	1.53	93
Alter field size/gateways				0.61-0.81	60-405
Sediment traps for tracks				0.81	8

2.5.3 Control measures for non-agricultural emissions

Phosphorus in STW influent arises mainly from human waste and the use of phosphate-containing detergents for washing. P can be removed from effluent using chemical or biological treatment processes. In England and Wales, the majority of STW use chemical phosphorus removal as it can be retrofitted to existing plants, has a low capital cost, and is easy to operate. Only around 12 STW in England and Wales use biological phosphorus removal.

Chemical phosphorus removal

The chemical phosphorus removal process works by dosing effluent with suitable metal salts which precipitate phosphate as an insoluble metal phosphate/hydroxide complex. These solids are then removed by settlement in a settling tank. Using metal salts can provide an effluent of suitable quality to meet phosphorus discharge limits for almost any treatment plant.

A range of chemicals are used in this process, including liquid ferric, ferrous or aluminium salts as chlorides or sulphates. These can be added to the effluent before primary sedimentation, as part of the activated sludge process, or just before the final settlement tank. However, there are a number of issues with the process, including the high operating cost, increased amount of sludge produced, and problems due to changes in the composition of the water (WRc, 2009).

Biological phosphorus removal

Biological phosphorus removal works by using activated sludge aeration tanks, which are configured to select a specific type of bacteria, to store additional phosphorus as polyphosphate (two to eight per cent, compared to one to two per cent for normal bacteria). This means that the phosphorus removal rate increases from about 20 to 80 per cent. Several variants are available, but in all these processes the aeration tanks are configured with both an anaerobic and an aerobic zone to enable selection of phosphorus-accumulating organisms (PAOs) and enhance phosphorus removal.

In most cases, the waste sludge is then recycled to agricultural land, making this process more sustainable than chemical removal, while it also produces less sludge. However, biological removal is more expensive than chemical removal, the process is less stable and it relies on the wastewater having the correct composition (WRc, 2009).

Other technologies

Other methods are available for phosphorus removal, including waste stabilisation ponds and the use of beds of media that bind phosphorus (such as clay). These technologies are predominantly used in smaller STW which serve populations under 250, where chemical precipitation or biological phosphorus removal may not be appropriate.

One example of a technology used to reduce P in effluent is source reduction. For example, phosphate has been successfully replaced by zeolite in laundry detergents. In the UK, no legislation has been introduced banning the use of phosphates in such situations, and no voluntary agreements are in place.

P in detergents

In England, 78 per cent of the phosphorus load to the water environment comes from households (White and Hammond, 2009), with around 16 per cent of this coming from dishwasher and laundry detergents (Defra, 2009b).

A consultation was undertaken in 2008 in England and Wales on options for controlling phosphate in laundry cleaning products (Defra, 2008b). This was followed by an impact assessment in September 2009 (Defra, 2009b), and a ban is due to be implemented in 2015.

A number of countries, including Italy and Norway, have banned the use of phosphate in automatic dishwasher detergents. In the UK dishwashers are currently only found in a third of households, compared with over 60 per cent in Germany and 80 per cent in Sweden (WRc, 2010). As such, phosphates in dishwasher detergents could become a bigger problem as the market expands.

Non-phosphate detergents have been found to be as effective and cost competitive as phosphate-based detergents, while environmental consequences are not expected to be significant. If the UK banned phosphate-based detergents, around £2 million per year would be saved in operational costs from STW (WRc, 2010).

2.5.4 Incentives and obligations

Mechanisms to tackle phosphorus pollution fall into three broad categories: supportive, economic and regulatory. Mechanisms can also be classified according to whether they are voluntary or mandatory, and whether financial support, compensation or penalties are offered or applied. ADAS (2002b) has categorised mechanisms as follows:

- Voluntary, no financial pressure
- Voluntary, with compensation
- Voluntary, with financial pressure
- Mandatory, without compensation
- Mandatory, with compensation.

The first two of these categories can be regarded as supportive, the third involves the use of economic instruments, and the final two require regulation.

Financial support for farmers to adhere to the Good Agricultural and Environment Condition (GAEC) and the WFD is provided in the form of Environmental Stewardship (Entry Level and Higher Level Stewardship schemes and counterparts in the devolved governments) to help farmers improve soil, crop, livestock and nutrient management on farms. Advice and technical support is further provided by the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) that has established officers in 50 priority catchments covering more than one-third of England's farmed land. In the first two years of the programme (2006-08), 15 per cent of farmers within ECSFDI priority catchments received advice, covering a quarter of the total area (ECSFDI, 2008). At present, the emphasis is on voluntary measures. The UK currently has no statutory controls over P inputs, although these have been introduced in other countries to limit accumulation of surplus P in soils (Maguire *et al.* 2009). However, any statutory controls would be limited by the extent of available financing.

Business planning in water companies is done in five-year cycles in the form of Asset Management Plans (AMP). At the start of each cycle the water companies submit a business plan to Ofwat, which is used to determine what water companies can charge for the next five years. The results of this process tell the water companies how much money is available for investment and d measures to improve water quality. This money is then used to fund improvements such as chemical or biological P-removal technologies at WwTW.

Significant amounts of non-agricultural P come from unconsented point sources such as septic tanks and on-site wastewater treatment systems (OSWTS) for homes not connected to mains sewerage. Such systems can cause problems in areas where consented point sources have been reduced such as Lough Neagh in Northern Ireland (Lough Neagh Advisory Committee, 2008). The efficiency of these systems depends on the age and level of maintenance, and problems are difficult to track. The registration scheme is discussed in detail in Section 2.2.2.2.

2.5.5 Cost-effectiveness

The costs of adopting mitigation measures vary greatly depending on the type of method, its particular design, specific site conditions and the range in runoff P concentrations that need to be mitigated. The financial impacts on farmers may be short to medium term (labour/machinery) or involve major capital expenditure, or a combination of these. Cultural mitigation methods generally entail lower costs than structural methods. Fertilizer and feed P reductions involve no cost but do not have a large impact on P loss at farm scale. According to the cost curve approach of Haygarth *et al.* (2009), approaches involving crop and soil cultivation management achieve the largest reductions in P export on arable farms at relatively modest cost (less than £100 per kg of P saved). On livestock farms, livestock management and containment of P is more cost-effective than adjusting the timing of fertilizer/manure application because the latter requires slurry storage. Buffer strips and constructed wetlands are cost-effective but vary in their effectiveness (Table 2.7). The latter can also be costly and raises affordability issues for farmers.

Using Ofwat figures, the Environment Agency estimated that water companies have spent a total of around £950 million in capital expenditure in AMP2, 3 and 4 on installing phosphate and nitrate stripping in SA. The associated annual operational expenditure is approximately five per cent of this (Defra, 2008a). Ofwat generally allows these costs to be incorporated into prices charged by water companies to their customers.

Most investment has been directed at large STW (serving populations above 2,000) that come under the terms of the UWWTD. Pretty *et al.* (2003) predicted that P-removal at these STW would cost water companies £58-84 million per year in capital expenditure and £1.5 million per year in operating expenditure during the period 2000-2010. A significant investment was also made to comply with the requirements of the BHD. Pretty *et al.* (2003) estimated that for 44 of the 65 STW where P-removal was approved due to the potential impact of discharges on SSSI, the capital cost was £49 million, with an average annual operating cost of £60,000.

Little is known about the comparative cost-effectiveness of chemical and biological P removal and source reduction measures.

Biological P removal is complex and difficult to operate, and is not generally considered to be cost-effective at plants below 100,000 PE in size (WRc 2009).

Chemical P removal is simpler and can be used with any treatment process. Some methods, such as dosing into the humus tank, can cause formation of a precipitate which is generally carried over into the final effluent, making the process less effective and failure to meet standards more likely. Where biofilters are used, tertiary filters are often installed to meet consents, but this increases the costs of installation and adds to the energy cost.

Chemical P removal is relatively simple to operate and can be made to perform effectively on all processes. Chemical dosing in activated sludge plants (ASP) and oxidation ditches is normally into the aeration tank. In biofilters, it is often possible to meet the P consent by dosing into the primary tank but in some cases, especially where the consent is 1 mg l^{-1}, it is necessary to dose into the humus tank. This is effective, but the precipitate formed by the dosing is light and tends to be carried over into the final effluent. There is then a risk of failure to meet both the P consent and the iron consent. It is common to install tertiary filters to remove solids if iron salts are added to biofilter humus tanks. A tertiary filter is assumed to be required at the larger biofilter sites (10,000 PE and 100,000 PE) to meet the P consent. Where a tertiary filter is required, significant energy costs are associated with influent pumping and backwashing.

The performance of P-free detergents is reported to be comparable to those containing P under standard conditions, indicating that P-free detergents do not require extra energy or water to be effective. There is no evidence that P-free detergents cost more. At WwTW, use of P-free detergents could mean reductions in coagulant use and sludge production, but both are likely to depend upon proper monitoring and control. However, there may be a small increase in biological degradability load, and a consequent increase in energy demand, although this increase is likely to be small.

3 Tier 1 assessment: England and Wales

3.1 Source and exposure pressure

3.1.1 How much phosphate from different agricultural activities gets into surface waters? How does this compare to non-agricultural sources?

White and Hammond (2009) estimated that the total P load to surface waters in England and Wales is around 53 kilotonnes (kt) per annum. Source apportionment studies have been carried out to attribute the P load to individual sources using different approaches. These separate into: methods based on data from river flow and water quality monitoring programmes; and methods based on calculating P loads from inventories of point sources plus estimates of diffuse sources based on empirical equations or mechanistic models.

Studies of P in rivers in England and Wales have found phosphorus to originate from a wide range of sources, and have estimated the proportions contributed by different sources (Table 3.1). In England and Wales, 65-76 per cent of the annual TP load in waters derives from point sources, mostly sewage treatment works (STW). Agriculture is estimated to contribute between 18 and 28 per cent (Environment Agency, 2010a). Agricultural sources are thought to be particularly significant for lakes, protected habitats and headwaters because these are less likely to be located in areas affected by large amounts of point source pollution. However, other rural and urban sources including septic tanks and urban road runoff may add to riverine P loads, contributing up to 40 per cent locally (Environment Agency, 2010a). White and Hammond (2009) reported that for England and Wales, the amount of SRP apportioned to agriculture is 5.2 kt y^{-1} (13%), to households is 33.3 kt y^{-1} (81%), to industry is 0.5 kt y^{-1} (1%) and to background sources is 1.8 kt y^{-1} (5%). The Environment Agency and UK Water Industry Research (UKWIR) have jointly developed a source apportionment tool to generate estimates on a national scale. The model (SAGIS) uses up-to-date WwTW input data and outputs from the ADAS PSYCHIC model plus intermittent combined sewer overflow (CSO)/storm tank inputs, urban runoff and septic tank data to give a complete picture of sources of P. Model predictions will become available during 2012.

Table 3.1 Estimates of the proportion of P entering the aquatic environment from different sources in England and Wales (taken from Environment Agency, 2010a)

Source	Environment Agency 2007 National P compliance assessment using SIMCAT[1] (%)	ADAS 2007a (%)	White and Hammond 2009 (%)
Human, household and industry sources	70 (detergents = 18%, P dosing of drinking water = 11%)	73	76.3
Agriculture	27	18-24	20.1
Background	-	-	3.6

[1] This involves forward modelling of component P sources, coupled to a hydrological water balance model (Lowflows 2000) and simple in-stream decay of dissolved P using national SIMCAT model.

However, point and diffuse source estimates vary considerably between the regions and seasonally. While loads from point sources will maintain a relatively consistent output throughout the year, diffuse inputs vary due to land management practices, applications of fertiliser and manure, and variations in climatic conditions, particularly in rainfall. Loads from agriculture vary across catchments and river basin districts; the results from the RBDs are shown in Figure 3.1. These variations reflect differences in both population density and agricultural land use.

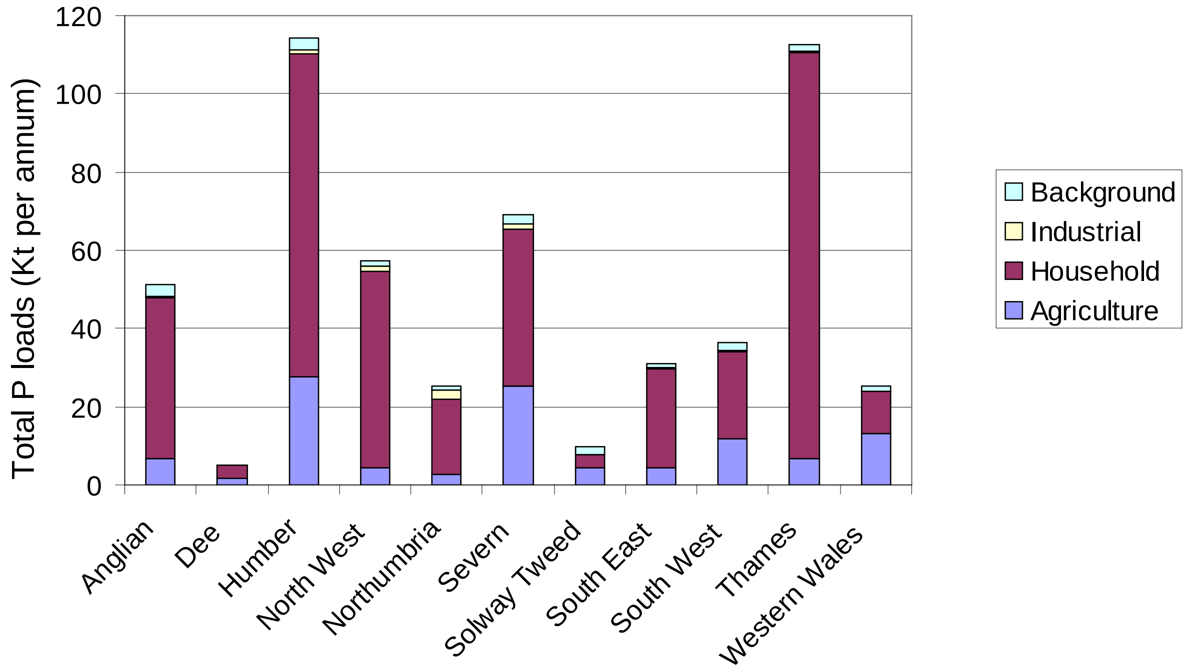


Figure 3.1 Regional contribution to total P loads (White and Hammond, 2009)

White and Hammond (2009) used modelling to look at the sources of riverine P in England and Wales. The study looked at a wide range of land use types, and calculated export coefficients based on land use, population and industry. Table 3.2 shows their source apportionment of total P and SRP loads by sector and by source, as a percentage of the total load. As shown, households and human inputs contribute the most significant proportion of both forms of P. Of agricultural activities, the highest loads come from improved grassland and livestock farming, particularly from cattle.

Table 3.2 Percentages of total P (TP) and soluble reactive P (SRP) loads in England & Wales (from White and Hammond 2009)

Source		TP load per year (%)		SRP load per year (%)	
Agriculture	Glasshouses	20.1	1.27	13.08	1.65
	Improved grassland		3.55		1.66
	Grassland moor/heath		0.16		0.11
	Arable cereals		1.97		1.20
	Horticulture		3.25		1.86
	Cattle		3.72		2.71
	Pigs		1.34		1.0
	Sheep		3.26		2.12
	Poultry		1.52		0.69
	Horses		0.12		0.08
Household	Humans	74.9	73.0	81.0	80.66
	Diffuse urban		1.89		0.37
	Septic tanks		-		-
Industry	Industrial point	1.36	0.88	1.27	1.15
	Industrial diffuse		0.48		0.12
Background	Bog/fen/marsh	3.62	-	4.64	-
	Woodland		0.09		0.05
	Atmospheric deposition		3.53		4.59

Under the WFD, the Environment Agency has collected large amounts of information from local area staff about the reasons why water bodies associated with diffuse sources of P are failing standards for P and diatoms. 43% of failures are attributed to point sources of P, and 37% are attributed to diffuse agricultural pollution (these proportions do not relate to the number of water bodies but to a count of the top 10 reasons for failure; there may be more than one reason for failure assigned to each water body) (Environment Agency, 2010a).

Anthony *et al.* (2009) estimated the losses of P from different livestock sources (excreta and manure management) compared with loads from fertiliser applications and soil supply as part of a modelling study looking at the cost-effectiveness of different mitigation measures. The biggest losses of P to rivers came from soil supplies (including short-term response to fertiliser application and residue decomposition), whilst amongst the livestock sources dairy and cattle and sheep farms contributed the largest amounts, as shown in Table 3.3. These figures were calculated before the implementation of any mitigation measures.

Table 3.3 Losses of P from different agricultural sources (Antony et al., 2009)

Pollution source	P loss (%)
Soil	51.6
Fertiliser	9.5
Beef	18.8
Dairy	14.8
Sheep	3.6
Pig	1.9
Poultry	0.2

Despite recent reductions in P loads from STW through improvements under PR04 and PR09, human sources, and in particular loads from the water industry, remain the most significant sources of P in England. Agriculture is thought to be the second most important source, followed by detergents and dosing of drinking water (phosphate is added to drinking water to reduce plumbosolvency), which are smaller but still significant sources. Other sources, such as contributions from combined sewer overflow (CSO) discharges, are more difficult to quantify due to their intermittent nature, while industrial sources and non-agricultural diffuse sources such as woodland are insignificant on a national scale (ICF International, 2007).

3.1.2 How, when and in what form is phosphorus mobilised and transferred from agricultural land to surface waters, and then within surface waters? What is the potential for re-release and what are the mechanisms and likely timing?

At present, understanding of how P travels from agricultural sources to water bodies is based on the source-mobilisation-delivery model, where source indicates the input to agricultural land, mobilisation is the process by which P molecules move through the soil and delivery is the pathway between the source and the water body.

Agricultural losses of P arise in three main ways (Wiederholt and Johnson, 2005):

- Soluble forms of P are lost rapidly when rainfall occurs soon after fertilisers or manure have been applied to the land, or when applications are made to waterlogged or frozen ground. These sources can contribute more than all the other processes if applications are not timed correctly.
- P is lost more slowly from soil-water, and the largest amounts are lost where P levels in the soil are high.
- Particulate forms of P are lost through soil erosion processes when P is bound to the soil. This source is most significant on arable lands, and is less important for grasslands.

Areas with higher levels of risk include those which are over-fertilised, recycle large amounts of manure, or are highly vulnerable to soil loss. These vulnerable areas may include farms which cultivate forage maize or potatoes, or those which undertake intensive dairy or pig farming. These sites are particularly high risk if they are in close proximity to a watercourse, on steep slopes or on underdrained land. The differences between these types of farms and others can have a significant impact on P inputs to watercourses (Environment Agency 2010a).

Neal *et al.* (2010) investigated the relationship between soluble and particulate phosphorus in nine major UK rivers, including 26 major tributaries and 68 monitoring sites across England and the border with Scotland and Wales. Each was monitored on a weekly to fortnightly basis for over a year between the early 1990s and 2008. Soluble reactive phosphorus (SRP) generally contributed the largest proportion of total P (average 63 per cent), with particulate (PP) and dissolved hydrolysable (DHP) forms contributing smaller amounts. However, SRP proportions varied with land use, with the highest levels occurring in urban areas and lower levels in agricultural and rural settings. Total P levels in these rural areas were also lower, and in some areas SRP made up less than half of the TP, with PP and DHP making up more than 30 per cent of the total.

White and Hammond (2009) found that grassland areas lose the largest proportion of their P in soluble forms, as the dense vegetation prevents particulate losses. Improvement of grassland areas may lead to an increase in P loads to watercourses, but the extent and duration of these losses depend on the method of improvement and fertiliser applied. Around 60 per cent of P losses from improved grassland are estimated to be in dissolved forms, although improved drainage systems can increase the proportion of particulate P. Fields which have artificial drainage systems have higher losses of both dissolved P as adsorption is reduced, and particulate P as flow pathways are improved through field drains. P losses from arable land are generally more likely to be dominated by particulate forms, but in some areas high proportions of dissolved P may be found, with SRP constituting up to 71 per cent of TP in well-drained clay-loam soils. Forests rarely lose P in particulate forms.

Sampling of storm runoff from a number of rural sources over a two-year period showed that P from impervious surfaces (farmyards and roads) or septic tanks was not only more concentrated than P from runoff from pasture and cultivated land, but also contained a higher proportion of dissolved forms of P (Withers *et al*., 2009a). P lost from impervious surfaces or septic tanks is thus likely to be more ecologically damaging due to its high concentrations and more bioavailable forms. These sources are also likely to generate releases continually throughout the year.

While significant amounts of P may be retained in stream systems as a result of biological uptake, P dissolved from particles may be remobilised in the particulate P load, or deoxygenation of waters and reducing conditions at the sediment-water interface may cause dissolution of P in sediments and release of SRP into the waters. However, the speed of cycling from dissolved P to particulate P and back to dissolved P may vary significantly, from minutes to years, or P may never be released from particulate forms.

The processes controlling P cycling and transformation are poorly understood, and further work including direct measurements of P is needed to improve understanding. More information on P cycling in freshwaters, and re-release is given in Section 2.3.1. For more information on how receptors respond to reduced levels of P, see Section 2.4.4.

3.1.3 What is the contribution of agriculture to P concentrations in groundwater and hence to groundwater-dependent ecosystems and in-stream loads/concentrations?

Holman *et al.* (2010) investigated the concentrations of P in groundwater across the UK and Ireland. Over 48,000 measurements were included in the study, but more than 14 per cent of the groundwater bodies in England and Wales had no observations of P. Groundwaters in

England and Wales showed an apparent increase in P concentrations up until 1987, followed by a progressive decrease, although there were fewer samples in the first half of the data.

Median groundwater P concentrations were found to be generally higher in England and Wales than in Scotland and the Republic of Ireland, but large areas of the country still had concentrations below 0.03 mg l^{-1}. However, a large number of groundwater bodies had concentrations above 0.05 mg l^{-1}, in particular in South-East and North-West England. This distribution is shown in Figure 3.2.

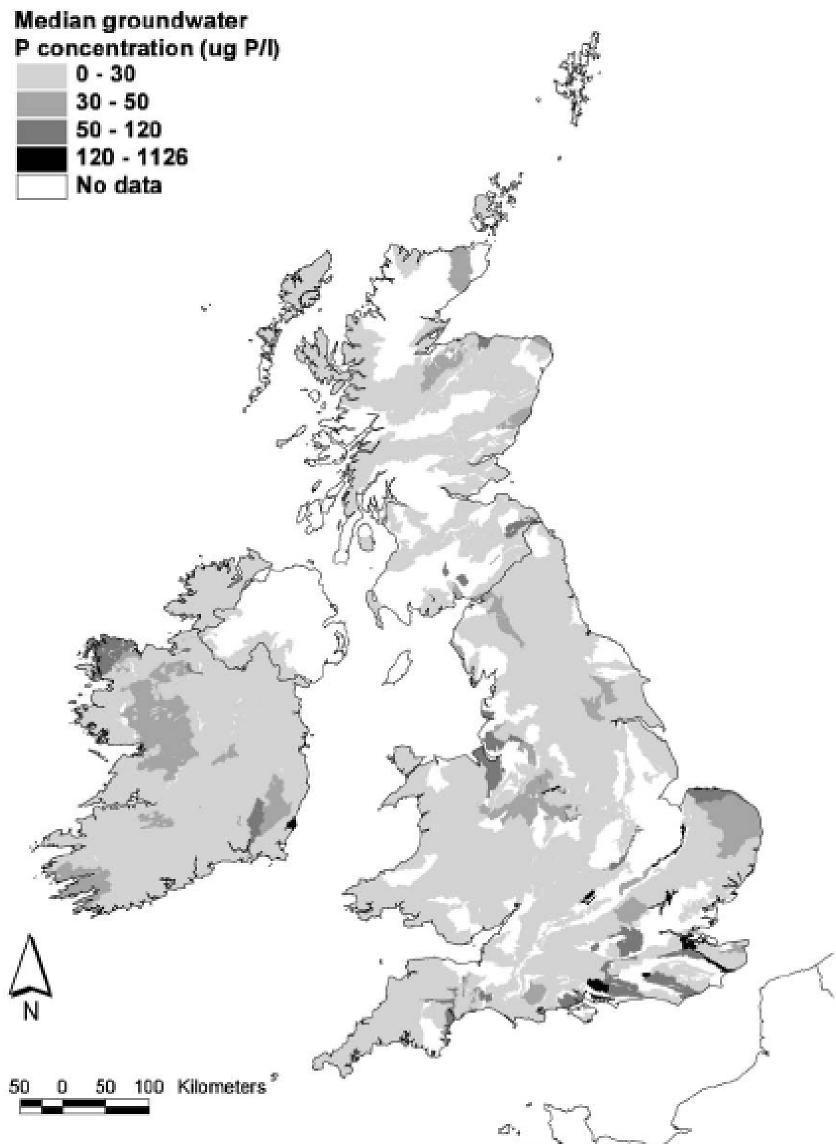


Figure 3.2 Median groundwater P concentrations in the UK and Ireland (Holman *et al.* 2010)

Compared to the three good ecological status river thresholds (UKTAG, 2008a), around nine, five and one per cent of groundwater bodies respectively exceeded the 0.04, 0.05 and 0.12 mg P l^{-1} standards (Holman *et al.* 2010).

The contribution of baseflow from groundwater to river flow varies across the country (Figure 3.3). In areas with a high Base Flow Index (BFI), high groundwater P levels will also lead to high P in rivers and groundwater-dependant terrestrial ecosystems (GWDTE). At present, little is known about the magnitude and extent of P problems in GWDTE.

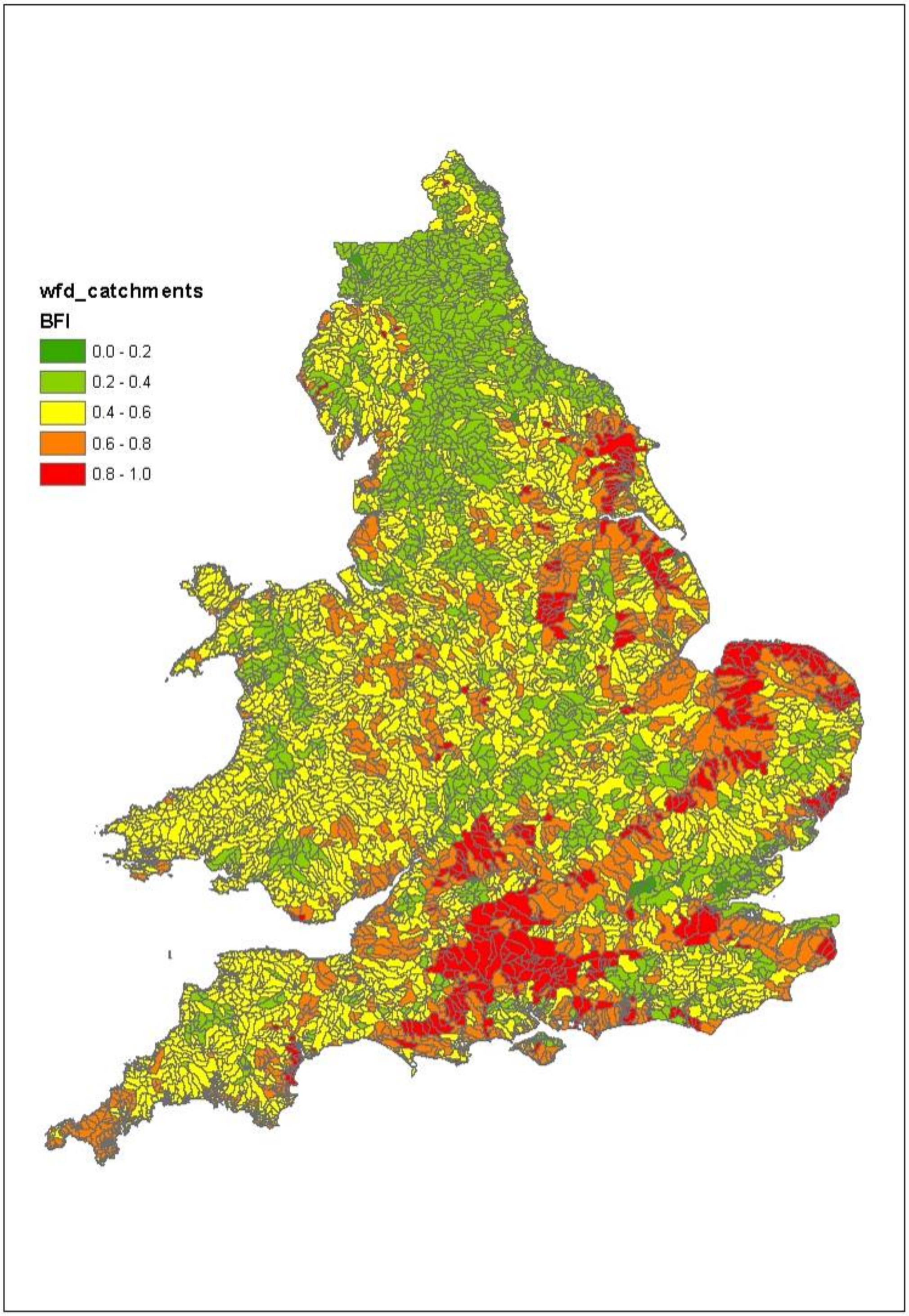


Figure 3.3 Base Flow Index for water bodies in England and Wales (supplied by Environment Agency)

Shand *et al.* (2007) examined baseline groundwater quality in the major aquifers of the UK, looking at a wide range of determinands including total dissolved phosphorus. This study identified the main source of groundwater P as apatite (phosphate minerals including hydroxylapatite, fluorapatite, chlorapatite and bromapatite), but anthropogenic sources such as fertilisers, farmyard slurry and water treatment works also have an impact.

In chalk aquifers, levels of P vary over one to two orders of magnitude, and despite the fact that P mobility is generally limited in carbonate aquifers, as it is easily sorbed to calcite under alkaline conditions, high concentrations can remain. Sources and transport of P in the chalk aquifer are poorly understood, but both natural and anthropogenic sources are likely to have an impact.

In Permo-Triassic sandstone aquifers, the range of P concentrations is similar to that in chalk. High concentrations have been found in aquifers overlain by thick drift deposits, so some of the P is likely to be from natural sources.

Understanding of the behaviour of P and the range of natural and anthropogenic pollution sources is limited, making determination of a baseline for groundwater P concentrations very difficult. However, as high P concentrations have been found in old groundwaters under confined conditions, natural ranges are likely to be high under such conditions. Median concentrations, however, are likely to be enhanced in most aquifer systems.

Figure 3.2 shows the variation in P concentrations in different types of aquifers across the UK.

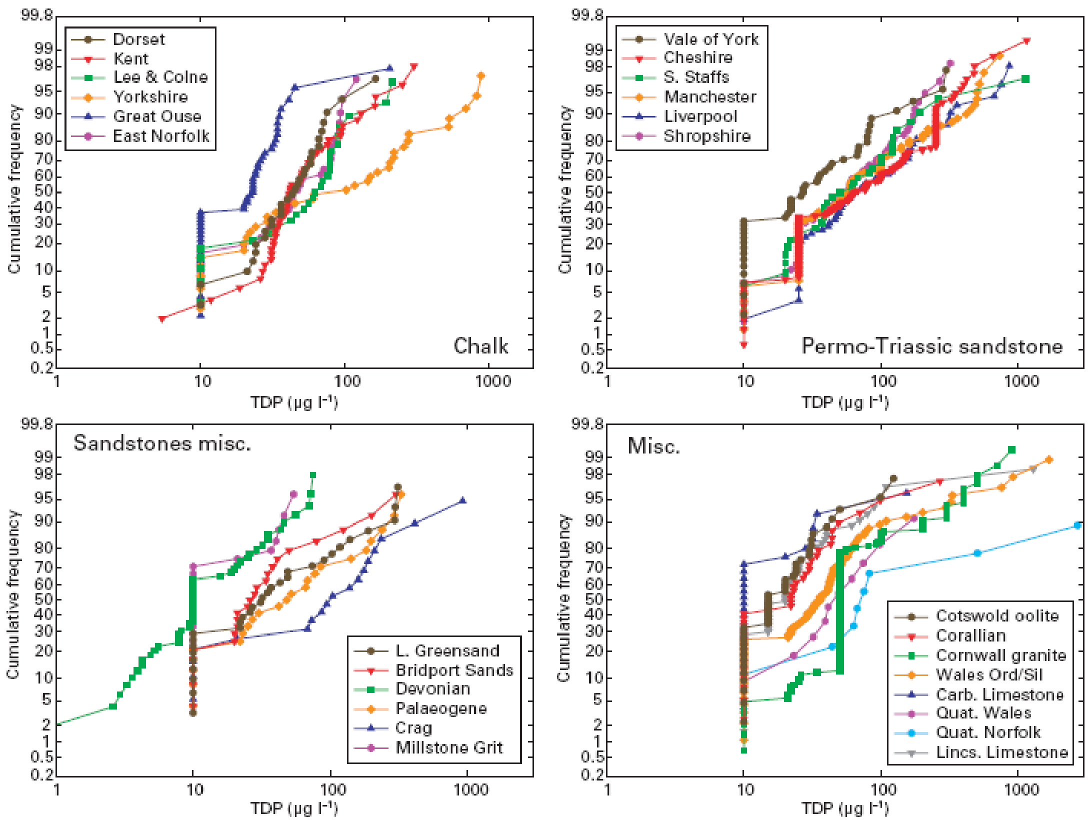


Figure 3.4 Distribution of P concentrations in UK aquifers (from Shand *et al.* 2007)

While it is possible to assess the concentrations of P in groundwater across the UK, the interaction between surface sources of P and the groundwater is less well understood. This makes it difficult to assess the contribution of different sources of P to groundwater concentrations, including agriculture.

3.2 Water quality and ecological impacts

3.2.1 Where does P from agricultural sources contribute to excess, ecologically significant levels of P, and what is the effective contribution? What impact on surface water body status (P and ecological) could reductions in the agricultural load have?

Eutrophication due to elevated phosphorus concentrations is a serious and widespread problem in rivers in England and Wales. In pristine rivers, concentrations of SRP are low – typically below 0.03 mg l^{-1}. The 2008 GQA results estimated that over a third of English rivers (half by length) and eight per cent of Welsh rivers (8.5% by length) had high (above 0.10 mg l^{-1}) concentrations of SRP; this compares with 69 and 26 by length, respectively, in 1990 (Figure 3.5). By comparison, nitrates are a less widespread problem, with a third of English rivers and only one per cent of Welsh rivers having mean concentrations greater than 30 mg l^{-1} in 2008 (Defra, 2009a).

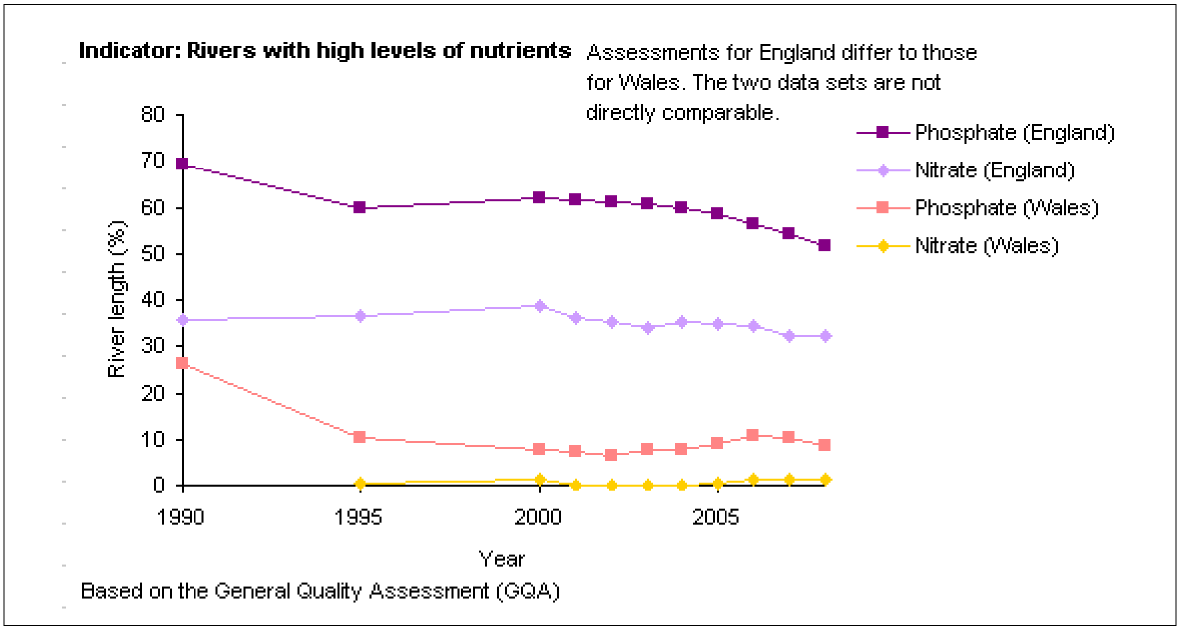


Figure 3.5 General Quality Assessment results for phosphate and nitrate 1990-2008

Higher levels of phosphorus are generally found in the lowlands of central, eastern and southern England, reflecting variations in geology, rainfall and population. Lowland arable rivers generally have higher nitrate and phosphorus levels than lowland pastoral rivers, which in turn have higher levels than upland rivers. This is primarily due to the low rainfall, and hence low dilution capacity, which characterises arable agricultural areas. The GQA is only based on dissolved inorganic phosphorus (SRP) and so the figures do not provide a profile of total phosphorus losses from agriculture, where a large part of the phosphorus will be in particulate, as opposed to soluble, form. The percentage of water bodies at less than ‘good’ status for England and Wales is shown, split into regions, in Table 3.4.

Table 3.4 Regional distribution of water bodies at less than ‘good’ status based on P standard (2008 assessment) (from Environment Agency, 2010a)

Region	Number of river water bodies: classified for WFD	Number of river water bodies: with P data	Number of river water bodies: at less than good status (50% confidence)	% of all WFD river water bodies at less than good status
Anglian	790	611	407	52
Midlands	732	674	423	58
North East	881	693	159	18
North West	603	504	160	27
South West	1061	707	276	26
Southern	407	299	159	39
Thames	366	313	199	54
EA Wales	**978**	**659**	**79**	**8**
English regions	**4840**	**3801**	**1783**	**37**
England & Wales	**5818**	**4460**	**1862**	**32**

Patterns of seasonal variation in phosphorus levels depend on the surrounding catchment. In most of the rivers investigated by Jarvie *et al*. (2006), the highest SRP conditions occurred during summer low flows, and concentrations decreased as flows increased. This was especially the case in areas with significant STW discharges, where inputs of phosphorus were relatively constant over time. In more agricultural areas, loads of particulate phosphorus increased with high flows during storm events, but these events caused less change in SRP concentrations.

Until the Environment Agency’s WFD monitoring programmes became operational in December 2006, there was no comprehensive monitoring scheme for lake water quality in England and Wales and few data are currently available to make a national assessment of lake trophic status. However, a review of research studies on 129 lakes in England and Wales found that 69 per cent of the lakes had total phosphorus concentrations greater than 0.10 mg l^{-1} P (Foy and Bailey-Watt 1998). In a study for English Nature of 102 SSSI suspected of being eutrophic, 84 per cent showed symptoms (Carvalho and Moss, 1995). These studies suggest that a majority of lakes in England and Wales are nutrient enriched to some extent. However, only 18 per cent of lakes assessed under the WFD failed to reach ‘good’ status for P standards (Environment Agency 2010a).

Groundwater may also be an issue, with eight per cent failing to reach ‘good’ status due to elevated levels of P (Environment Agency 2010b).

The relative contribution of point and diffuse sources of phosphorus to biological changes in rivers is an area of active scientific enquiry and debate. Jarvie *et al.* (2006) argued that point-source emissions of phosphorus from STW pose a greater risk of eutrophication than diffuse emissions from agriculture. Loads of soluble, and hence immediately bioavailable, phosphorus from STW enter the river continually throughout the year and are at minimum dilution during the spring and summer when plant growth is high and most sensitive to phosphorus concentration in the water column. In contrast, most of the annual load of phosphorus from agricultural land is attached to soil particles and not immediately bioavailable. Moreover, it is delivered to rivers and lakes in storm runoff and a significant proportion of the load will be immediately flushed out of the system by the high river flows. For these reasons, point sources provide the most significant risk of eutrophication in rivers, even in rural areas with high agricultural phosphorus losses (Mainstone and Parr, 2002; Jarvie *et al.* 2006), although the relative influence of point and diffuse sources will vary from one catchment to another.

Some particulate phosphorus may be deposited on the river bed and can enhance plant growth in subsequent growing seasons directly via root uptake or indirectly via desorption and diffusion into the water column. In slow-flowing rivers, accumulated sediment can provide an in-stream reservoir of phosphorus that may mask the effect of any changes in point source discharges. Furthermore, diffuse sources of dissolved phosphorus can be significant in some circumstances. For example, Foy and Lennox (2006) postulate that intensively managed grassland can release dissolved reactive phosphorus from the surface application of manures, accumulation of soil phosphorus at the surface and the creation of bypass flow pathways that encourages the loss of phosphorus from soil to water. Clearly, both diffuse and point sources of phosphorus can cause eutrophication but their relative contribution will vary within and between catchments.

Waters suffering from eutrophication are a significant and increasing problem:

- 295 (five per cent of) river water bodies are currently designated as “eutrophic” under the Urban Wastewater and Nitrates Directives, with a total river length of 4,100 km.
- Sixteen (two per cent of) lakes and eight WFD TraC waters are currently designated as “eutrophic” under the Urban Wastewater and Nitrates Directives.
- A further 44 river water bodies are being considered as candidates for designation as eutrophic in the current round of designations (Environment Agency 2010b).

Inputs from wastewater treatment works (WwTW) in the River Ribble catchment were modelled using the Environment Agency’s SIMCAT tool to look at the proportion of P from WwTW and scenarios for P removal. After calibration, the model estimated that WwTW discharges produced around 88 per cent of the P in the rivers, with agriculture contributing around nine per cent (Crabtree *et al.* 2009). The calibrated model was then run with scenarios to look at reductions in P from WwTW alone and in conjunction with measures to reduce diffuse P inputs. In total, over 80 different scenarios were simulated, examining various consenting and monitoring approaches and options for reducing diffuse pollution. One scenario looked at the benefits of investment at six WwTW under the UWWTD. This scenario predicted a load reduction of 761 kg per day (58%) from the WwTW, giving a reducing P release by half in the catchment. However, only 0.5 km more river would pass WFD standards. If Best Available Technologies (BAT) were applied at all WwTW, the effluent load could be reduced by a further 263 kg day^{-1}, which would result in an extra 3 km of river passing the WFD standard.

To fully comply with WFD standards, the daily load from all sources would need to be reduced to 82 kg per day, a reduction of around 90 per cent from current levels. As such, meeting WFD river quality standards for P is possibly the greatest technical and financial challenge facing the UK in meeting the requirements of the WFD. However, the issues discussed here are similar to those faced in many other large catchments, and this study has shown that catchment modelling can be used to ensure that catchments are satisfactorily protected, but at a reasonable cost (Crabtree *et al.* 2009).

With point sources dominating total P loads in many catchments, and dominating ambient river P concentrations in spring and summer in many more catchments, reductions in P loads from agriculture are likely to have minimal ecological impact. This is not the case in headwater catchments with only small human populations or sensitive SSSI catchments where a combined (point and diffuse) approach to P reduction is needed. Phosphorus data in the SIMCAT model are currently being updated, and a national source apportionment tool is being developed to work with the SIMCAT models as part of a UKWIR project. These developments should help with future investigations of P from agricultural and other non-point sources.

For more details on areas where agricultural sources of P cause excess, ecologically significant, P levels, and the contribution from agriculture, see Section 4.3.

3.3 Management responses

3.3.1 What voluntary measures can be used by those who manage land and water to reduce the level of phosphate reaching surface water bodies?

Under the WFD, measures must be implemented to ensure that water quality status does not deteriorate, and that 'good' ecological status is met. No actions are prescribed by the directive, only that they should be 'proportionate, technically feasible and cost-effective'. Under the UWWTD, specific measures must be applied to control nutrient losses.

As we know which sectors are responsible for the largest P loads, and in which regions these losses occur, it is possible to assign national, regional or sector-level targets. However, uncertainties over the effectiveness of measures to tackle biological conditions and the local contributions of individual activities (such as dairy farming) make identification of suitable measures difficult (Environment Agency, 2010a). Measures should also be targeted to areas where there is evidence of a biological impact as well as a simple failure to meet the P standard. Waters where the maximum benefit is expected are those with only slight or moderate impacts, where ecological improvement can realistically be expected.

Despite reductions in P loads from STW over the last 20 years, Defra's preliminary cost-effectiveness analysis (pCEA, ICF International, 2007) indicated that additional measures are needed to reduce loads from the two main sources – STW and agriculture. These further measures should be targeted within and across RBDs and should be accompanied by national measures for P from detergents and drinking water. Regulatory measures to reduce point sources of P from STW and slurry stores are known to be effective, but measures to tackle both agricultural and non-agricultural diffuse sources are less certain. Further work is now needed to assess the costs and benefits of such measures. No regulatory measures are in place to directly address diffuse P pollution in England and Wales.

Under the WFD, further regulatory measures are planned, including:

- Further improvements in P stripping at STW through PR09.
- A ban on P in domestic laundry detergents by 2015.
- Measures to tackle P loss within Safeguard Zones (SGZ). SGZ are an approach to tackling pollution in Drinking Water Protected Areas; where existing measures will not be sufficient to ensure that abstraction waters meet Article 7.3 of the WFD, a SGZ may be declared. The zone could be large or small depending on the problem and may not cover the whole of a catchment or sub-catchment.
- Measures under the Nitrates Directive may have a minor impact on P losses from controls on manure management.

Non-regulatory measures will also be implemented, including advice for farmers on improving nutrient planning and manure management through Catchment Sensitive Farming (CSF), implementation of sustainable drainage systems (SUDS), and local pollution campaigns to tackle misconnections and misuse of drainage systems.

However, modelling of these options shows only small reductions in P loads.

Voluntary measures can be split into three categories – source inputs, mobilisation and transport (Defra 2003).

Measures for reducing P inputs

1. Reducing fertiliser applications to arable, grassland and horticultural lands with high P soil indices. This measure is widely implemented as it reduces costs to farmers, and can be applied to significant proportions of agricultural land. Precision farming can also be used on all farms, including soil and yield mapping for fertiliser applications.

2. Livestock management measures including using lower P feed for dairy cattle, pigs and poultry and reducing stocking densities of dairy cattle and sheep. Alternative feed are generally accepted as any cost increase is insignificant, but costs to reduce stocking densities can be high, up to £400 ha^{-1} to reduce dairy cattle from 2.25 to 1.75 cows per hectare, due to losses in income. This type of measure could only be used for intensive farming units in highly vulnerable areas.

3. Changes in land use could have a significant impact if applied across five per cent of the arable area, targeted at marginal lands which have only been in arable production for a shorter time. Changing from arable production to beef and sheep could result in a significant loss of income of up to £300 ha^{-1} based on current wheat prices and rental averages. Alternatively, changing land use to willow coppice is seen as a zero cost option, due to the available grants.

Measures for reducing P mobilisation

1. Better management of P fertilisers through appropriate timing of applications (not during wet periods), use of slow-releasing fertilisers, and incorporation and drilling of fertilisers could reduce P losses by up to 90 per cent in some areas. The change in fertiliser type and timing of applications would have no additional cost, but incorporation and drilling could cost up to £6 ha^{-1}, resulting in costs of £31 million if carried out on all arable land.

2. Manure management could be improved in similar ways, largely by improving the timing of applications, and by increasing levels of incorporation and injection of manures, all of which would reduce surface wash-off and could result in reductions in P losses of up to 75 per cent. However, changing the timing of application could increase storage costs, and injection of manures would generally need to be carried out by contractors, increasing costs.

3. Restricting livestock access to marginal areas could be used in both the dairy and pig industries. While P losses could be reduced by up to half (50 per cent reduction of incidental P for cattle based on five per cent of the land area; 25 per cent reduction of detached P from outdoor pigs across a fifth of land area with pigs), additional costs would arise in the form of additional feed and fencing, which could reach up to £51 ha^{-1}.

4. Changes in cropping patterns could involve the use of cover crops to protect soil over the winter, and avoidance of late sowing in high risk areas. Cover cropping would incur significant costs (cultivation and seed), but could also produce additional income, while earlier drilling may need to involve the use of a contractor, increasing costs. Net costs for

a dairy farm could be £400, but over £3,000 for an arable farm (Newell-Price *et al*., 2011). However, P losses could be reduced by 20-80 per cent.

5. Better cultivation management could involve a range of measures, including reduced tillage, contour cultivation, use of soil stabilisers, mulching of waste crop materials, changes in tramline management and increasing surface roughness. Some measures such as contour cultivation have no additional cost but take longer than traditional cultivation, while others such as increasing surface roughness and use of soil stabilisers may have costs ranging from £83 to £240 ha^{-1}. Mulching of waste materials can generate additional income, as the mulch can be sold for use on vulnerable soils. These measures can reduce P losses by between 25 and 50 per cent.

Measures for reducing P transport

1. Construction of wetland/sedimentation ponds can entail high excavation costs, and will also reduce productive land area, but these methods can reduce P losses by up to 40 per cent by reducing sediment-attached forms of P.
2. Moving gateways away from points of drainage will reduce drainage pathways for runoff, reducing P losses by up to 30 per cent. Initial costs can be up to £200 per gate moved, but there is no further expenditure.
3. Reduction of field sizes and installation of hedges also reduce runoff pathways, reducing P losses by around a quarter. Costs are high, at up to £2,250 per field, but again, there is no additional expense.
4. Farm track sediment traps can be installed to intercept runoff before it reaches watercourses, reducing P losses by 30 per cent. Each trap will cost around £350 to install, and one trap would be needed for each 50 ha of farm.
5. By increasing riparian zones/grass buffer strips, P losses can be reduced by 60-80 per cent as runoff is stopped before it reaches watercourses. Costs for creation of these areas are relatively low, but on arable farms they can result in some loss of income.

Clearly, not all measures would be suitable for the total area of farmed land in England and Wales, so targeting of measures is necessary. For example, measures to reduce sediment in watercourses by restricting livestock access to rivers would only be suitable on livestock farms, while measures reducing fertiliser inputs would be best suited to arable areas. A measure may only be applicable to a proportion of farms in a local area; for example, converting manure management systems from slurry to solid waste would only be appropriate where a local supply of straw is available. The relevance of some measures may also be affected by the size of the farm (ADAS, 2007b).

Risk and Policy Analysts (2003) investigated the indicative costs of agricultural measures, noting that some measures are specific to certain soils, hydrology or farming systems. This makes definitive statements on what will be required in different locations difficult, and incorrect application of measures may exacerbate, rather than improve, problems, resulting in worsening water quality and unnecessary costs.

In particular, P loss is affected by soil structure and vulnerability to erosion (which increases levels of particulate forms of P). Where activities lead to formation of wheel ruts and cultivation marks, or degrade the soil structure, P losses can be increased. In areas of leaky soil types such as chalk or sand, mismanagement of manure applications or applications during wet conditions can cause particular problems. In these areas, appropriate measures should be applied to tackle these issues.

3.3.2 What will be achieved, in terms of annual and seasonal P load reductions, P concentrations in water bodies, and WFD ecological status, if these voluntary measures are adopted? How quickly may any benefits be realised?

Current and planned measures are not expected to be sufficient to achieve WFD objectives in a large proportion of water bodies by 2015. By 2015, RBMP aim to meet WFD objectives in 71 per cent of river water bodies (70 per cent of reach length, compared to 64 per cent now) based on £1.5 billion capital expenditure (Environment Agency, 2010a).

Phosphorus pollution has been reduced over the last 20 years by installation of P removal at STW under PR04 and PR09. Modelling of water quality has shown a decrease of around 30 per cent in P loads since 2000. Loads from point agricultural sources have also been reduced by improvements in storage of farm slurries and silage under the Silage, Slurry and Agricultural Fuel Oil (SSAFO) regulations, and pollution incidents from agriculture have fallen significantly since the 1990s (Environment Agency, 2010a). However, further measures are needed to reduce P loads from the water treatment and agricultural sectors if WFD targets are to be reached.

CSF measures are expected to reduce the agricultural P load by about five per cent in most catchments, although this will vary spatially, and may reach up to 20 per cent in some areas. Measures under the Nitrate Vulnerable Zones (NVZ) programme may also result in a reduction in P losses in some areas. However, there are high levels of uncertainty about the effectiveness of measures to tackle diffuse agricultural P, and their ecological impacts. Some measures may have wider reaching benefits than currently thought, as they reduce P loadings to downstream water bodies, but others may be less effective. However, measures may not be sufficient to tackle P issues in water bodies which are at ‘poor’ and ‘bad’ status (Environment Agency, 2010a).

A recent evaluation of Phase 2 of the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) using modelling indicates that improvements in management practices will result in significant reductions in pollutant losses (Environment Agency, 2011). Reductions in the first four years of the ECSFDI are generally predicted to be between five and 10 per cent across target areas, but could be up to 36 per cent. These reductions translate into decreases in river pollutant concentrations of similar magnitude. Responses vary for different pollutants and priority catchments, due to variation in advice and uptake and the significance of agricultural pollutant sources. Water quality monitoring has demonstrated reductions in pollutant loads and concentrations resulting from the ECSFDI. These reductions were up to around 30 per cent across targeted sub-catchments within representative catchments. In some cases, predicted phosphorus reductions from the ECSFDI will help meet WFD standards for good ecological status.

Anthony *et al.* (2009) used a computer-modelling framework to evaluate the cost and effectiveness of measures to control diffuse pollution from agriculture. The 69 measures were taken from the revised ‘User Manual’ of methods (Newell-Price *et al*. 2011) and were mostly improvements in practice within existing systems rather than new technologies. The impacts of each type of measure were estimated from literature and represented as a percentage reduction for each source type, area and delivery pathway. The methods used included the ‘use of improved livestock genetic resource’, ‘use of plants with improved nitrogen efficiency’, ‘sowing clover to grass swards’ and the ‘integration of fertiliser and manure nutrient value’ in fertiliser planning. However, the models assumed that mitigation measures were implemented across all of the land area, and did not take into account less common measures.

Table 3.5 shows modelled estimates of P loss from different types of farm before and after the adoption of mitigation measures based on 2004 data. The most significant reductions are seen on dairy and mixed farms, falling by 62 and 56 per cent respectively. The total reduction in P in England and Wales, following implementation of measures across all of the land area, is estimated at 53 per cent.

Table 3.5 Reductions in P loss to rivers after implementation of mitigation measures in England and Wales (Anthony *et al.* 2009)

	Before measures	After measures	Change in load
	Load in kilotonnes		%
Dairy	1.1	0.4	62
Horticulture	0.0	0.0	46
Lowland cattle and sheep	0.6	0.3	53
Mixed	0.6	0.2	56
Mixed combinable and pig	0.4	0.2	53
Outdoor pigs	0.0	0.0	38
Roots and poultry	0.3	0.1	52
Upland cattle and sheep	0.9	0.5	43
Winter combinable and pig	0.5	0.2	51
Total	4.4	2.0	53

Anthony *et al.* (2009) also looked at the reductions from different types of mitigation measures across England and Wales. Table 3.6 shows the calculated reduction from selected methods relative to the baseline prior to implementation of the measure. As shown, the methods which have the largest impact on P losses are avoidance of slurry spreading in high risk periods, integration of fertiliser and manure nutrient supplies and use of fertiliser recommendation systems.

Table 3.6 Calculated percentage reduction in national P losses from implementation of selected mitigation methods

Method	Per cent reduction
Avoid slurry spreading in high risk periods	4.0
Integrate fertiliser and manure nutrient supply	2.9
Use fertiliser recommendation system	2.9
Reduce tillage	2.1
Avoid spreading fertiliser in high risk periods	0.9
Use improved genetic resources in livestock	0.7
All methods used simultaneously	12.9

Anthony *et al.* (2009) assessed the cost-effectiveness of the measures included in the models. Table 3.7 shows the 10 measures with the highest levels of P reduction, and the cost per kg of P abated. The highest reductions in P loss were found when soil compaction was reduced, and fertiliser management systems were improved.

Table 3.7 Measures with highest reduction in P levels (from Anthony *et al.* 2009)

Type	Measure	Cost £million	Reduction in P (%)	£ kg^{-1} saved
Soil	Loosen compacted soil layers in grasslands	48	12.2	78
Soil	Cultivate compacted tillage soils	18	8.8	40
Fertiliser	Use a fertiliser recommendation system	-69	6.0	-229
Fertiliser	Do not apply P fertilisers to high index soils	-46	6.0	-154
Fertiliser	Integrate fertiliser & manure nutrient supply	-568	5.9	-1,910
Manure spreading	Do not spread slurry at high-risk times	5	4.0	23
Manure spreading	Use slurry injection techniques	83	3.7	441
Connectivity	Establish and maintain artificial wetlands - field runoff	7	3.6	36
Connectivity	Establish and maintain artificial wetlands - steading runoff	52	3.2	321
Connectivity	Fence off rivers and streams from livestock	47	3.1	298

Table 3.8 shows the 10 measures with the lowest cost per kg of P abated. The most cost-effective measures were found to be improvements in livestock genetics, and improvements in fertiliser management systems.

Table 3.8 Measures with lowest cost per kg P abated (from Anthony *et al.* 2009)

Type	Measure	Cost £million	Reduction in P (%)	£ kg^{-1} saved
Research	Make use of genetic resources in livestock	-173	0.6	-5,386
Fertiliser	Integrate fertiliser and manure nutrient supply	-568	5.9	-1,910
Soil	Adopt reduced cultivation systems	-107	2.8	-765
Fertiliser	Use a fertiliser recommendation system	-69	6.0	-229
Fertiliser	Do not apply P fertilisers to high index soils	-46	6.0	-154
Livestock	Improved feed characterisation	0	0.7	0
Manure spreading	Incinerate poultry litter	0	0.1	0
Manure spreading	Do not spread solid manure at high-risk times	3	3.1	22
Manure spreading	Do not spread slurry at high-risk times	5	4.0	23
Manure storage	Store solid manure heaps away from watercourses and drains	2	1.0	32

4 Tier 2 assessment: Anglian River Basin District

4.1 Introduction to Anglian RBD

The Anglian River Basin covers an area of 27,890 km^2 from Lincolnshire in the north to Essex in the south, and Northamptonshire in the west to the East Anglian coast. The area covered by the RBD is shown in Figure 4.1. This map also shows the main rivers and urban areas in the RBD.

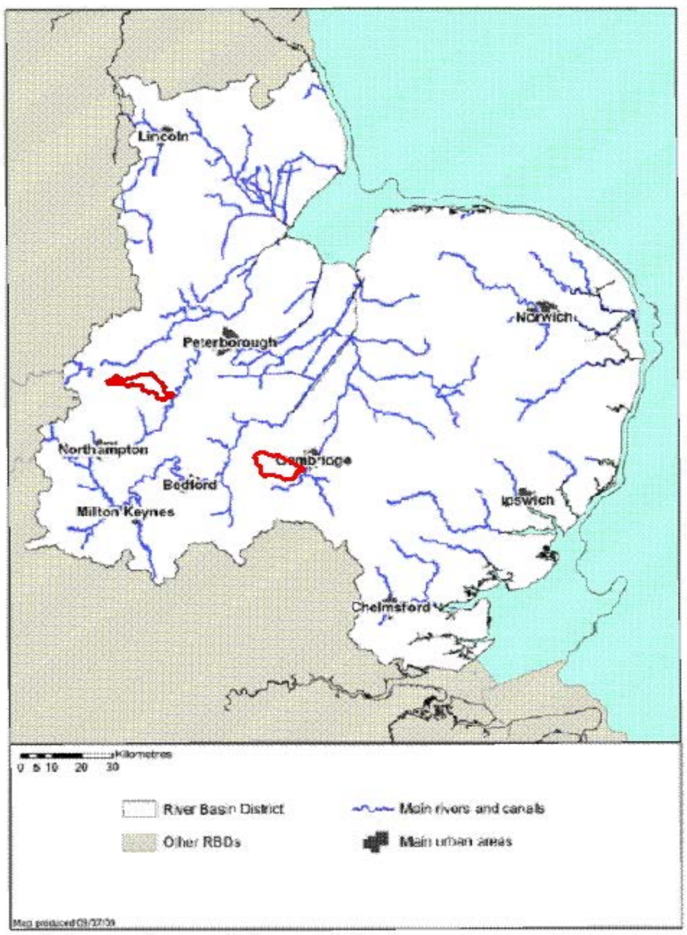


Figure 4.1 Anglian River Basin District and the main rivers and urban areas with pilot catchments outlined in red

There are a total of 758 river water bodies in the Anglian RBD, with a total length of 6,968 km. The district also has 49 lakes and reservoirs, 18 estuaries, 11 coastal water bodies and 31 groundwater bodies. Of the total of 867 water bodies, 154 are designated as artificial water bodies, and 431 are heavily modified.

Eleven main catchments have been identified in the RBD; these are shown in Figure 4.2.

Figure 4.2 Main catchments in the Anglian RBD with pilot catchments outlined in red

At present, 18 per cent of water bodies in the RBD have 'good' ecological status. A third of water bodies have 'high' or 'good' biological status. A range of determinands including physico-chemical, biological and specific pollutants are used to assess ecological status, where a status is given for each determinand, and the lowest status of these components is the final ecological status. Biological status is one component of the ecological result. Figure 4.3 indicates that status for phosphorus is much lower than the other chemical determinands, such as ammonia and specific pollutants, with less than 35 per cent having 'high' (blue) or 'good' (green) status.

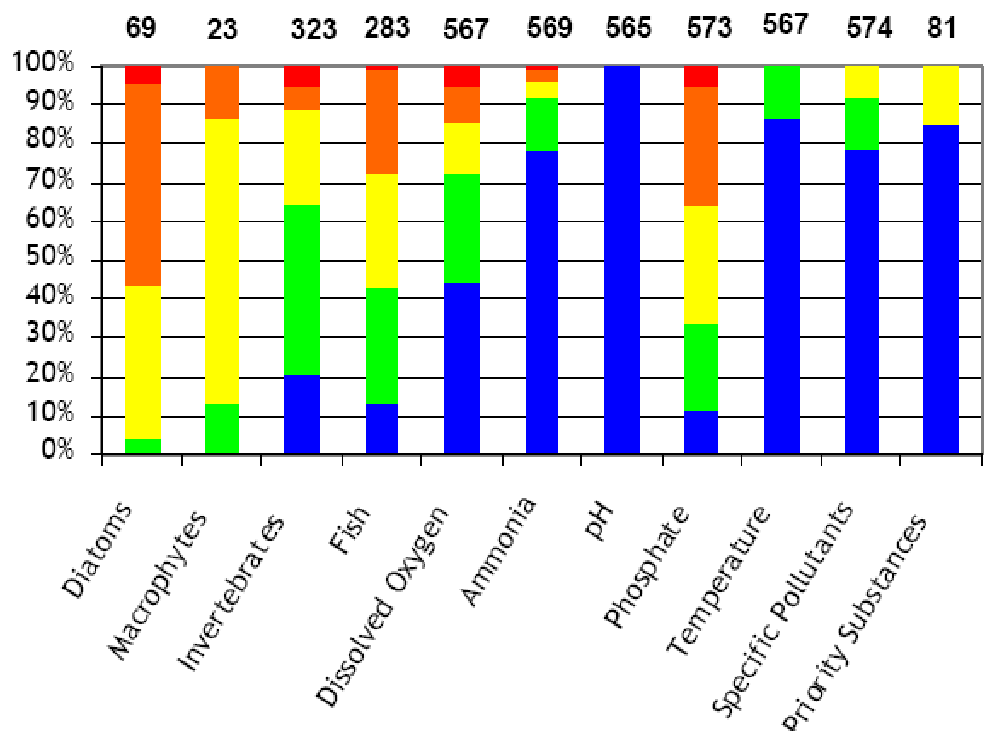


Figure 4.3 Status and number of sites assessed for individual determinands (blue = high status, green = good, yellow = moderate, orange = poor, red = bad)

The River Basin Management Plan for the RBD includes a target to improve the class for phosphorus in 772 km of rivers and seven lakes by 2015.

4.2 Physical and human characteristics

4.2.1 Physical characteristics

The geology of the area is varied and consists of four main rock types: chalk, sandstone, limestone and argillaceous rocks (a type of sedimentary rock like sandstone that contains a significant amount of organic material). A band of limestone is found in the west of the region, chalk in the north on the Lincolnshire coast and in the centre of the region, sandstone on the East Anglian coast, and argillaceous rocks in the south and western areas, as shown in Figure 4.4. Soils are also varied, with widespread clays and loams, to sandy soils in the central area, and peat soils in the fenlands.

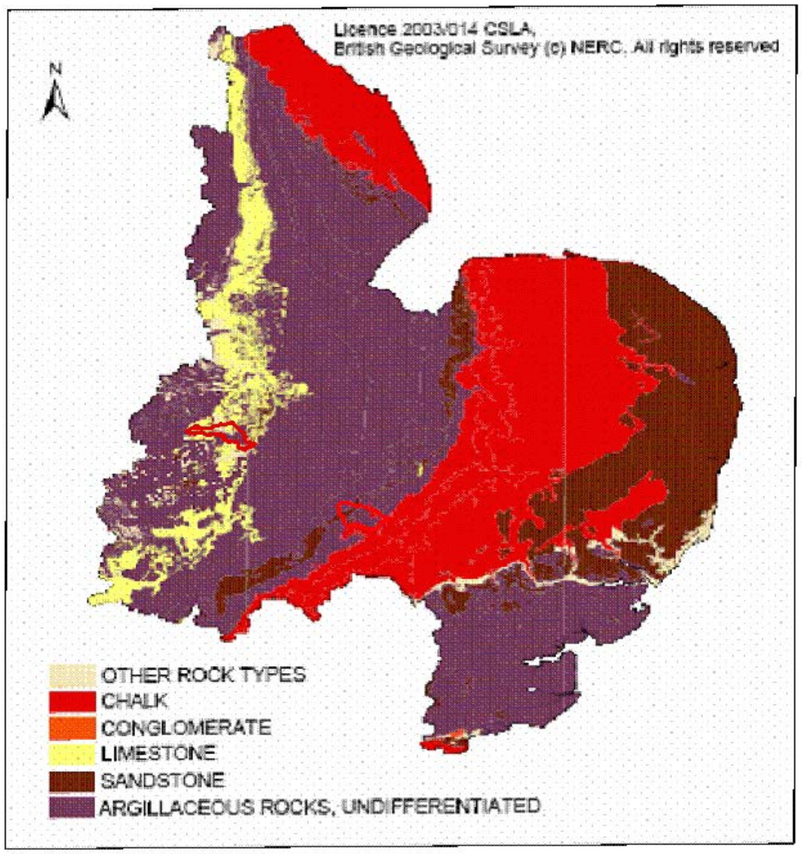


Figure 4.4 Geology in Anglian RBD. Copyright Licence2003/014 CSLA, British Geological Survey (c) NERC. All rights reserved. (From Environment Agency 2010c)

The landscape is varied, ranging from gentle chalk and limestone ridges in the north and west to the extensive flat lowlands of the Fens and the coastal estuaries and marshes.

The Anglian RBD is dry compared with the rest of England and Wales, with some areas receiving only two-thirds of the UK's average rainfall at around 700 mm per year, compared with the UK annual average of 800-3,200 mm.

The region has a wide diversity of habitats and flora and fauna, ranging from wetlands to woodlands and including the Brecks and Chalk Streams. This is reflected in the large number of protected sites designated under UK or European legislation. There are over 560 protected sites in the east of England covering 6.6 per cent of the region.

The river habitat survey carried out across the region in 2007-2008 found that:

- Habitat modification is extensive across the RBD. Most of the sites regarded as near natural or predominantly unmodified are found in the centre and south-east.
- Resectioning is especially common; only 15 per cent of sites have no resectioning, most of which are smaller streams in the centre of the region.
- Bank reinforcement is less widespread, and was only recorded at a fifth of sites. Due to the low energy and gradient of most rivers in the region, the risk of erosion is low, so less protection is needed.

- Over 70 per cent of baseline sites have channel shading from bankside trees. Most of the sites without bankside trees are located in the fens to the north of Cambridge.
- The distribution of features like exposed bankside tree roots and large woody debris reflects the distribution of channel shading, but these features are less common than in other parts of the country due to the lower energy of flows and the extent of channel resectioning.
- Invasive species (Himalayan Balsam, Giant Hogweed and Japanese Knotweed) are found at few sites compared to the rest of England and Wales. Himalayan Balsam was the most widespread, found at five per cent of baseline sites, most commonly in the south of the RBD, while Giant Hogweed was found at only two per cent of sites, and Japanese Knotweed at less than one per cent.
- Unvegetated depositional bars, which act as habitats for invertebrates, were found at just over a fifth of baseline sites. These features were absent in many of the lower-lying survey sites around the central area and coastlines. The low-lying, low energy nature of rivers means that bars are less common than in the rest of England and Wales. Bars are also likely to be absent where river channels have been straightened and deepened.
- All metrics, except the presence of Himalayan Balsam, showed some improvement since the previous survey in 1995-96.

4.2.2 Human characteristics

The Anglian RBD is a predominantly rural area, with a total population of around 5.2 million people in small and medium-sized towns. There are no extensive metropolitan areas. More than 70 per cent of the land area is used for agriculture, and the region is known to be one of the most productive agricultural landscapes in the world, accounting for around 28 per cent of the total agricultural land in the UK. The highest percentages of agricultural land are in Lincolnshire and Norfolk (Environment Agency, 2010c).

The total number of farms in the region in 2008 was around 33,000, with almost half of these being less than five hectares in size. The area is best known for its cereal crops, with farms producing cereal crops making up a quarter of the total number of farms in 2008, and producing more than a quarter of England’s barley and wheat. The area is also known for horticulture, with a wide range of fruits, vegetables and flowers grown. Livestock levels in the region are low, with only 17 per cent of the land dedicated to grass and grazing, and dairy and beef herds and sheep flocks in this area are small compared to other parts of England. However, pig and poultry numbers in some areas are high (Environment Agency, 2010c).

There are a total of 2,304 STW in the RBD, 759 of which have a PE value of more than 250. In addition to these discharges, there are around 19,000 other permitted discharges, including 650 trade discharges, and 1,750 agricultural discharges, which include site drainage and discharges from fish farms.

4.3 Source and exposure pressure

As part of the river basin characterisation process, the Environment Agency assessed the pressures on water bodies and the risk of failing to meet the objectives of the Water Framework Directive by 2015. **These assessments do not reflect the current quality or status of a water body, rather the risk that they may fail objectives as result of pressures acting on them. Being 'at risk' does not mean that a water body has already failed its objectives, only that it might do so.** The risk assessments have been used to target monitoring programmes and to provide the evidence to help develop measures to meet environmental objectives.

There are four categories of risk:

Risk of failing to meet Water Framework Directive objectives	Colour key
Water body at significant risk of failing	Dark purple
Water body probably at significant risk of failing	Light purple
Water body probably not at risk of failing	Pink
Water body not at risk of failing	Pale pink

The Environment Agency has produced guidance on the process of the pressure assessments. As an initial guide, please refer to the Environment Agency Risk assessment technical note (Environment Agency, 2007) which provides a technical overview of the process. This document explains the generation of risk categories and how these categories are combined to create the overall assessment for each pressure. This paper was developed in response to several queries during the Characterisation Review in 2004.

The PSYCHIC model developed by ADAS give estimates of P loss from different sources based on land use, rainfall and soil type, among other factors. The results of this model are used below to show the variation between different sources.

The maps shown in Figures 4.5 to 4.7 are produced from modelled P inputs to rivers on a 1 km x 1 km grid scale taken from the results of PSYCHIC models (the one-km^2 gridded data is converted to ha by dividing by 100). As shown in Figure 4.5, P losses in the RBD are highly variable, with small areas of very high contributions distributed across the RBD. Diffuse P losses are highest in the Northern and Eastern areas of the RBD, peaking at over 1.5 kg ha^{-1} per year. The areas where the pilot catchments are located have lower levels of diffuse P, generally below 0.5 kg ha^{-1} a^{-1} (Figure 4.6a). Losses from point sources reach levels of over 10 kg ha^{-1} a^{-1} in some places (in the PSYCHIC model, human contributions are worked out using gridded population census data and a population equivalent, not STW data). The highest levels are located in small areas across the catchment, which may well be associated with large STW (Figure 4.6b).

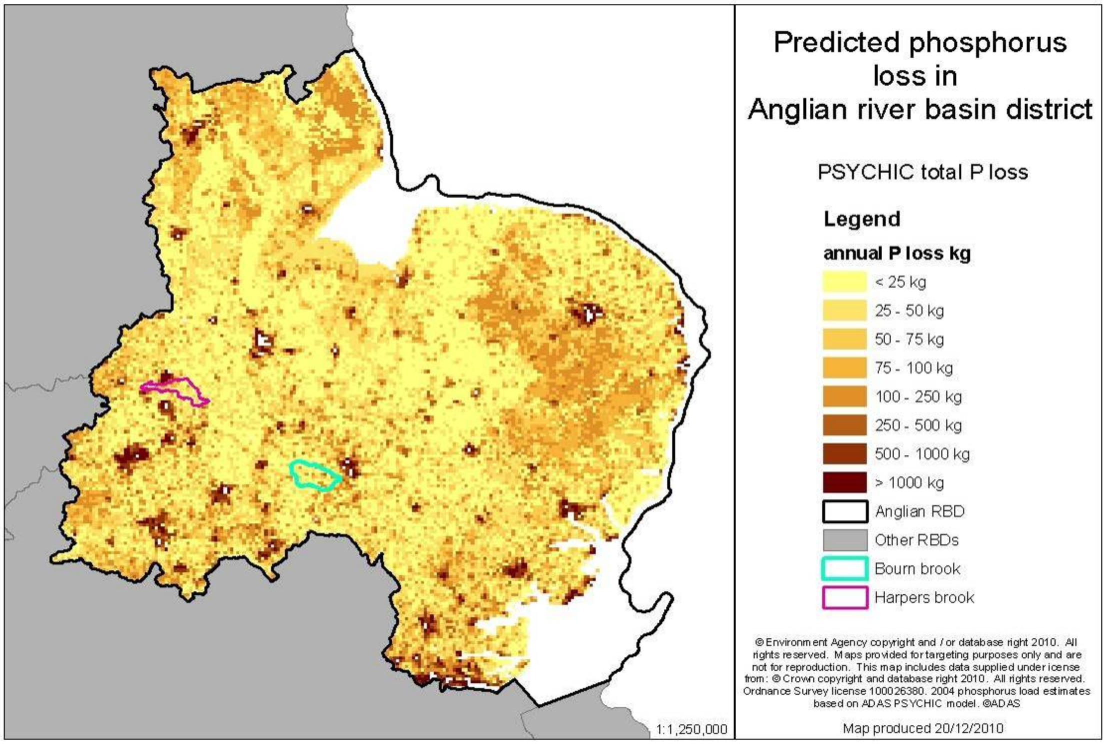
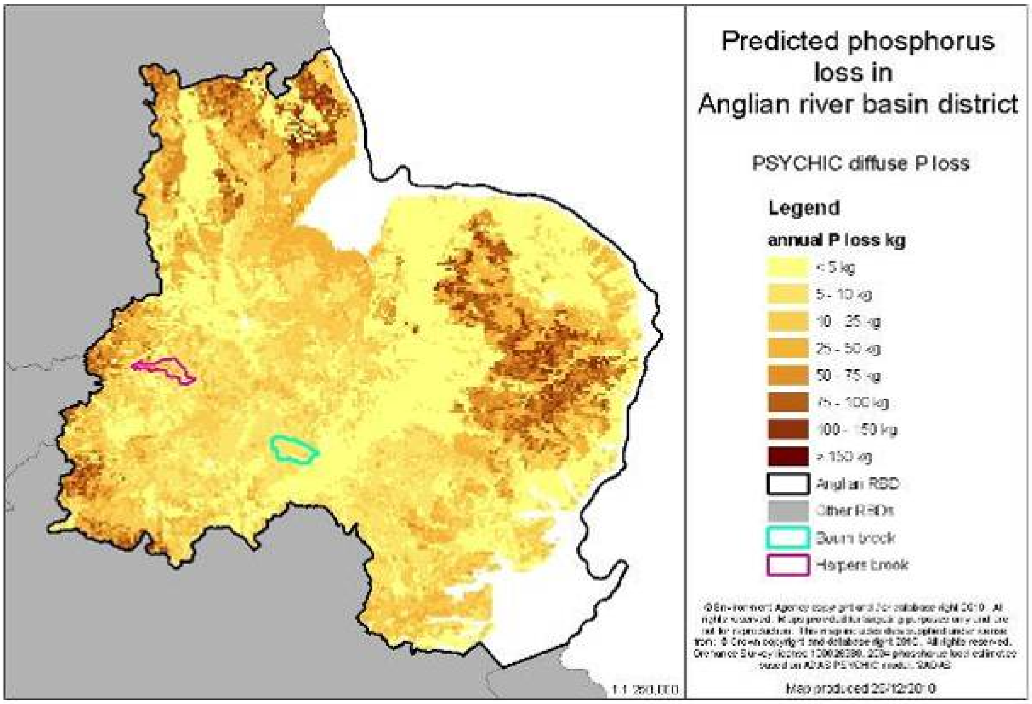
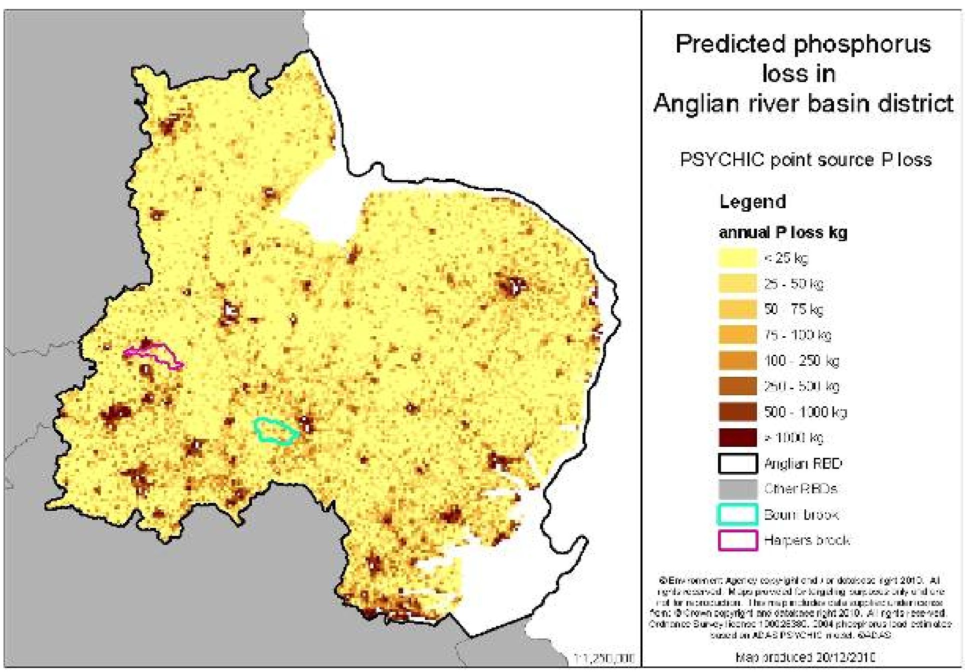


Figure 4.5 P losses from all sources in Anglian RBD



a) Diffuse sources

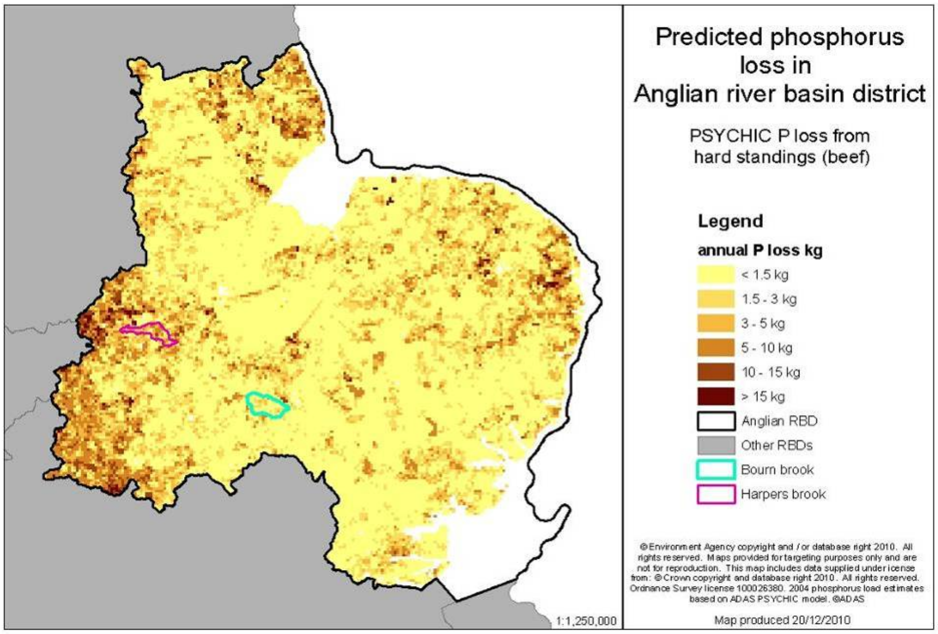


b) Point sources

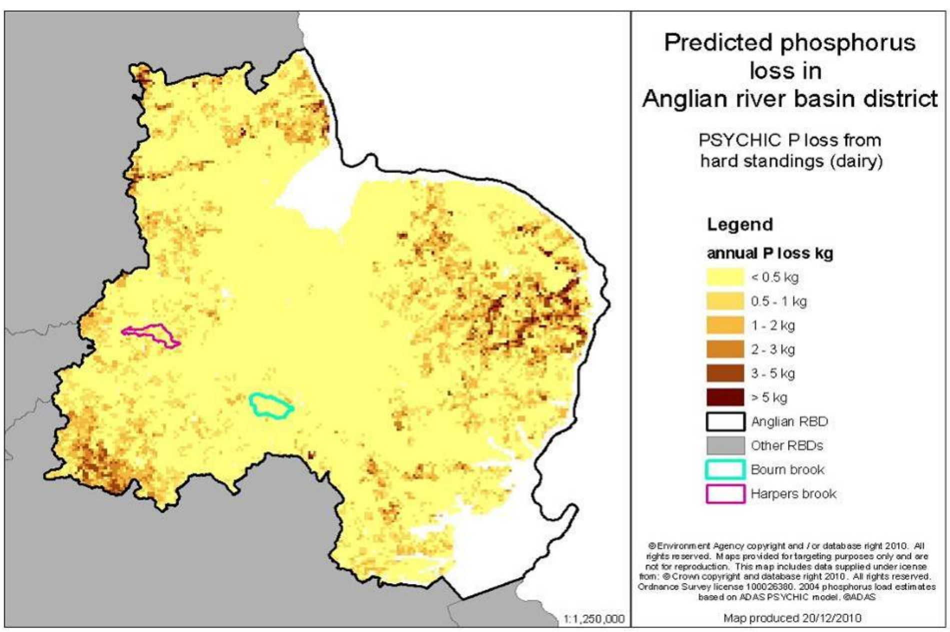
Figure 4.6 P losses from diffuse sources (a) and point sources (b) in Anglian RBD

Agricultural inputs of P come in a range of forms. These include diffuse sources, such as runoff of fertilisers and manures from land, and point sources, such as runoff from areas of hard standing on beef and dairy farms. The PSYCHIC model results show that areas of hard standing showed low overall levels of P loss, peaking at 0.15 kg ha^{-1} a^{-1} on beef farms (Figure 4.7a), and 0.05 kg ha^{-1} a^{-1}on dairy farms (Figure 4.7b). Levels were highest in the south western and eastern parts of the RBD. Losses from diffuse fertiliser sources were also low, with levels at 0.05-0.1 kg ha^{-1} a^{-1} across much of the RBD (Figure 4.7c). Losses from manures were more variable, peaking at 0.6 kg ha^{-1} a^{-1}, but these losses were concentrated in small areas, and across most of the RBD levels were below 0.10 kg ha^{-1} a^{-1}(Figure 4.7d).

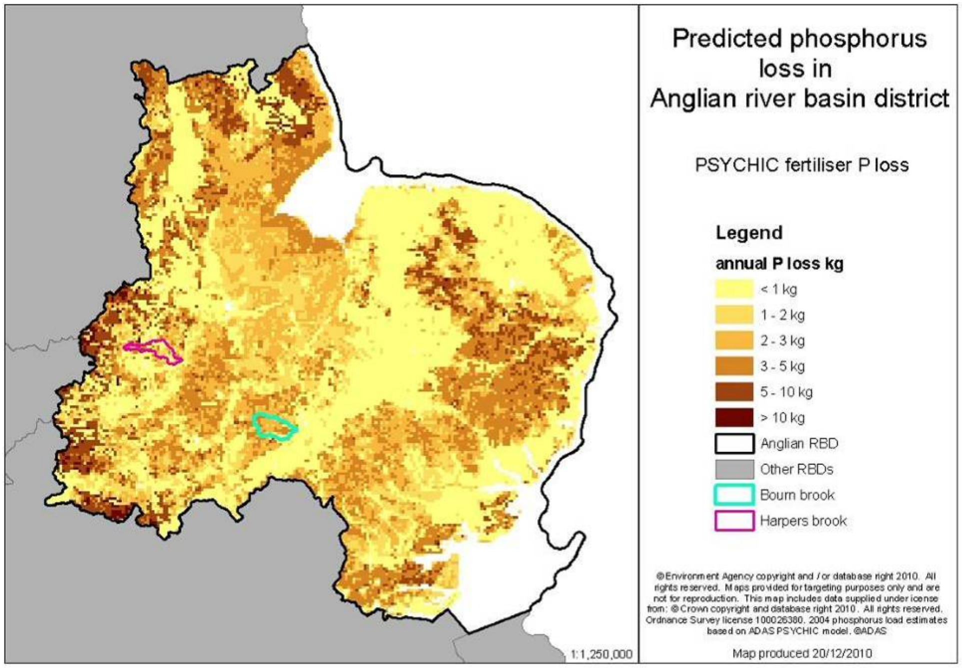
a) Hard standing (Beef)



b) Hard standing (Dairy)



c) Fertiliser



d) Manure

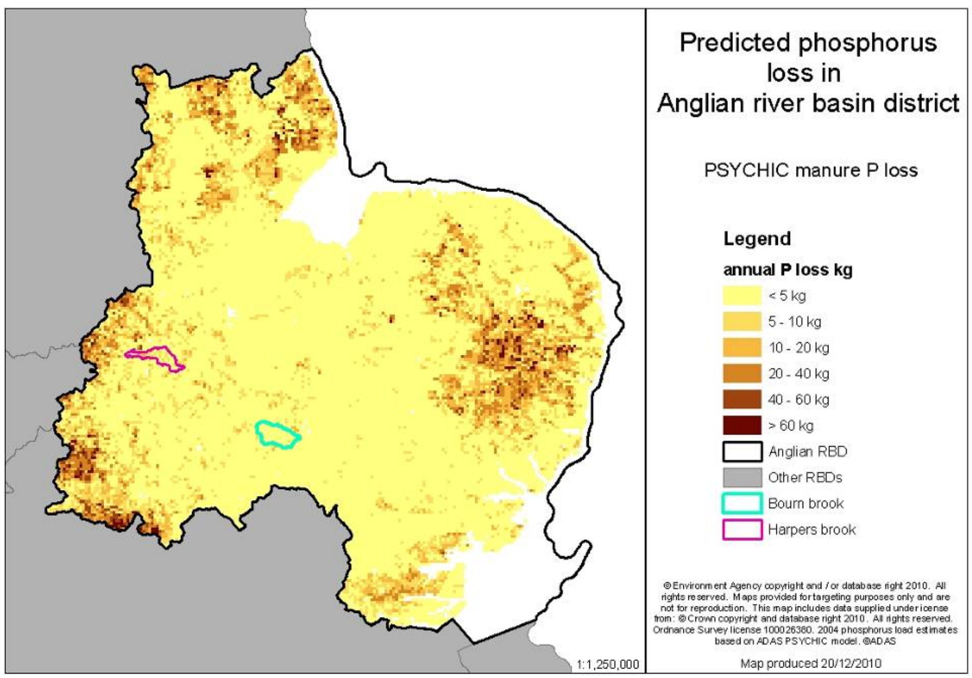


Figure 4.7 P losses from different agricultural sources

Phosphorus losses combine to give a level of risk for all rivers across the RBD from combined point and diffuse sources. Figure 4.8 shows that the Environment Agency found most of the RBD to be ‘at risk’ or ‘probably at risk’, with only limited areas in the east and north ‘not at risk’. Nationally, over 80 per cent of rivers are under pressure from diffuse pollution (Environment Agency, 2010c).

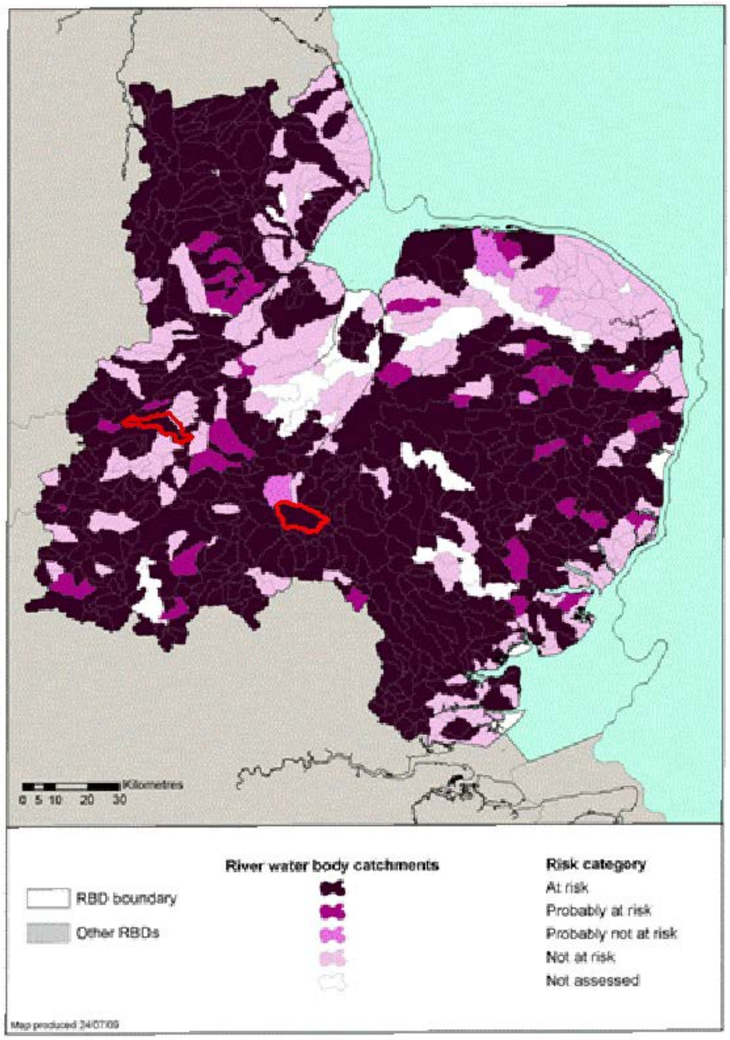
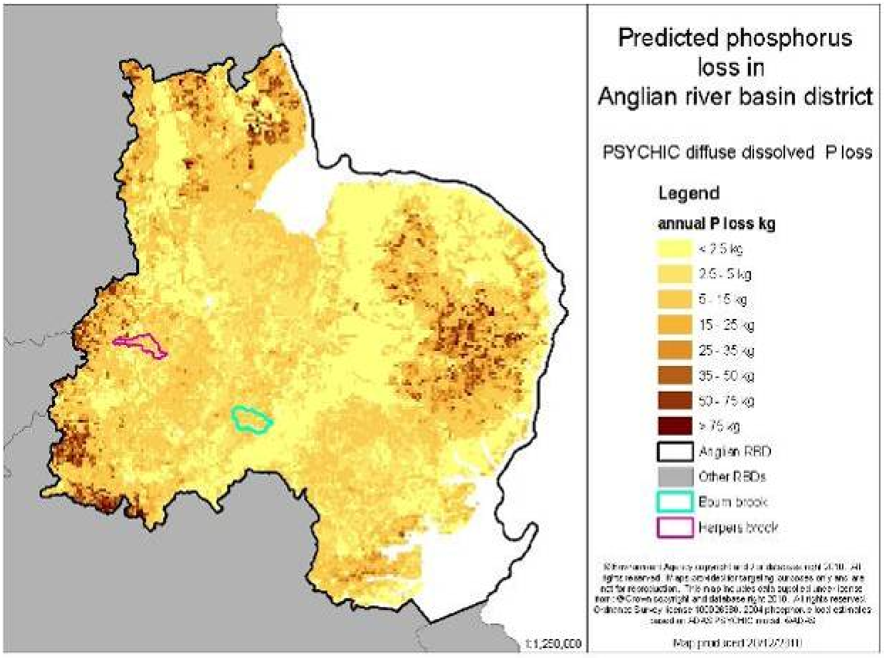
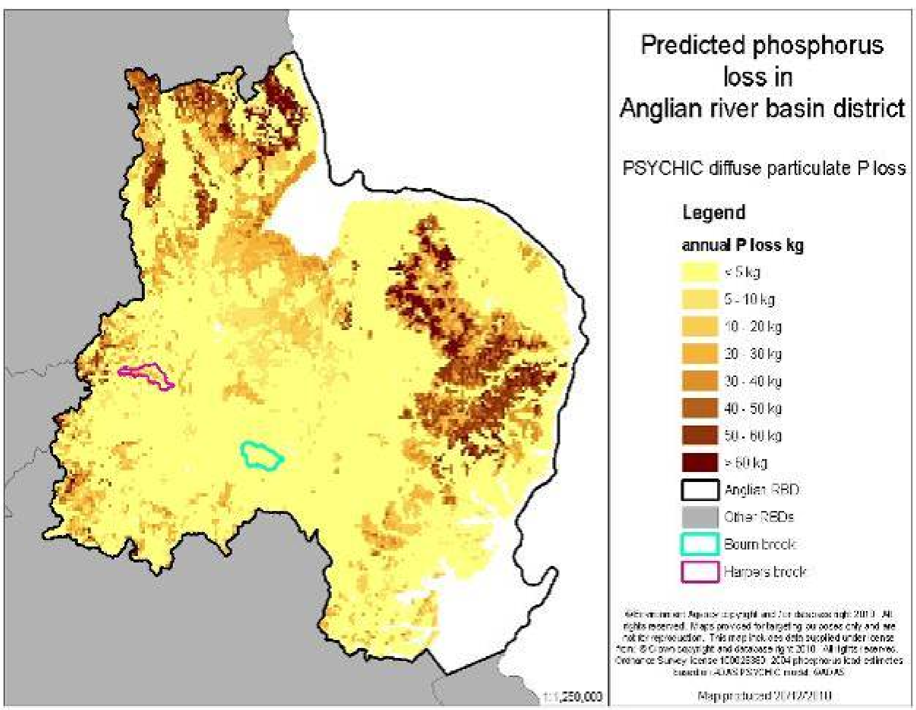


Figure 4.8 Risk to water bodies in Anglian RBD from combined sources of P

The PSYCHIC models also provide information on the form in which P is mobilised, splitting diffuse P losses into dissolved and particulate forms. Overall, concentrations of dissolved P in the catchment are similar to concentrations of particulate P, peaking at over 0.75 kg ha^{-1} a^{-1}, as opposed to 0.60 kg ha^{-1} a^{-1} for particulate forms. Both forms show the highest losses in the eastern and northern parts of the RBD, but concentrations of particulate P are more concentrated in these areas, while dissolved forms have a smaller area of very high values but slightly higher concentrations across the RBD (Figure 4.9).



a) Dissolved



b) Particulate

Figure 4.9 Dissolved and particulate forms of diffuse P in Anglian RBD

Holman *et al.* (2010) investigated groundwater P concentrations across the UK, finding the highest levels in the south east of England, including Anglian RBD, with levels of over 0.12 mg l^{-1} in some areas. As the RBD is predominantly used for arable farming, P levels may be adversely affected.

As shown in Figure 4.10, the Environment Agency found no areas of groundwater in Anglian RBD 'at risk' or 'not at risk' from diffuse sources of P. The RBD is split roughly between 'probably at risk' or 'probably not at risk'. This lack of certainty could be down to a lack of monitoring data, or a lack of evidence of the impacts of diffuse sources of P. Nationally, over three-quarters of groundwaters are thought to be at risk from diffuse sources of pollution (Environment Agency, 2010c).

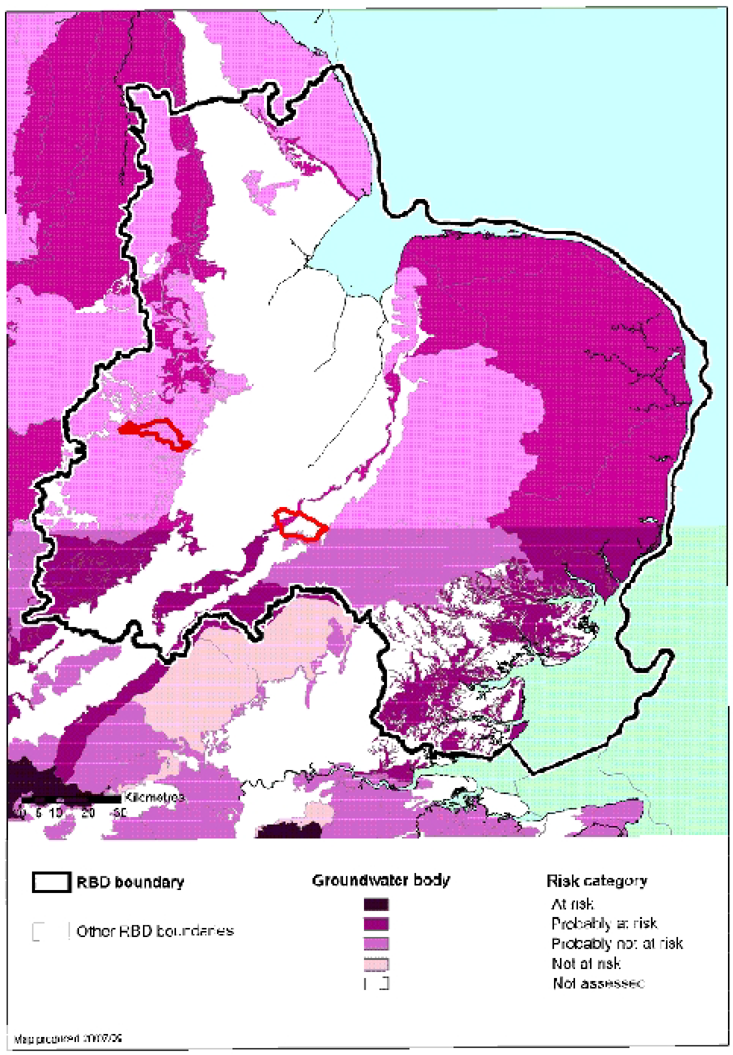


Figure 4.10 Risk to groundwater bodies in Anglian RBD from diffuse P sources with pilot catchments outlined in red

4.4 Water quality and ecological impacts

The WFD RBC2 risk assessment maps identify water bodies at higher or lower risk from agricultural sources of P. This map is based on Environment Agency monitoring data (TRP) and estimates of agricultural P loadings to the upstream contributing catchment, taken from the ADAS PSYCHIC model. In water bodies where the estimates of P loading from agriculture (per unit rainfall) are relatively high and observed water quality exceeds the WFD TAG agreed P targets for good chemical status, the water body said to be at high risk from P *and* agricultural pollution is *more likely* to make a material contribution to that risk. **The risk maps do not claim that agriculture is the main or only contributor to the observed P excess over threshold, and should not be interpreted on a water body-by-water body basis.** Figure 4.11 shows the Environment Agency's assessment of the level of risk to each river water body from agricultural sources of P. Most of the RBD is 'probably at risk', with the highest level 'at risk' in the east of the catchment. Only a few water bodies (around four) are definitely 'not at risk'.

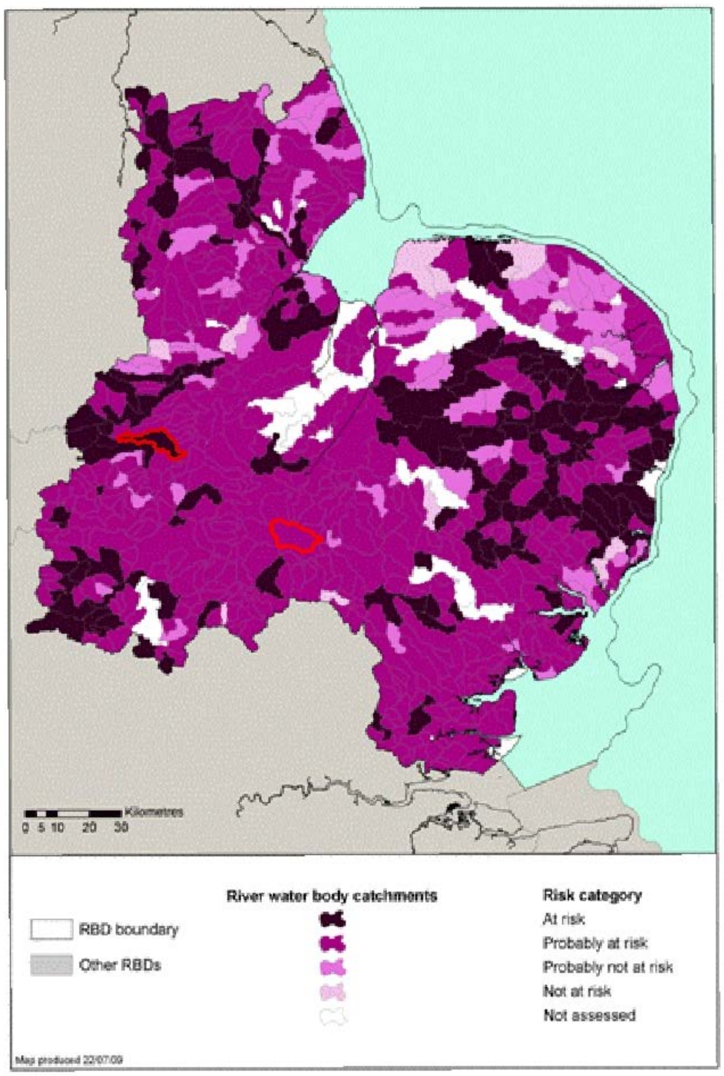


Figure 4.11 Risks to river water bodies from agricultural sources of P with pilot catchments outlined in red

Recent years have seen a significant decrease in areas using phosphate fertilisers, from half the RBD land area in 2005 to 40 per cent in 2008, in line with the decrease of 13 per cent across England and Wales. For the East of England, application rates are consistently higher than the rest of the country, despite recent decreases (Environment Agency, 2010c). However, it is the historic use of P fertilisers which has resulted in high concentrations of P in soils.

STW are another significant source of P inputs to the region. STW consents restricting phosphorus input to rivers in the RBD were applied to 56 works in 51 separate water bodies under the AMP3 and AMP4 investment programmes. The WFD ecological status in these catchments is shown in Table 4.1 below.

Table 4.1 Ecological status of 51 water bodies containing STW with P consents

Ecological status	Number of water bodies
High	0
Good	2
Moderate	42
Poor	5
Bad	1

These water bodies were also evaluated to see how much risk was posed to them by agricultural sources of P; the results of the assessment are shown in Table 4.2.

Table 4.2 Risk from agricultural P to water bodies containing STW with P consents

Risk status	Number of water bodies
At risk	12
Probably at risk	26
Probably not at risk	5
Not at risk	0
No data	8

These tables indicate that the majority of water bodies which contain STW with consents on P discharges are still likely to fail the WFD ‘good’ status, and be at risk from agricultural sources of P. All but one of the water bodies not already at ‘good’ status have targets set to achieve this status by 2027, as ‘good’ status is not likely to be achievable by 2015.

Impacts on the ecology of the RBD from P are difficult to assess as the communities which are affected by increased levels of P (macrophytes and diatoms) are only measured at 23 and 69 sites respectively. Where these assessments are undertaken, very low status classes are found, with less than 15 per cent of macrophytes sites and less than five per cent of diatom sites at good status, and no sites at high status. It may be that only areas with problems are being assessed, but more wide ranging sampling would obtain a comprehensive picture of the ecological state of the RBD. The diatom and macrophyte classification tools are being revised, and will be used to re-evaluate the current status as part of the WFD investigation work.

4.5 Management responses

Table 4.3 shows the changes in pollutant losses with no measures in place and with maximum level of implementation for 2004, comparing changes in Anglian RBD with England and Wales as a whole (Anthony *et al.* 2009).

Table 4.3 Calculated pollutant losses with no implementation and maximum implementation of measures for the year 2004 for Anglian RBD and England and Wales (from Anthony *et al.* 2009 Appendix A)

Type of agriculture	P losses in Anglian RBD (kt)		P losses in England & Wales (kt)	
	No measures	Maximum implementation	No measures	Maximum implementation
Dairy	0.0	0.0	1.2	0.4
Horticulture	0.0	0.0	0.0	0.0
Lowland C&S	0.0	0.0	0.7	0.3
Mixed	0.1	0.0	0.7	0.2
Mixed combinable & pig	0.1	0.0	0.4	0.2
Outdoor pigs	0.0	0.0	0.0	0.0
Roots and poultry	0.1	0.1	0.4	0.1
Upland C&S	0.0	0.0	0.9	0.5
Winter combinable & pig	0.1	0.1	0.6	0.2
Total	0.5	0.2	5.0	2.0

The net cost of reducing P losses following maximum method implementation across the whole of England and Wales was estimated at around £900 kg^{-1}, which was much higher than all other determinands investigated (suspended sediment was estimated at £3 kg^{-1} and nitrate at £20 kg^{-1}). Costs for P reductions were highest on roots and poultry (£204 kg^{-1}) and dairy (£129 kg^{-1}) farms, and lowest on horticultural lands (£16 kg^{-1}). The Anglian RBD also had the highest costs of all RBDs for reducing P losses at £158 kg^{-1} for full method implementation, although costs to tackle other pollutants were lower relative to other regions.

Of the 69 methods used, 11 had no net cost, or showed a net saving for the agricultural sector, and nine cost less than the estimated benefits to society (for all determinands combined). A cautious conclusion would be that a 30 per cent reduction in P could be achieved using only methods which saved money or were of direct interest to farmers. However, estimates of cost savings in this project are vulnerable to assumptions about a number of measures. In addition, incentive from cost saving may mean that some measures may already have been implemented.

A wide range of measures are being implemented in the RBD to tackle pressure on water quality from phosphorus and other nutrients from agricultural activities. These include:

- Schemes under the Environment Agency's England Catchment Sensitive Farming Delivery Initiative (ECSFDI) in areas at risk from diffuse pollution from agriculture.
- Further measures to tackle issues including soil erosion and diffuse pollution under Natural England's Entry Level and Higher Level Stewardship (ELS and HLS).
- Compulsory schemes in Natura 2000 protected areas, including agri-environment measures, changes to discharge/PPC consents, specific works to address water quality, investigation of water quality and morphology pressures and development of pollution action plans.
- Education on diffuse pollution through local urban partnerships and campaigns.
- Promotion of good land management practices to reduce pollution incidents.
- Mapping of drainage discharges to assess their impact on water quality status.

A number of local projects are aiming to tackle nutrient inputs. These include: investigation of the effectiveness of agricultural pollution control measures in the Wensum catchment; advice for farmers on best practice in soil management to reduce erosion and runoff; and nutrient manure and pesticide management to reduce inputs to land. Further local projects are looking at:

- grazing and conservation to improve water quality and quantity in Halvergate marshes;
- increasing grazing in Lincolnshire Coastal Grazing Marsh;
- one-to-one farm advice and workshops on a range of topics under the River Ouse Strategic Partnership;
- specialist workshops to address pressures in areas identified via Environment Agency's Agricultural Evidence Base.

5 Tier 3 assessment: Anglian pilot catchments

5.1 Background

Pilot catchments have been used within this project to provide evidence of the agricultural contribution to phosphorus pollution.

In order to introduce voluntary agricultural mitigation measures, sufficient confidence in the evidence on a range of issues is needed, including current chemical and ecological status, identification of the pressure causing the problem, source apportionment for the pressure, and the cost-effectiveness and proportionality of the measures proposed. The use of pilot catchments will help to improve this confidence.

5.2 Catchment selection

5.2.1 Initial selection

The two pilot catchments were selected using a two-stage screening process. In the first phase of the process, water bodies were selected if:

- They fell entirely within the boundaries of the East of England or East Midlands Regional Government Offices.
- They were not included in a catchment initiative to tackle diffuse pollution.
- They failed to achieve good status for phosphorus, with no other failing elements.

A total of 75 water bodies were selected, with 13 in the central Anglian region and 62 in the north Anglian region.

The second stage of the screening process was more subjective, with input from a range of partners. It was decided that catchments should be chosen where conditions could be replicated easily and where a range of contrasting soil types and farm systems were present. Catchments considered by the Environment Agency to have high proportion of agricultural P input were deemed desirable, to help to engage farmers.

The two catchments initially selected for investigation were Harpers Brook and the River Linnet. These catchments are both designated as heavily modified water bodies under the WFD, and both currently have ‘moderate’ ecological potential. The objective for both catchments is to reach ‘good’ ecological potential by 2027, with no targets set for 2015 due to disproportionate expense and technical infeasibility. Phosphorus is the only determinand preventing achievement of ‘good’ status in each of the catchments. Both catchments are also designated as protected areas (NVZ) under the Nitrates Directive (91/676/EEC).

The River Linnet was selected as it is a small, discreet catchment with suitable amounts of monitoring data, and strong links with the farming community. However, the Harpers Brook catchment is a better representative of the complete RBD than the River Linnet.

5.2.2 Catchment walkovers

Once the initial catchment selection was made, catchment walkovers were undertaken to confirm the suitability of the catchments for investigation. The two catchments were visited over a two- to three-day period in September 2010 to identify potential sources of P and, in conjunction with information on landscape attributes (topography, soil type), river flows and P data and catchment modelling (PSYCHIC) outputs, help assess the contribution of farming activities to riverine P loads. The overall purpose of the visits was to determine whether a targeted campaign to adopt more sensitive farming practices and P mitigation practices in the catchment would reduce ambient river P concentrations and help achieve the P targets set by the Water Framework Directive (WFD).

Starting at the catchment outlet, upstream sections of each river were surveyed to assess the type of farming (crops, cultivation type), physical landscape features (slope, presence of buffer features), proximity of impervious surfaces (farmyards, tracks, roads), proximity and predominance of urban development (villages, towns, sewage treatment works, septic tanks) and evidence of runoff pathways (surface runoff and drainflow). In each catchment, at least one farmer was interviewed for more precise details of catchment characteristics, the historical development of farming activities (crops and livestock) and on soil P levels and P inputs. Full notes from the catchment walkovers can be found in Appendix A.

The River Linnet catchment was subsequently found to be unsuitable for this work because:

- historic land use is substantially different from current activities, having included dairy farming and larger numbers of pigs than are currently present;
- a STW discharges into the top of the catchment at a P concentration of around 5 mg l^{-1}, having a significant impact on water quality;
- the river dries up at one point and flow recommences in the town of Bury St Edmonds. The monitoring point is in the town, so cannot be fully representative of the whole catchment;
- the current land use already includes wide buffer strips.

As a result, the Bourn Brook catchment was suggested as an alternative. A number of issues were identified with this catchment, in particular water quality downstream of the Bourn STW, which appeared to be dominated by the point source discharge. To avoid the influence of Bourn STW, the upper part of the catchment was proposed – see Figure 5.7.

5.3 Characteristics of Harpers Brook

Figure 5.1 shows the Harpers Brook pilot catchment, which is located in the Nene catchment, in the Northern area of Anglian RBD, as shown in Figure 4.2.

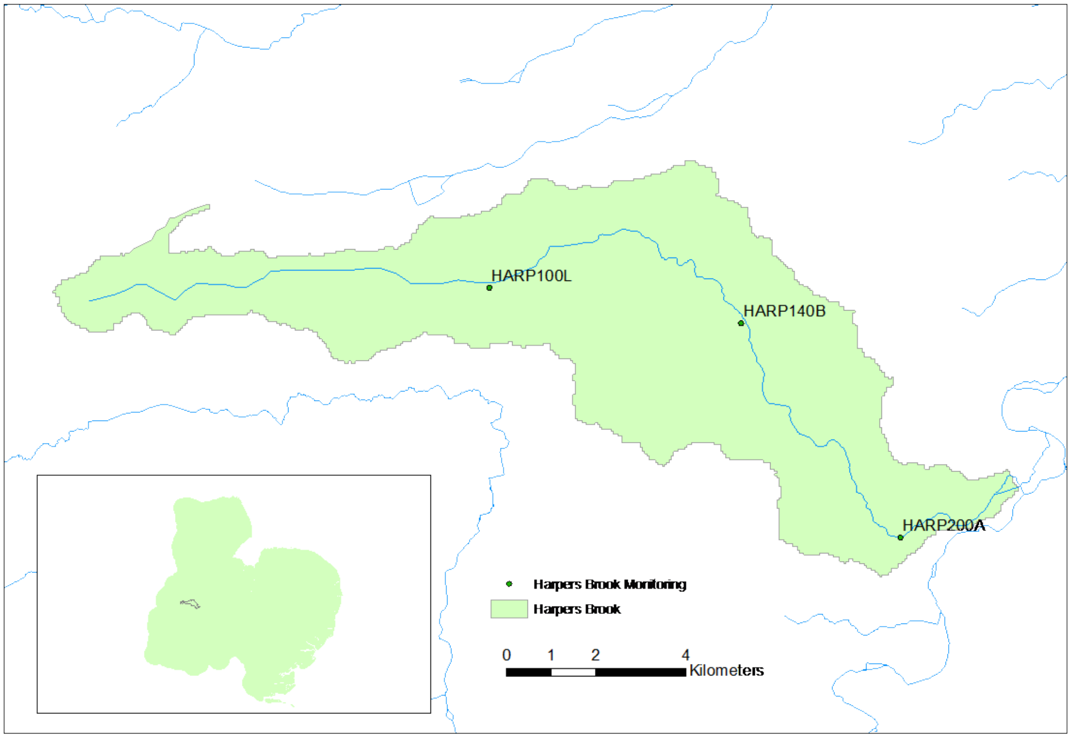


Figure 5.1 Map of Harpers Brook pilot catchment with water quality monitoring points

5.3.1 Physical characteristics

Harpers Brook is 16.76 km long, with a catchment area of 70.08 km^2. Harpers Brook is a tributary of the River Nene, and is a hydrologically discrete catchment, with no upstream water bodies. It is defined under the WFD as small, low lying and calcareous, with high alkalinity (P class 3n/4n).

The soils in this catchment are dominated by clay, with an area of deep loam in the centre of the catchment. Soil P levels obtained from anecdotal evidence during the catchment walkovers (Sep 2010) are not high (P Index 1/2) but little information was available to confirm this. The geology is a complex mixture of mudstone at the head and bottom of the catchment, with limestone dominating the centre alongside small amounts of sandstone and clay.

The elevation of the catchment ranges from 140 m AOD at the top of the catchment, to 30 m AOD at the confluence with the River Nene. Slopes in the catchment are relatively steep, as shown by the 10-m contours in Figure 5.2.

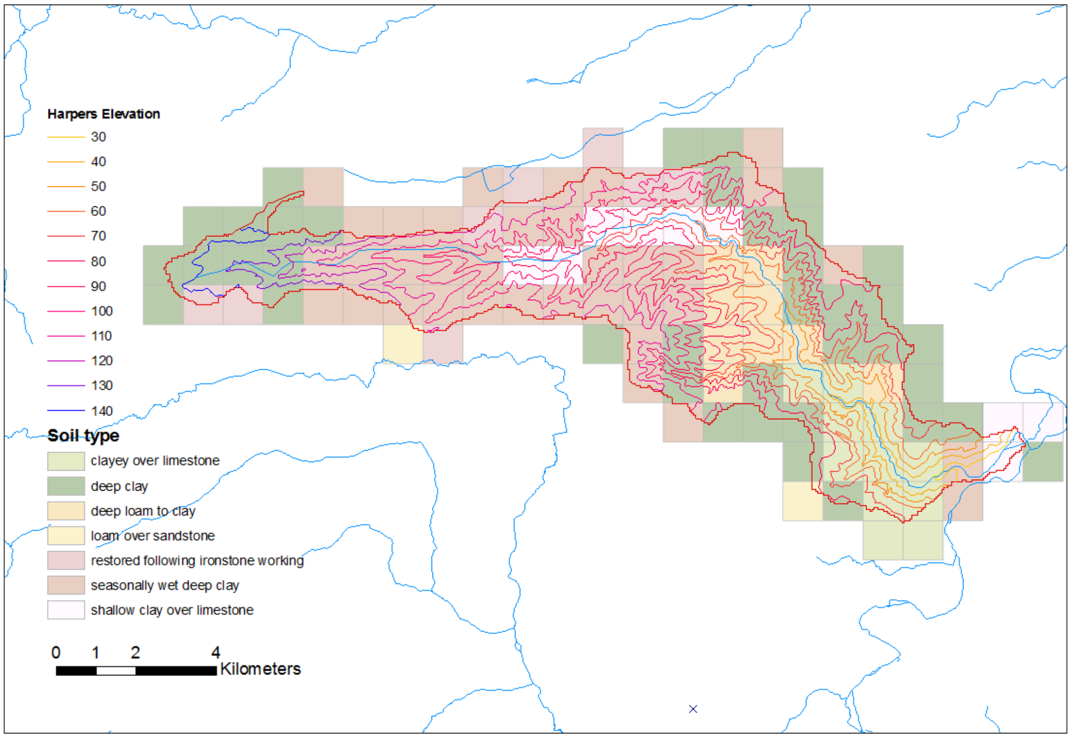


Figure 5.2 Soil type and elevation in Harpers Brook catchment

Rainfall levels in the catchment are generally highest in August and December, with the lowest levels seen in February. Long-term average flow data shows a different profile to the rainfall, with the highest flows in the winter months of January and February, and the lowest flows in July, August and September. Monthly flow and rainfall values are shown in Figure 5.3.

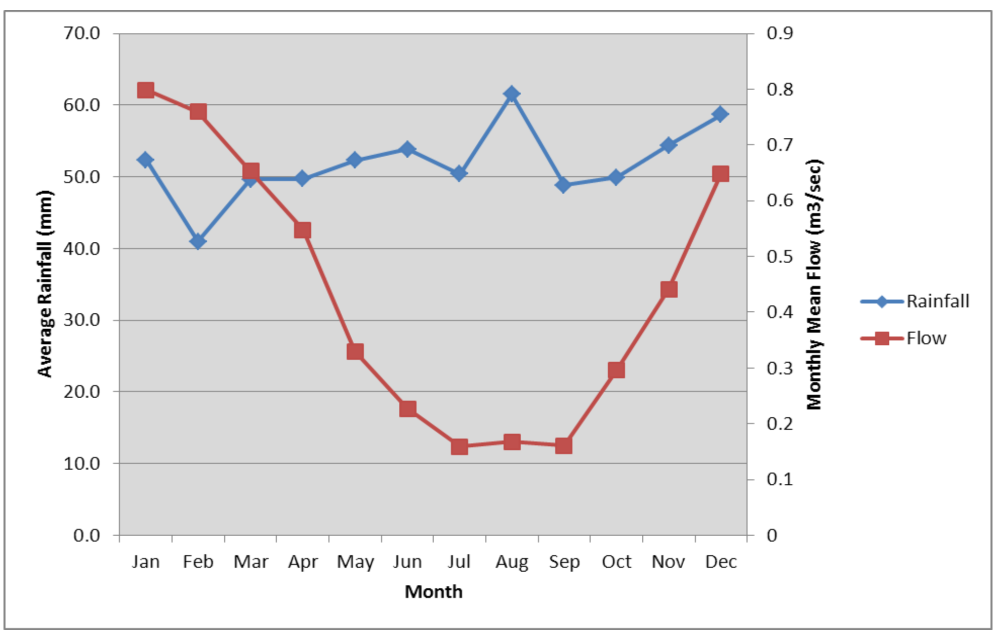


Figure 5.3 Monthly average flow and rainfall values in Harpers Brook catchment

The base flow index (BFI) of the water body is 0.43. A dense network of ditches drains the land area, and were still running when the catchment walk was undertaken in dry weather. There are also a large number of natural springs (often located in woodland) and anecdotal evidence (catchment walkover – September 2010) suggests that the brook responds rapidly to rainfall.

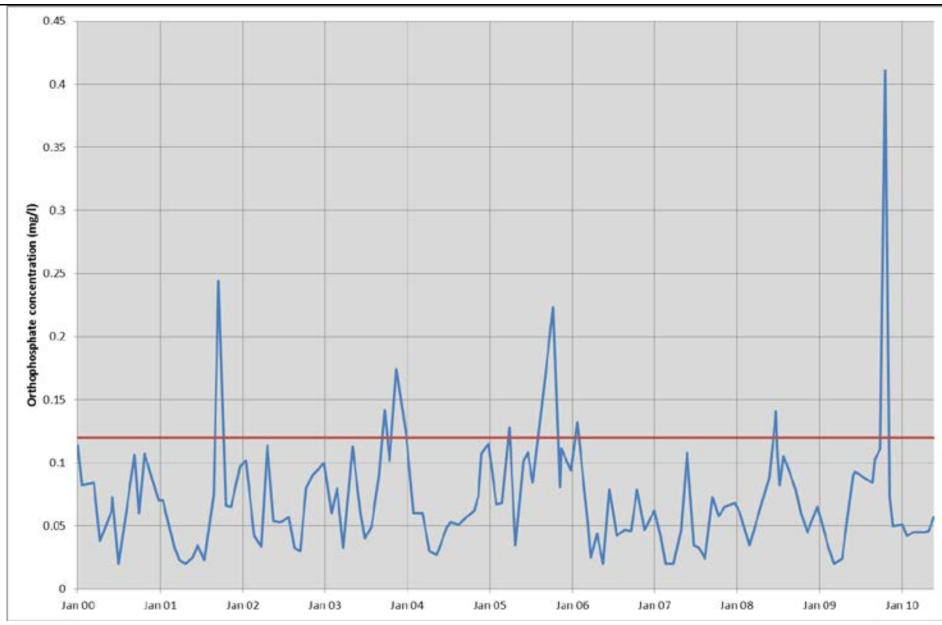
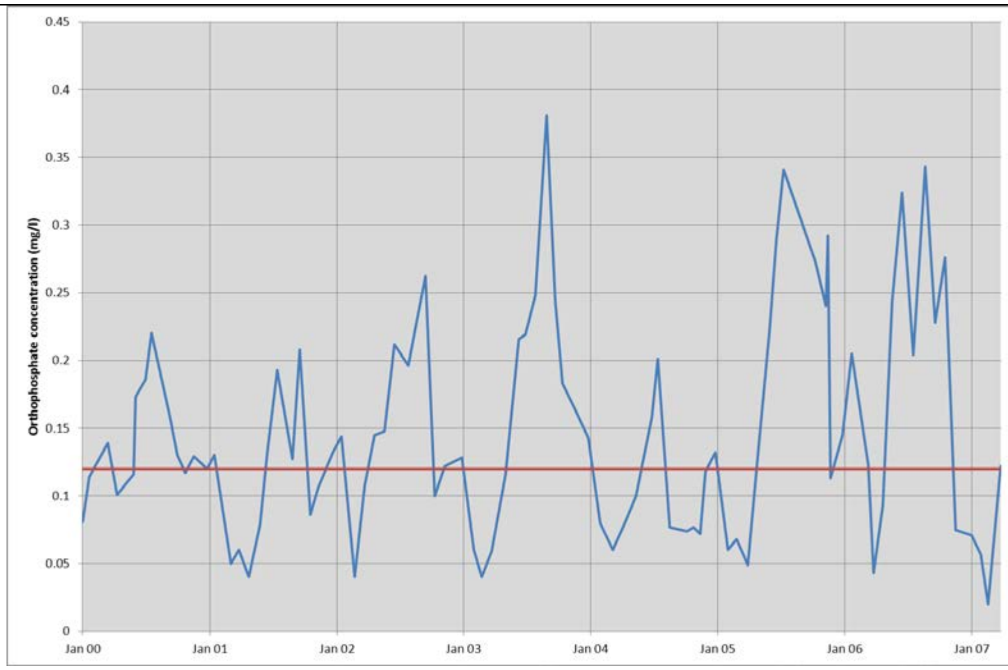
Water quality in the catchment has been assessed for the WFD. Biological quality is good (macroinvertebrates: high, fish: good), while physico-chemical status varies across determinands. Dissolved oxygen, pH and ammonia are all of 'high' status, while phosphorus status is assessed as 'moderate'. The hydrology of the catchment is 'good', and chemical status is not measured.

The WFD has identified a number of risks to the water body, which include diffuse inputs of pesticides, sediments and nutrients, as well as changes in morphology and impacts of alien species.

There are three water quality monitoring sites in the catchment, as shown in Figure 5.1, which all measure phosphorus levels. No monitoring has been done of macrophytes and diatoms. Measured orthophosphate levels are shown in Table 5.1, and trends at the monitoring sites are shown in Figure 5.4. These plots do not show any significant increasing or decreasing trends over the ten years of monitoring, but do show some seasonal variation. They clearly show the failure to achieve WFD P standards of 0.050 mg SRP l^{-1} (high) and 0.120 mg SRP l^{-1} (good).

Table 5.1 Results of orthophosphate monitoring in Harpers Brook catchment (monitoring sites shown in Figure 5.1)

Site	Years	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
HARP100L	2000-2010	0.07	0.13	0.41
HARP140B	2000-2007	0.15	0.31	0.38
HARP200A	2000-2010	0.18	0.33	0.38

HARP100L	
HARP140B	

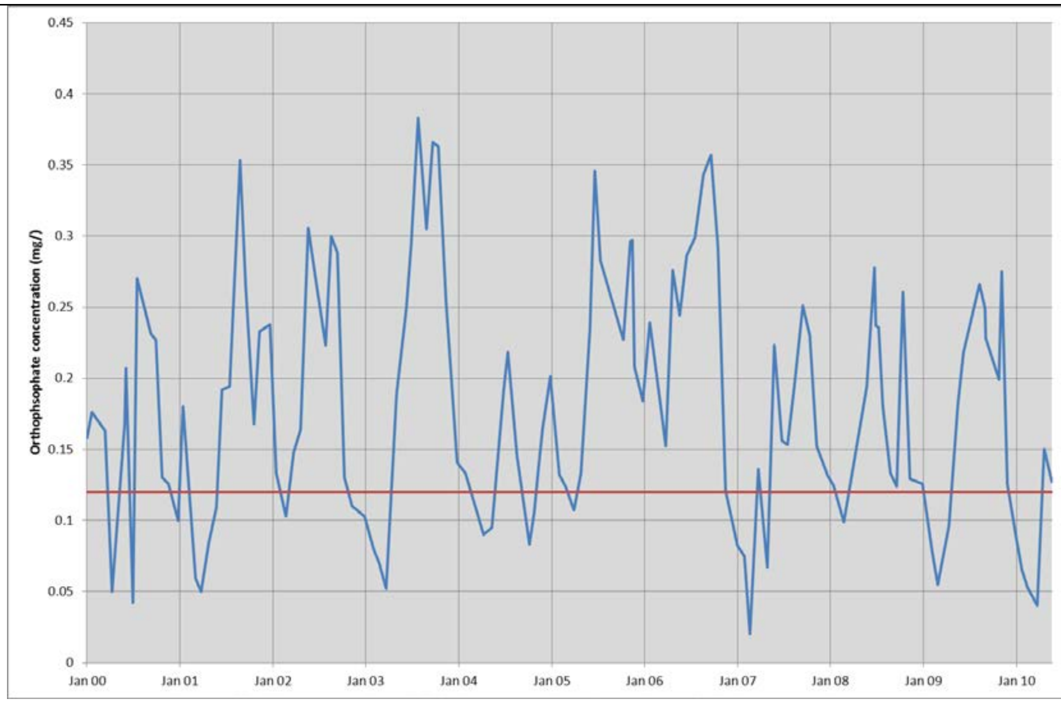
HARP200A	

Figure 5.4 Orthophosphate concentrations at monitoring sites in Harpers Brook catchment (red lines represent the WFD P standard)

The River Habitat Survey carried out on Harpers Brook in 2007-2008 showed no obvious modifications to the river channel, although there was minor channel realignment in some areas. The level of tree cover on the river banks varied from continuous to none, but was mostly rated as scattered or occasional clumps. Some alien plant species were present, in the form of Himalayan Balsam, but no Hogweed or Knotweed was observed. Chubb and trout were fairly abundant in the brook and water quality was considered by the people interviewed in the catchment walkovers to have improved in recent years. Regular flushing during storm events helps to maintain a gravel bottom in many river sections. The brook has been hydrologically modified in the past and at least one farmer interviewed suggested that surface runoff from recent development in Corby was responsible for much quicker flows and response than previously. Corby’s urban contribution to the diffuse and total P load is not currently quantified, but this will be part of catchment characterisation in Phase 2. Overall, the river was rated as being mostly natural or semi-natural, with some artificial features present, and impacted in some areas by agriculture and channel resectioning.

5.3.2 Human characteristics

Harpers Brook only fails to meet P standards by a small margin, and models suggest an agricultural P input of 37 per cent of the total (Environment Agency, 2010d). The catchment is designated as a heavily modified water body for flood protection, and as a surface water NVZ.

A number of mitigation measures are in place to tackle problems with morphology; these include vegetation control, control of invasive species, retention of marginal aquatic and riparian habitats and channel maintenance.

Agricultural land use in the catchment is dominated by arable farming with mostly cereal crops, oilseed rape and beans. Land use data for the catchment taken from the EDINA AgCensus

2004 database is shown in Table 5.2. There are only a few livestock farms in the catchment, of which at least one is organic, but dairy farming has been extensive in the past. As such, only small amounts of manure are recycled.

Table 5.2 Land use in Harpers Brook catchment (EDINA AgCensus 2004)

Type	Value	Unit
Temporary grass	2.84	ha km^{-2}
Rough	0.16	ha km^{-2}
Other arable	0.26	ha km^{-2}
Fallow	0.03	ha km^{-2}
Crops & bare fallow	40.27	ha km^{-2}
Woodland	2.40	ha km^{-2}
Set aside	6.80	ha km^{-2}
All other land	0.84	ha km^{-2}
Horticulture	0.01	ha km^{-2}
Permanent grass	11.49	ha km^{-2}
Total cereals	26.92	ha km^{-2}
Dairy	1.89	head km^{-2}
Beef	3.46	head km^{-2}
Cattle	17.79	head km^{-2}
Sheep	67.98	head km^{-2}
Goats	0.04	head km^{-2}
Pigs	1.38	head km^{-2}
Poultry	115.49	head km^{-2}
Ducks	0.04	head km^{-2}
Geese	0.01	head km^{-2}

The catchment walkovers noted that most fields in the river corridor are cultivated, with tramlines running both across slope and down slope. Minimal cultivation is widely practised. Riparian buffer strips varying from 2-10 m are widespread (mostly six metres wide). At one point, there was a tractor entry point into the brook.

There are two small STW in the catchment, Brigstock STW (population equivalent = 1,367) and Stanion STW (PE = 955), as shown in Figure 5.6. The orthophosphate levels in the STW effluent are shown in Table 5.3, and Figure 5.5.

Table 5.3 Effluent orthophosphate levels from STW in Harpers Brook catchment

Site	Year	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
Brigstock STW	2000-2009	3.52	4.94	5.96
Stanion STW	2000-2009	8.33	12.79	15.30

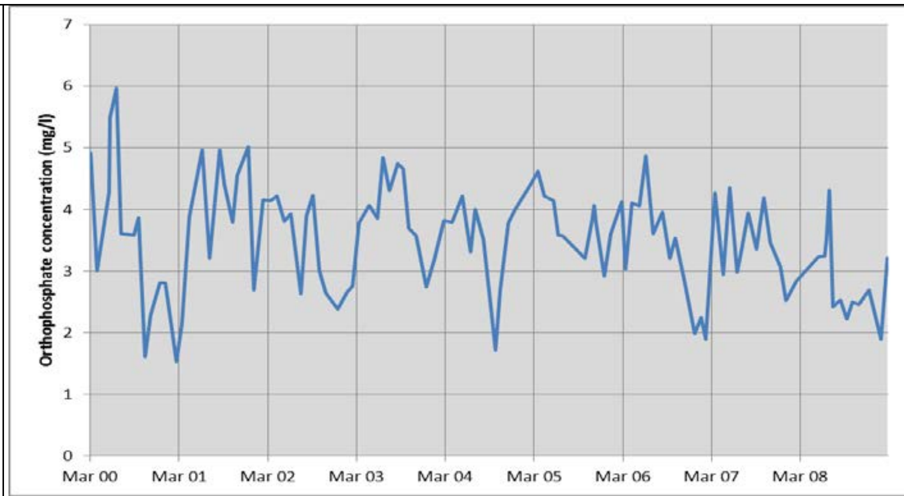
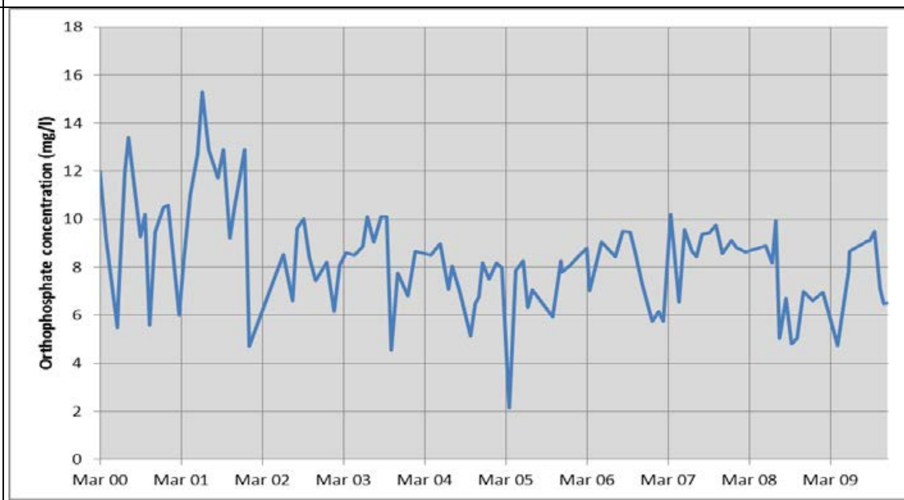
a) Brigstock	
b) Stanion	

Figure 5.5 Orthophosphate concentrations in effluent from STW in Harpers Brook catchment

There are 156 septic tanks in the catchment (from National Population Database data supplied by Chris Burgess, Environment Agency, Jan 2011), as shown in Figure 5.6. These tanks are located an average of 218 m from the river, and 1,236 m from the nearest sewer network. Other small consented discharges are also present.

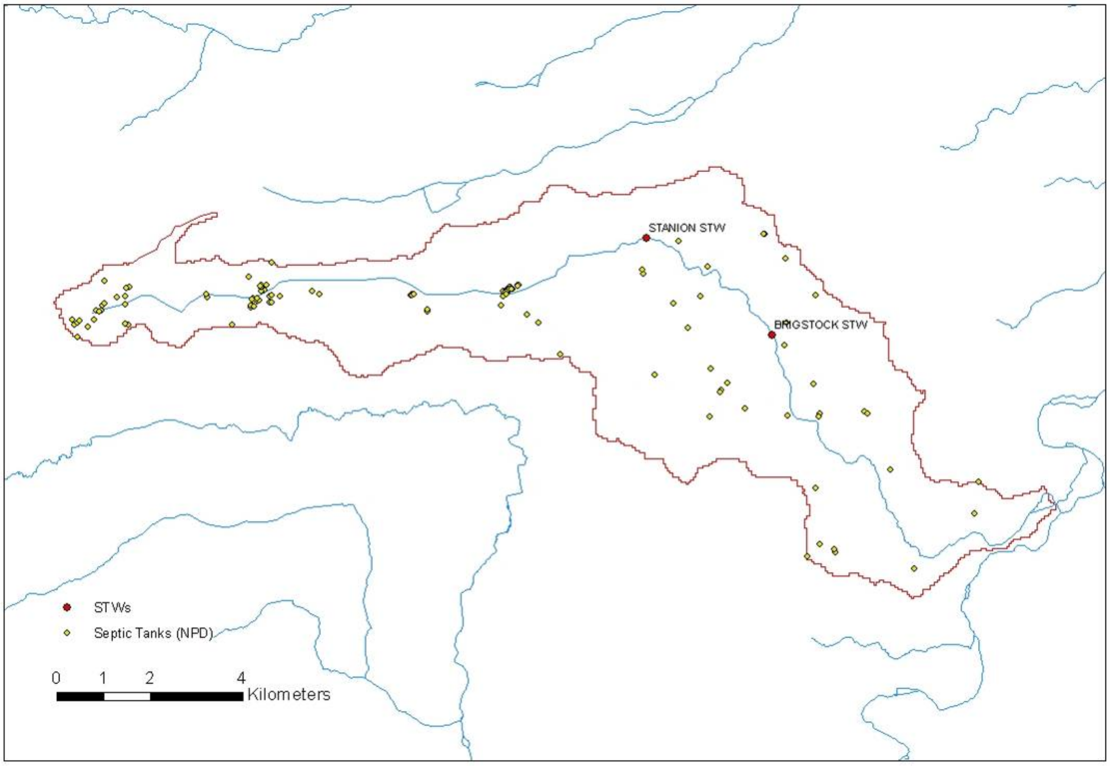


Figure 5.6 STW and septic tanks in Harpers Brook catchment

Based on observations during the catchment walkover, drainflow appears to be the dominant pathway of diffuse P loss in the catchment and drains run turbid during storm events, especially in autumn. Given the high ditch density, farming activities have the potential to influence ambient P concentrations within the river but no information is available on the likely range in P concentrations being discharged from land drains. High flow volumes during storm events do help to flush out any sediment accumulated within the stream channel and will dilute inputs of P from STW and other consented discharges. The urban development at Corby appears to be contributing to the stream.

5.4 Characteristics of Bourn Brook

Figure 5.7 shows the Bourn Brook subcatchment, which is located in the Cam and Ely Ouse catchment, in the central area of Anglian RBD, as shown in Figure 4.2.

Figure 5.7 Map of Bourn Brook pilot catchment with water quality monitoring points

5.4.1 Physical characteristics

The Upper Bourn Brook subcatchment is 8.11 km long, with a catchment area of 32.39 km^2. The Bourn Brook as a whole is longer than Harpers Brook, being 21.02 km long, with a tributary which is an additional 2.95 km, and a larger overall catchment area of 89.46 km^2. The brook flows into the River Cam. Like Harpers Brook, Bourn Brook is a hydrologically discrete headwater body, has high alkalinity and is defined in the Anglian RBMP as small, low and calcareous.

Soils in the catchment are predominantly clay, with some small areas of shallow silt, as shown in Figure 5.8. Information provided by a tenant farmer of the Countryside Restoration Trust (CRT) suggests that there are sand outcrops (estimated to be around a couple of feet deep) on the clay between water quality monitoring points 32M04 and 32M01. South of the brook between monitoring points 32M01 and 32M02, the land is quite light and is described by the farmer as marginal wheat land, that is, it is almost too light for wheat and better suited for barley. Drainage in the clay soils is impeded. No information was available on soil P levels but these are likely to be modest (P Index 1/2) given the soil type (catchment walkover – January 2011). The catchment is underlain by clay bedrock, with some areas of sandstone in the Upper Bourn, and some chalk in the Lower Bourn.

Elevation in the catchment ranges from 60 m AOD in the Upper Bourn to 25 m AOD near the confluence with the River Cam. Slopes in the catchment are moderate, and shallower than in Harpers Brook catchment, as shown in Figure 5.8.

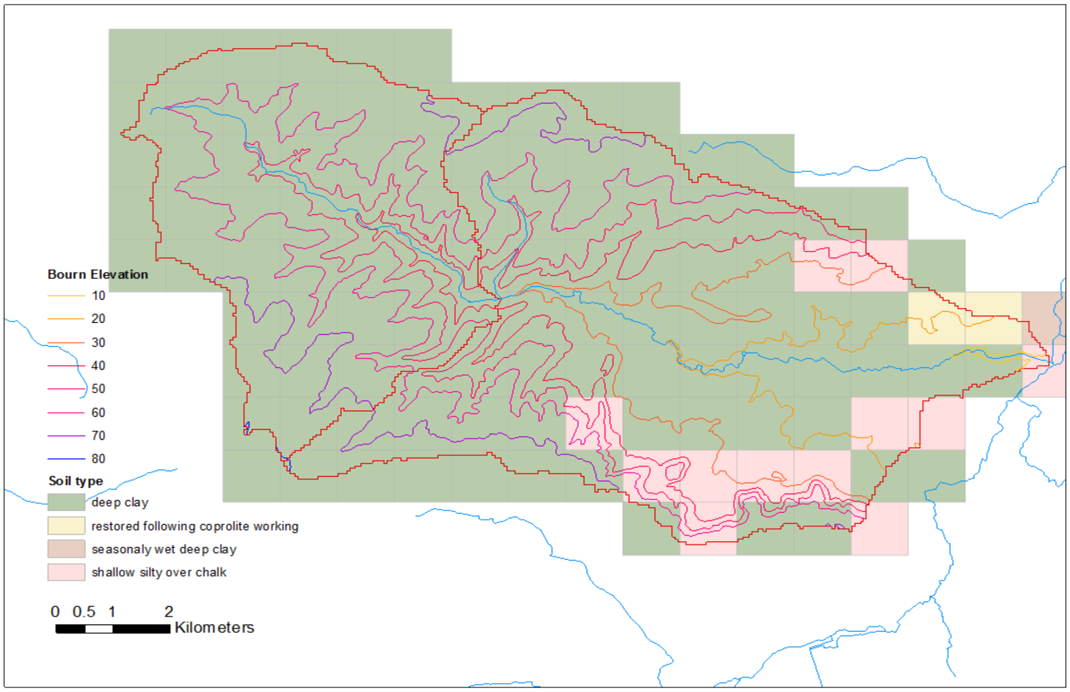


Figure 5.8 Soil type and elevation in Bourn Brook catchment

Long-term average measurements of rainfall show that rainfall patterns in the Bourn Brook catchment are similar to Harpers Brook, with peaks seen in August and November, and the lowest levels in February. July is considered by the local farming community as the worst month for heavy rainfall. Flash floods can be a problem as they occur with surprising rapidity following heavy rain.

Bourn Brook has no flow gauges on the river, but stage height measurements show a similar flow profile to Harpers Brook, with peak flows in December and January, and low flows in July and August. Monthly values for stage height and rainfall are shown in Figure 5.9. The catchment walkovers noted that stream density is high with numerous first order tributaries flowing into the main Bourn Brook, which flooded during the visit following heavy overnight rain. The base flow index (BFI) of the water body is 0.37. The upper catchment includes the new village of Cambourne which has *potentially* increased the flashiness of the brook's response in flow to rainfall. The influence of the village of Cambourne on flow will be investigated further during Phase 2 of the project.

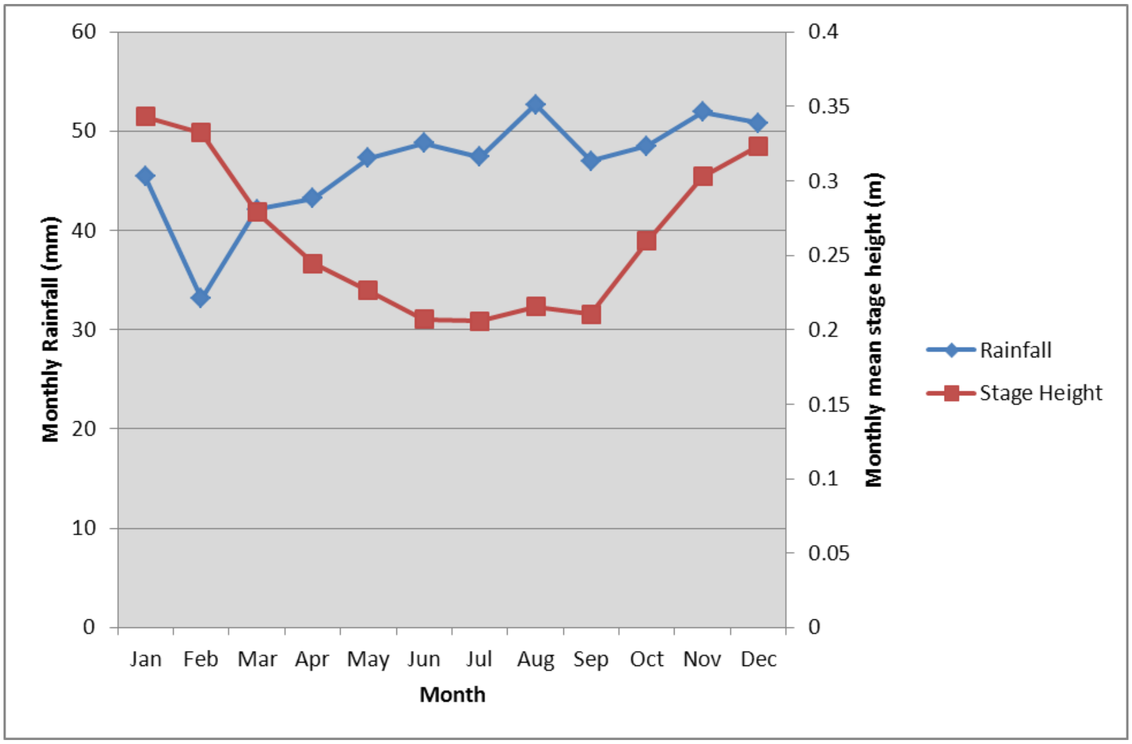


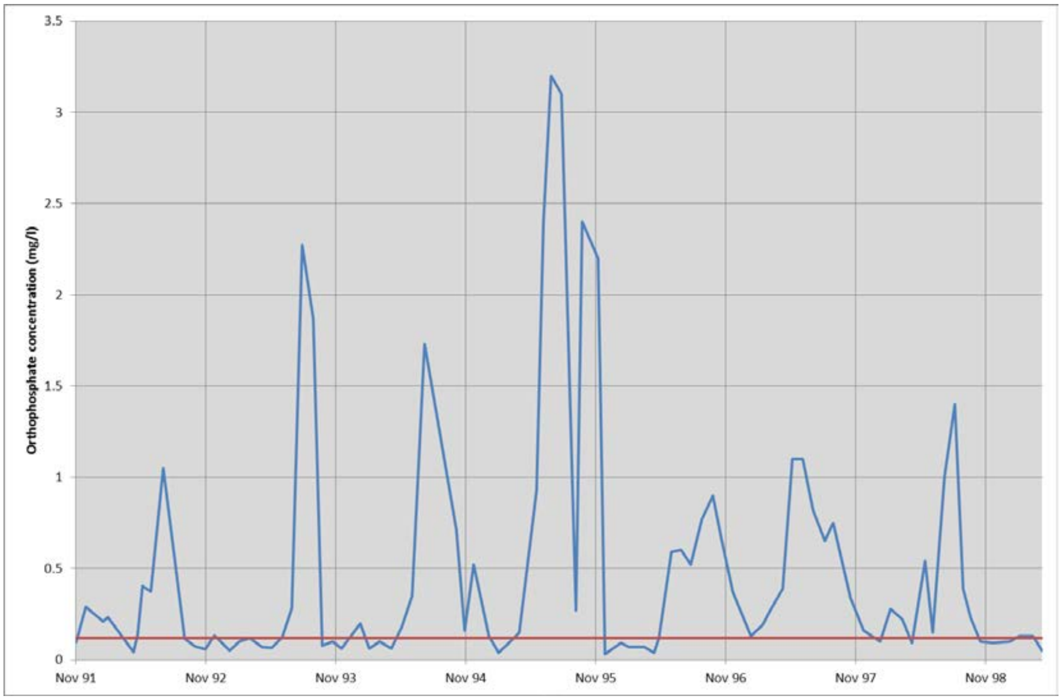
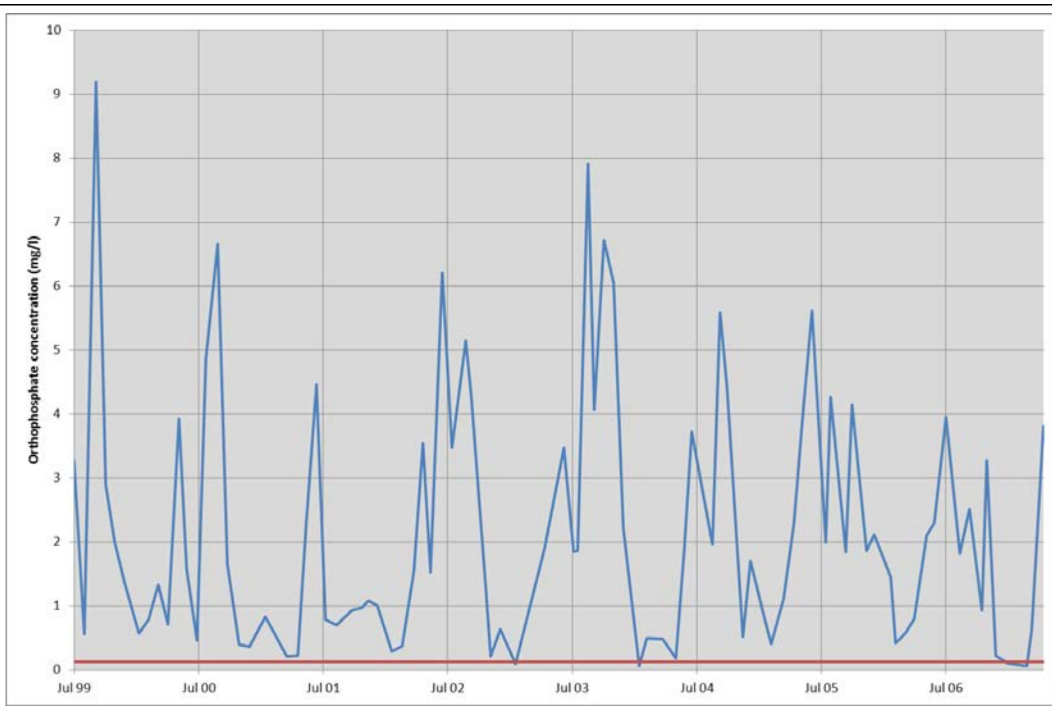
Figure 5.9 Monthly average stage height and rainfall values in Bourn Brook catchment

Biological quality has been assessed as 'good' (macroinvertebrates: good, fish: good) as part of the RBMP. Physico-chemical status is 'high' for ammonia, pH, temperature, zinc and copper, and 'good' for dissolved oxygen, but is 'poor' for phosphorus. The hydromorphology of the catchment is 'supporting good', while chemical status has not been assessed.

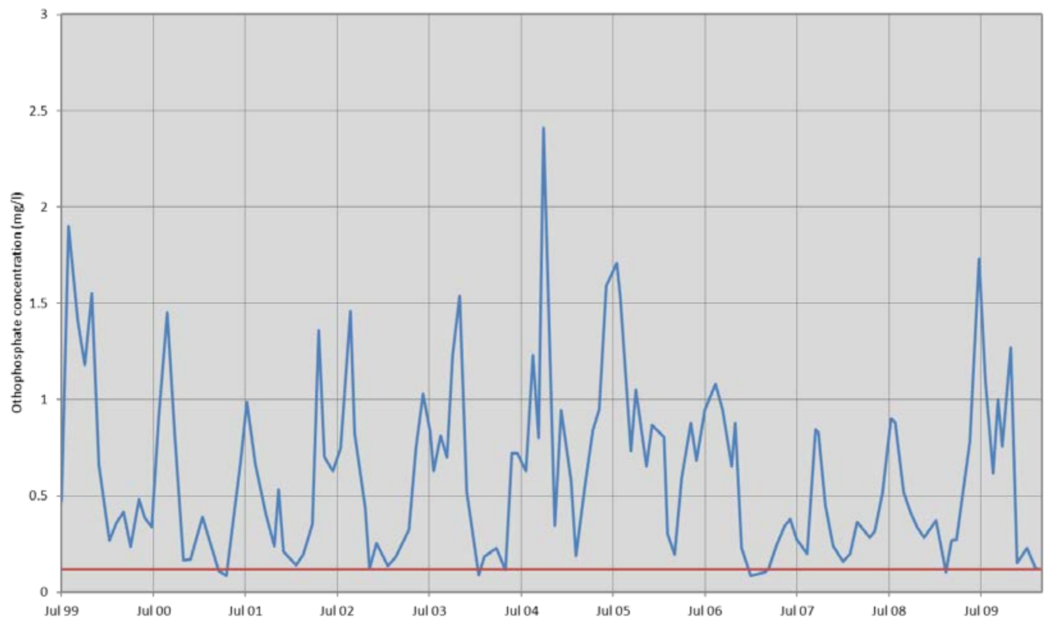
There are four water quality monitoring sites in the catchment, including one in the Upper Bourn catchment, as shown in Figure 5.7. No monitoring has been undertaken of macrophytes and diatoms. Measured orthophosphate levels are shown in Table 5.4 and Figure 5.10. Sampling point 32M05, upstream of the confluence with the Bourn STW tributary, was last sampled in 1999 and showed a seasonal pattern similar to that dominated by a point source discharge, but further investigation is needed to accurately identify phosphorus sources. At all monitoring stations, water quality clearly fails to achieve the WFD P standards of 0.050 mg SRP l^{-1} (high) and 0.120 mg SRP l^{-1} (good).

Table 5.4 Levels of orthophosphate measured in the Bourn Brook catchment (sites shown in Figure 5.7)

Site	Catchment	Year	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
32M05	Upper Bourn	1991-1999	0.50	2.35	3.20
32M04	Lower Bourn	1988-2007	3.32	9.19	13.70
32M02	Lower Bourn	1999-2010	0.62	1.54	2.41
32M01	Lower Bourn	1988-2004	2.12	6.81	13.70

a) 32M05	
b) 32M04	

c)
32M02



d)
32M01

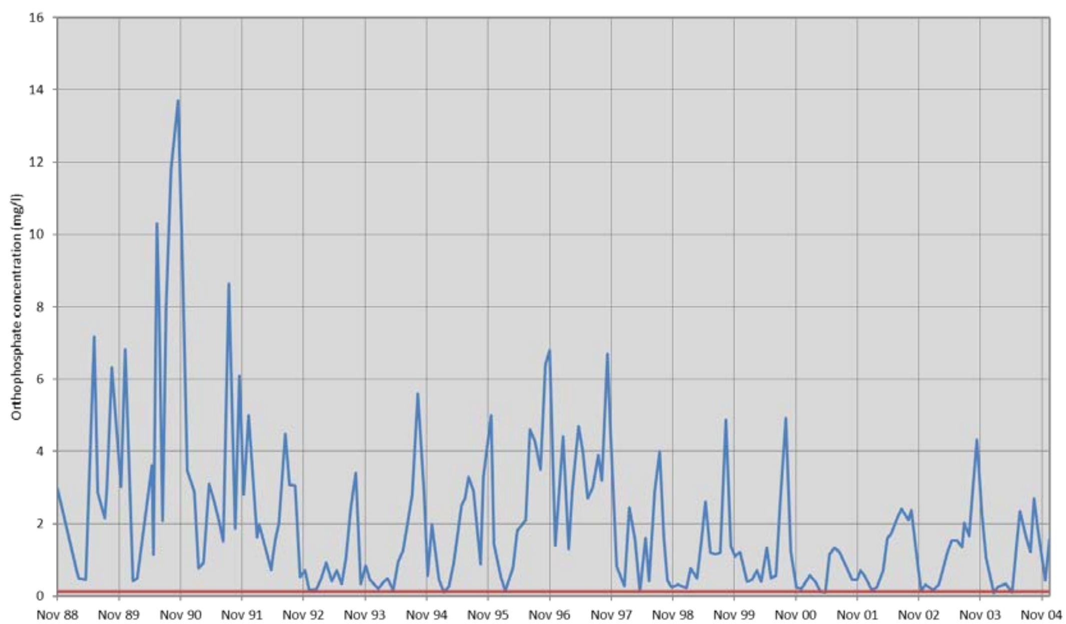


Figure 5.10 Orthophosphate concentrations at monitoring sites in Bourn Brook catchment (red lines represent the WFD P standard)

The River Habitat Survey carried out on Bourn Brook (2007-2008) showed a small amount of channel modification in the form of resectioning. The level of tree cover was generally higher than on Harpers Brook, being semi-continuous in most places. Himalayan Balsam was present, as was Hogweed, but no Knotweed was identified. Overall, the river was seen as being active, with some channel modifications, along with natural erosion of the river banks and areas of poaching by deer.

5.4.2 Human characteristics

Phosphorus levels in the river are rated as ‘poor’, indicating that they are significantly below the required WFD standard. A third of the phosphorus input to this catchment is estimated to come from agricultural sources. Like Harpers Brook, Bourn Brook is designated as a heavily modified water body for reasons of flood protection. The catchment is designated as a protected area under NVZ and freshwater fish directives. The Upper Bourn Brook is designated as a NVZ for both surface water and groundwaters, while the lower catchment is only designated for surface water in most parts.

Land use in the Upper Bourn catchment has historically been dominated by arable production, mostly cereal crops, set-aside land and permanent grass, with some small areas of other crops such as beans and potatoes. Land use information from the 2004 agricultural census is shown in Table 5.5. Little livestock farming is present, but there are some pig farms in the upper part of the catchment.

Table 5.5 Land use in Bourn Brook catchment (EDINA Ag Census 2004)

Type	Upper Bourn Brook	Bourn Brook	Units
Temporary grass	0.73	0.63	ha km^{-2}
Rough	0.11	0.68	ha km^{-2}
Other arable	0.05	1.03	ha km^{-2}
Fallow	0.04	0.09	ha km^{-2}
Crops & bare fallow	66.42	63.55	ha km^{-2}
Woodland	2.17	1.92	ha km^{-2}
Set aside	9.07	7.39	ha km^{-2}
All other land	3.07	2.37	ha km^{-2}
Horticulture	0.04	0.10	ha km^{-2}
Fruit	0.01	0.04	ha km^{-2}
Permanent grass	5.84	7.53	ha km^{-2}
Total cereals	48.04	43.06	ha km^{-2}
Potatoes	0.04	0.16	ha km^{-2}
Dairy	0.00	0.01	head km^{-2}
Beef	1.29	1.04	head km^{-2}
Cattle	6.98	5.56	head km^{-2}
Sheep	10.57	7.75	head km^{-2}
Goats	0.16	0.14	head km^{-2}
Pigs	69.29	34.30	head km^{-2}
Poultry	4.96	57.71	head km^{-2}
Ducks	0.18	0.19	head km^{-2}
Geese	0.30	0.24	head km^{-2}

The catchment walkovers noted that the majority of fields were minimally cultivated, tramlines occurred both across slope and down slope and six-metre riparian strips were fairly ubiquitous. Field margins adjacent to the brook collected standing water during wet weather but generally there was no evidence that this was flowing into the streams. One notable exception in the lower catchment area was where riparian fields were cultivated up to the river margins and the two-metre cross-compliance buffer strip had largely disintegrated in the field corner. A large area of this field was flooded on day two of the visit. Surface runoff inputs into the brook were therefore largely confined to overflow from rural tracks and roads, but these did not have a high density due the very large fields.

Although soil P levels are likely to be low, one farmer in the lower catchment interviewed during the catchment walkovers had applied biosolid (sewage sludge) in the past and one farmer in the upper catchment area had had pigs for a number of years. Discussions suggested that most P fertiliser was incorporated before drilling rather than top-dressed.

The brook flows directly through the villages of Caxton and Bourn in the upper catchment. Numerous 'pipes' in the villages discharge directly to the river. It was not clear where these originate but are likely to be yard or roof runoff or greywater discharges. Runoff from the road network in the villages is clearly a major input to the brook. There are two fords in the upper catchment area which were both heavily flooded during the visit. These have the potential to contribute sediment-bound phosphorus originating from the local roads.

The town of Cambourne is a rapidly expanding new settlement within the upper catchment. Building started in 1998 for a self-contained community of at least 3,300 houses, with approximately 10,000 residents. The three villages have been built on a green field site that was used for farming. There is also a Business Park, and around 10,000 people are expected to work in Cambourne overall. Cambourne predominantly drains away from the Bourn Brook, towards Barhill, but some land drains towards the brook. The influence of the Cambourne development on water quality and flow will be investigated further during catchment walkovers.

There is one small STW (Bourn STW) located in the Lower Bourn catchment, on the tributary to the main river, as shown in Figure 5.12.The STW has a population equivalent value of 3,400 and there are no P removal technologies in place. Orthophosphate levels in the STW effluent are shown in Table 5.6 and Figure 5.11.

Table 5.6 Effluent orthophosphate levels from Bourn STW

Site	Year	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
Bourn STW	1990-2008	6.52	13.43	15.60

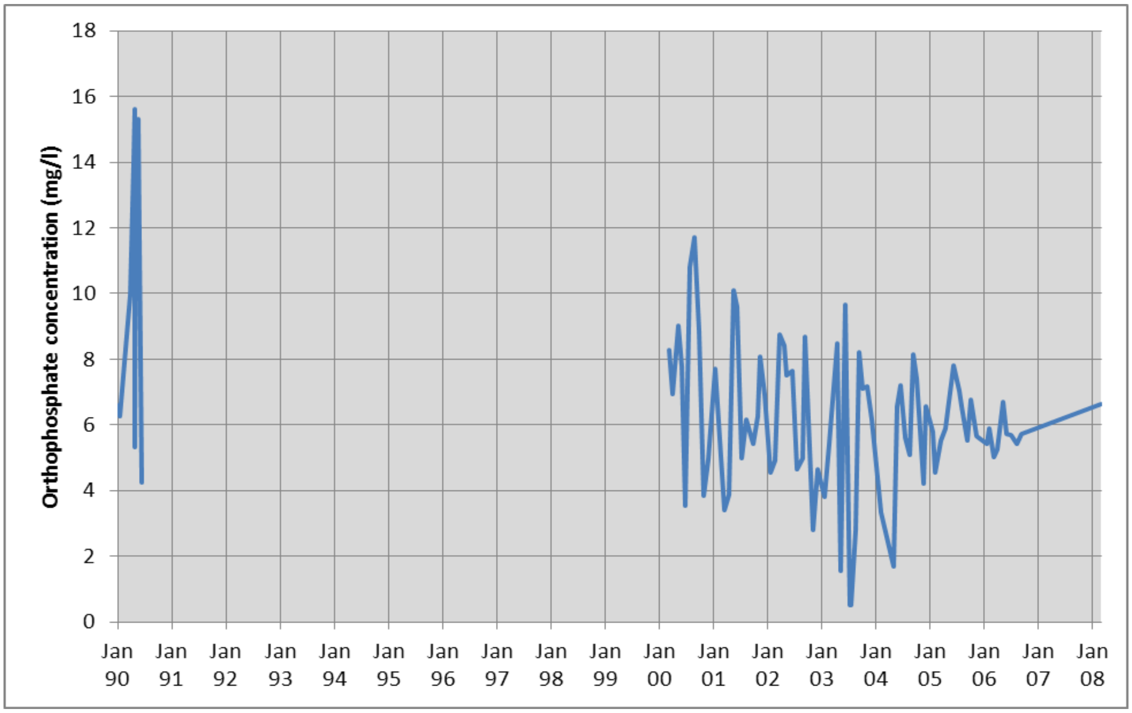


Figure 5.11 Orthophosphate concentrations in effluent from Bourn STW

There are 79 septic tanks in the Bourn Brook catchment, of which 46 are located in the Upper Bourn (from National Population Database data supplied by Chris Burgess, January 2011). The locations of these septic tanks are shown in Figure 5.12. The average distance between the septic tanks and the river is 182 m, and the average distance to the sewer network is 856 m, both of which are lower than in Harpers Brook.

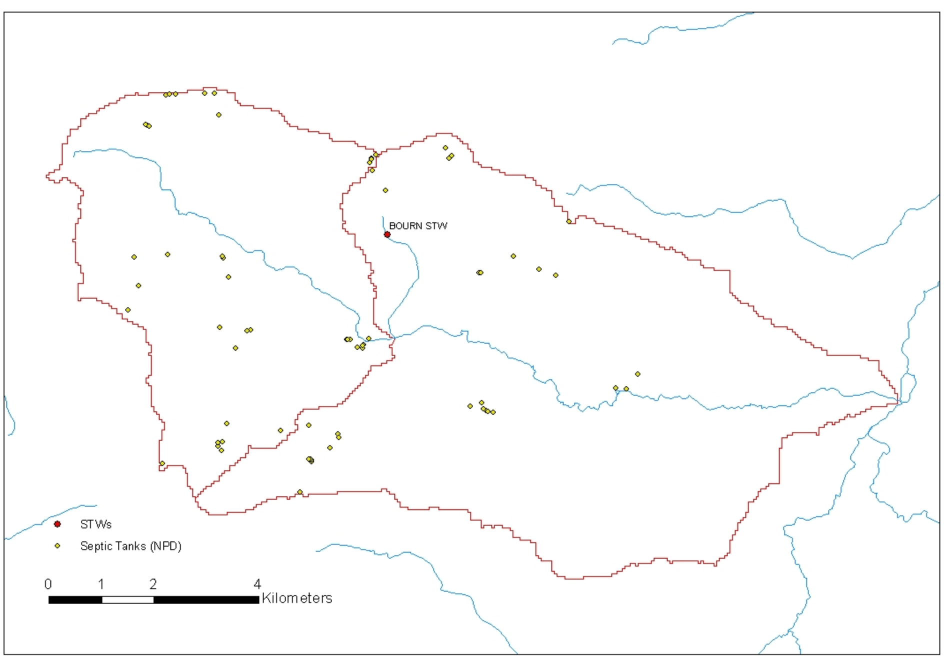


Figure 5.12 Bourn STW and septic tanks in Bourn Brook catchment

The catchment walkovers noted that sub-surface drainflow is the dominant pathway of P loss from farmed land and land drains run turbid during heavy rain, especially in autumn. One pig farm, which was very close to the brook, was using a low rate irrigation system to recycle pig slurry. This practice was considered to be high-risk due to the dilute nature of pig slurry and the presence of cracking and underdrained clay soils.

Overall, agriculture is not considered to be the main source of P to the brook but there are small areas of high-risk farming activity which probably contribute additional P to the general background sediment-associated P in drainflow during the winter drainage period. Recycling of pig slurry to cracked soils during summer/autumn may also cause localised increases in stream P concentrations.

5.5 Source and exposure pressure

There are a range of potential sources of P in the pilot catchments, which include point sources like STW, septic tanks, industrial discharges and runoff from agricultural areas of hard standing, and diffuse sources, such as runoff from farmland, roads or urban areas.

The PSYCHIC model developed by ADAS give estimates of P loss from different sources based on land use, rainfall and soil type, among other factors. The results of this model are used in Sections 5.5.1 and 5.5.2 below to show the variation between different sources.

The National SIMCAT model (WRc, 2006) has also been used to calculate estimates of the P contribution from STW in the catchment compared with the overall P load.

5.5.1 Harpers Brook

Sources of P

The maps below show modelled P inputs to rivers on a 1 km x 1 km grid scale taken from the results of the PSYCHIC models. As shown in Figure 5.13, P losses in the catchment are highest in central areas, reaching up to 0.75 kg ha^{-1} $a^{-1.}$ Diffuse P losses are highest in the upper and central areas of the catchment, but generally contribute less than 0.1 kg ha^{-1} a^{-1} (Figure 5.14). Losses from point sources peak at over 0.75 kg ha^{-1} a^{-1}, with the highest levels found around the STW at Stanion and at Brigstock (Figure 5.15).

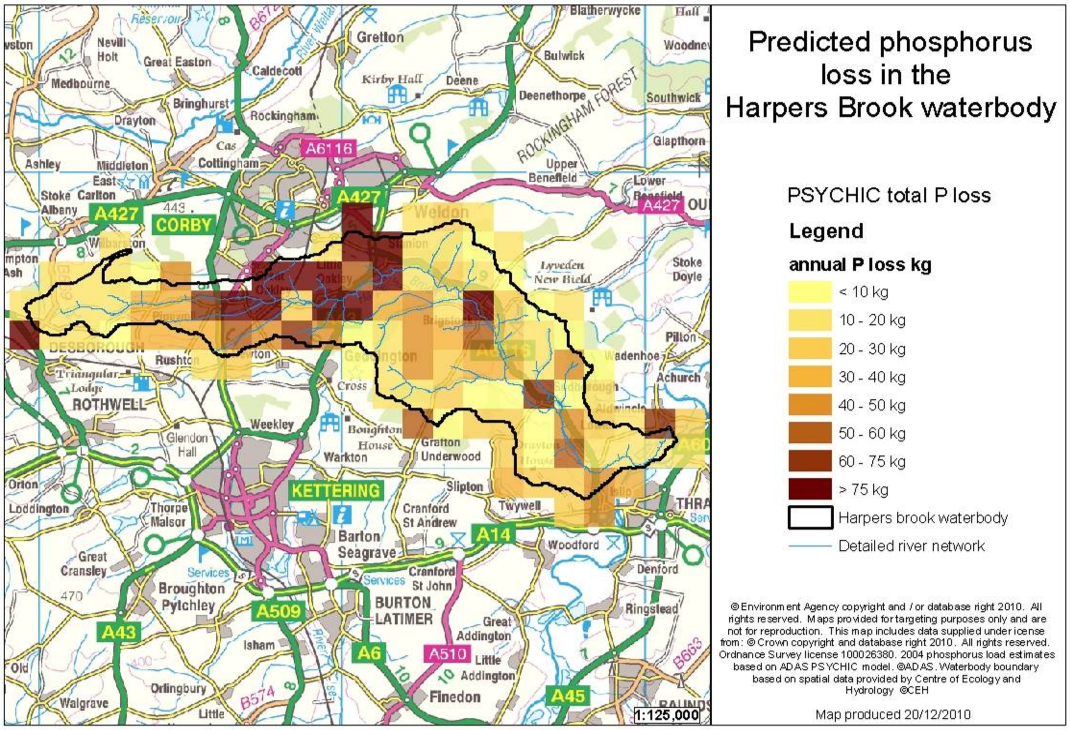


Figure 5.13 P losses from all sources in Harpers Brook catchment

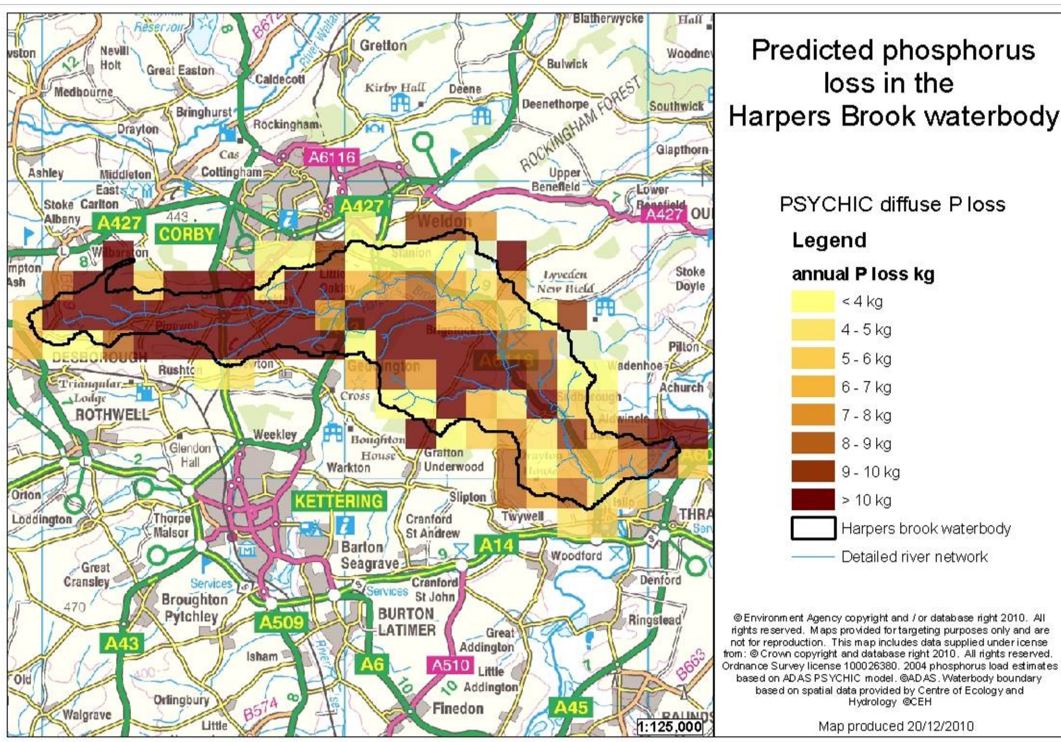


Figure 5.14 P losses from diffuse sources in Harpers Brook catchment

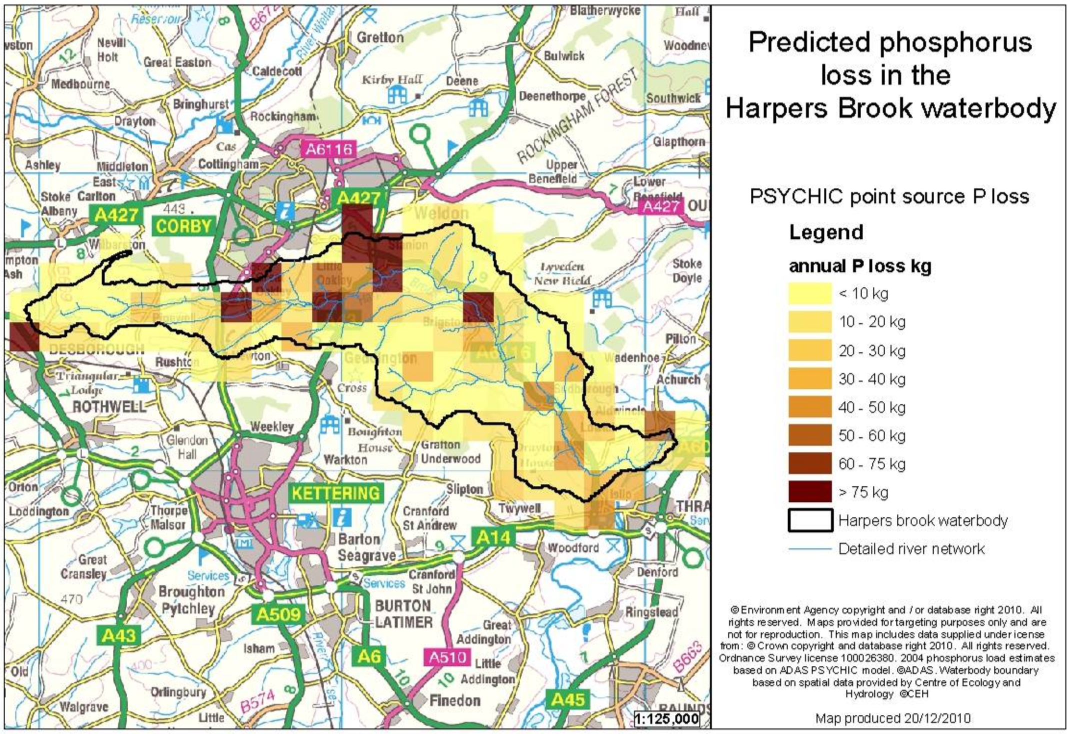


Figure 5.15 P losses from point sources in Harpers Brook catchment

Calculations made using the results of the National SIMCAT model (WRc, 2006) suggest that Stanion STW contributes around 60 per cent of the total P load at monitoring point HARP140B, and the two STW contribute 55 per cent of the P load at monitoring site HARP200A.

Agricultural losses of P come from three main sources: areas of hard standing (point agricultural sources of P), and manure and fertilisers (diffuse agricultural sources of P). In this catchment, areas of hard standing contribute less than 0.03 kg ha^{-1} a^{-1} in most areas, almost all of which comes from beef cattle (Figure 5.16). Contributions of P from fertilisers and manures peak at around 0.035 kg ha^{-1} a^{-1}, with losses from fertiliser mainly in the upper part of the catchment (Figure 5.17), while P from manure is spread more evenly across the whole catchment (Figure 5.18).

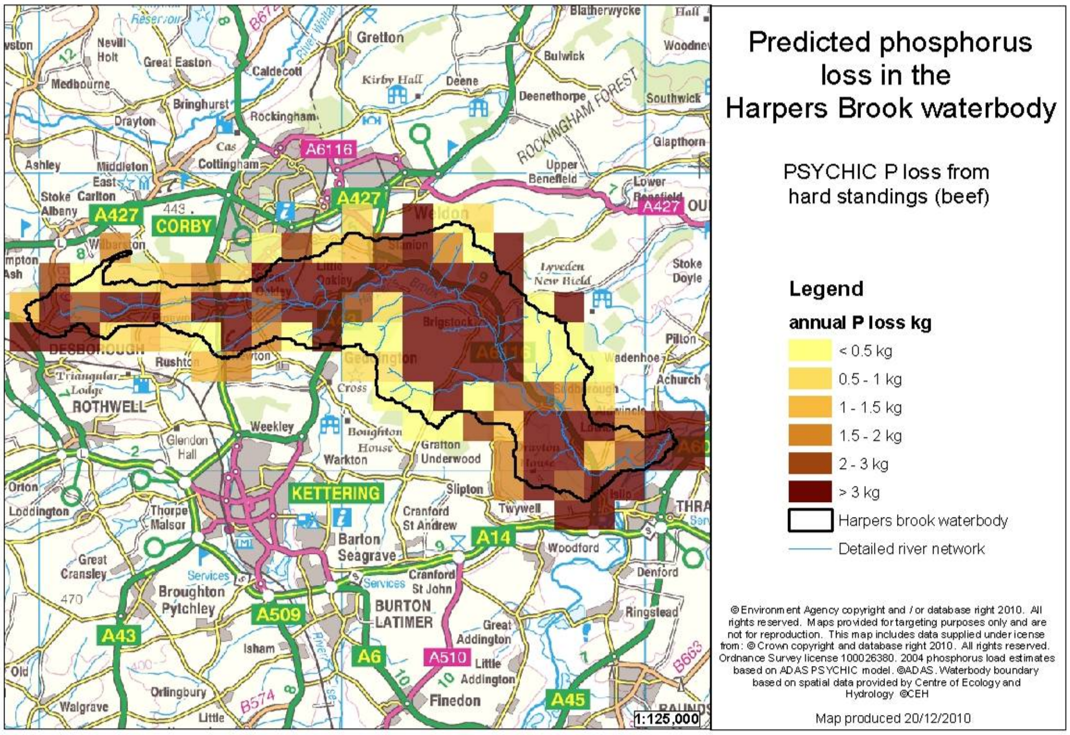


Figure 5.16 P losses from areas of hard standing in Harpers Brook catchment

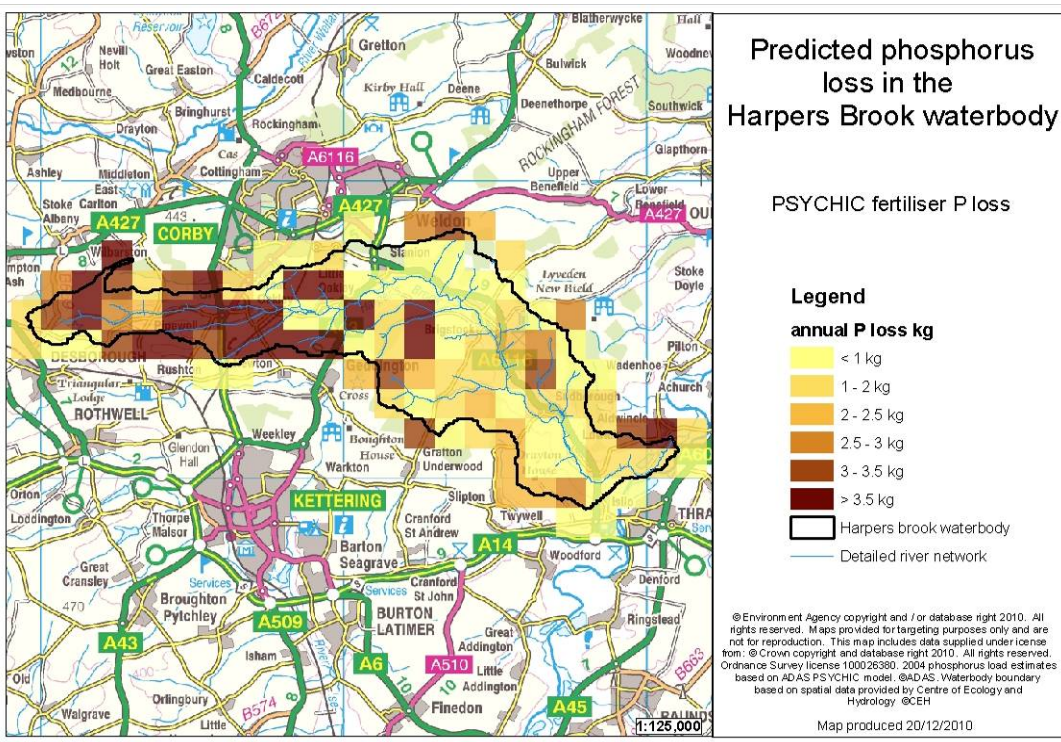


Figure 5.17 P losses from fertilisers in Harpers Brook catchment

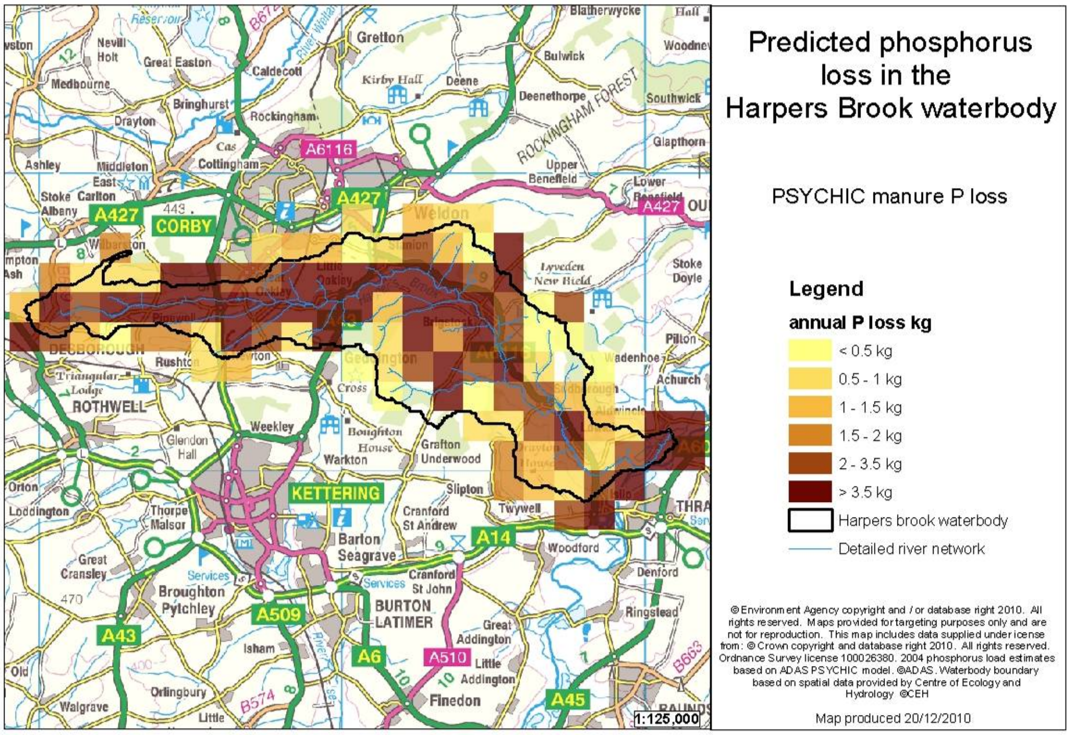
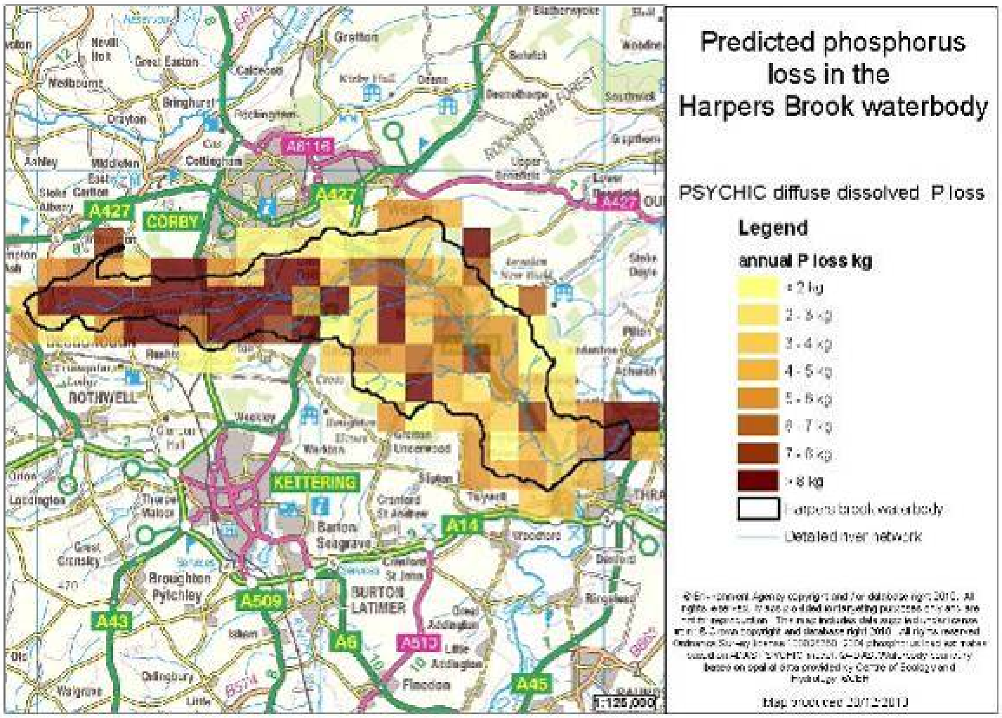


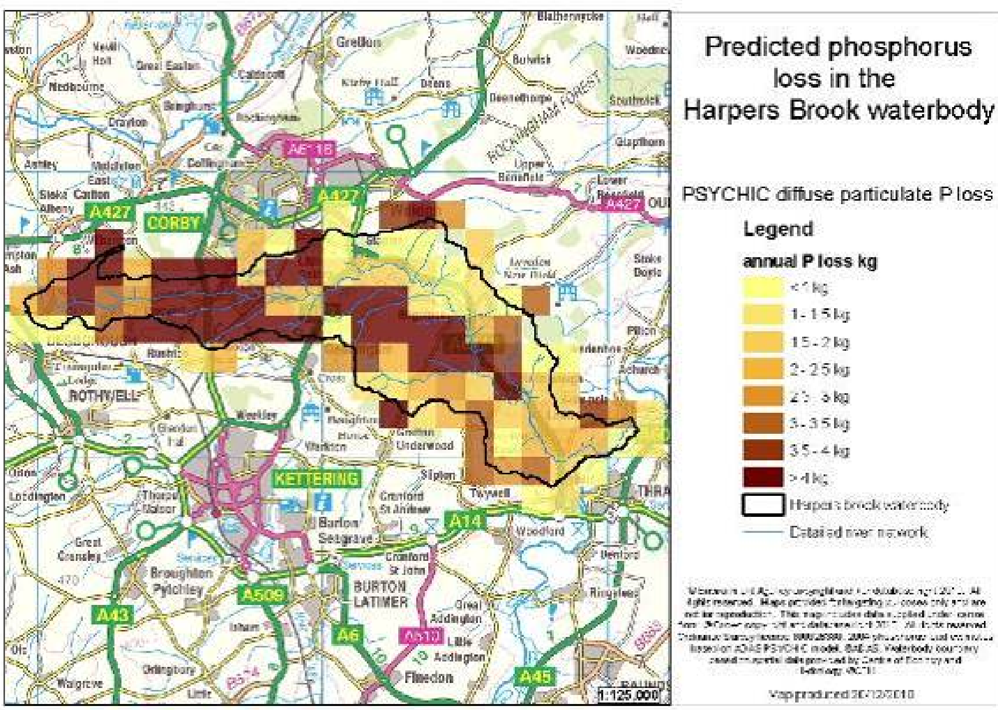
Figure 5.18 P losses from manure in Harpers Brook catchment

Forms of P

The PSYCHIC models also provide information on the form in which P is mobilised, splitting diffuse P losses into dissolved and particulate forms. Overall, concentrations of dissolved P in the catchment are around double the concentrations of particulate P, peaking at 0.08 kg ha^{-1} a^{-1}, as opposed to 0.04 kg ha^{-1} a^{-1}for particulate forms. Both forms show the highest losses in the upper part of the catchment, but concentrations of particulate P are also high in the central part of the catchment, as shown in Figure 5.19.



a) Dissolved P



b) Particulate P

Figure 5.19 Dissolved and particulate forms of diffuse P in Harpers Brook

Agricultural P in groundwater

Agricultural P can reach groundwater through two main pathways: leaching from field soils, and leaching from slurry lagoons and manure heaps. Other, non-agricultural sources of P include septic tanks and leaking sewers and water mains.

There are no groundwater monitoring sites within the Harpers Brook catchment. However, two groundwater springs are monitored on the same groundwater bodies as the catchment. As shown by Table 5.7, average P concentrations for one of these springs is low at 0.027 mg P l^{-1} (WELDON01), but significantly higher at 0.089 mg P l^{-1} at spring ID SP852587. Given the WFD phosphorus targets for this water body are 0.050 mg P l^{-1} (good) and 0.120 mg P l^{-1} (high), the groundwater contribution is likely to be significant to the surface water failures.

Table 5.7 Orthophosphate concentrations at groundwater monitoring points in Harpers Brook catchment

ID	Type	Years	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
WELDON01	Spring	1987-2010	0.027	0.036	0.037
SP852587	Spring	2005-2010	0.089	0.113	0.125

5.5.2 Bourn Brook

Sources of P

Similarly to Harpers Brook, total phosphorus losses in Bourn Brook catchment peak at over 0.75 kg ha^{-1} a^{-1}, with the areas of highest P spread across the catchment, as shown in Figure 5.20. There are fewer areas of very high P loss in the Upper Bourn Brook than in the lower part. Diffuse P losses are spread fairly evenly across the whole catchment, generally contributing only 0.08-0.09 kg ha^{-1} a^{-1} (Figure 5.21), while losses from point sources show isolated peaks at over 0.75 kg ha^{-1} a^{-1} (Figure 5.22). As in Harpers Brook, high concentrations are found around the site of the STW, but a number of other areas have high point sources of P.

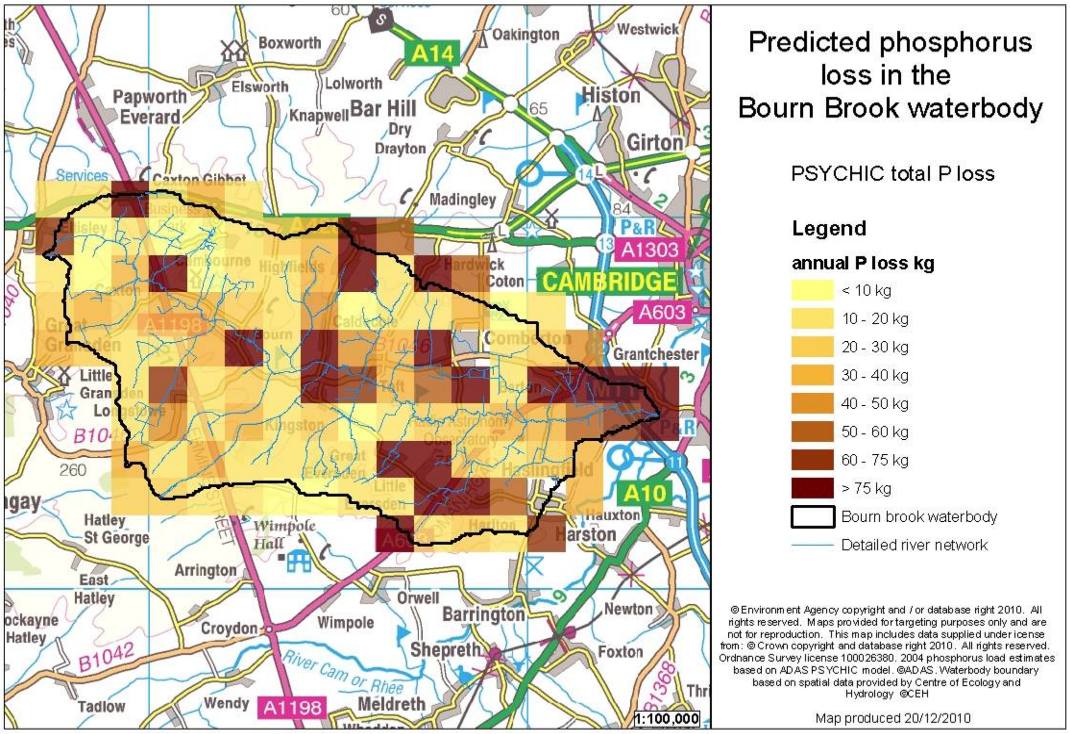


Figure 5.20 P losses from all sources in Bourn Brook catchment

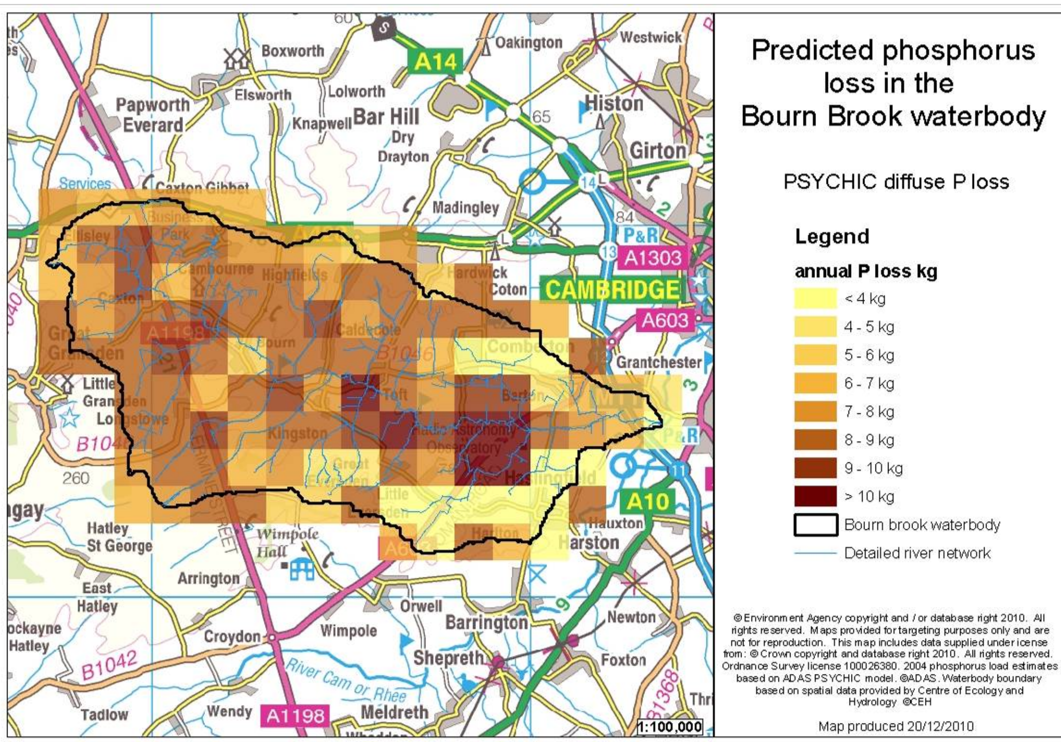


Figure 5.21 P losses from diffuse sources in Bourn Brook catchment

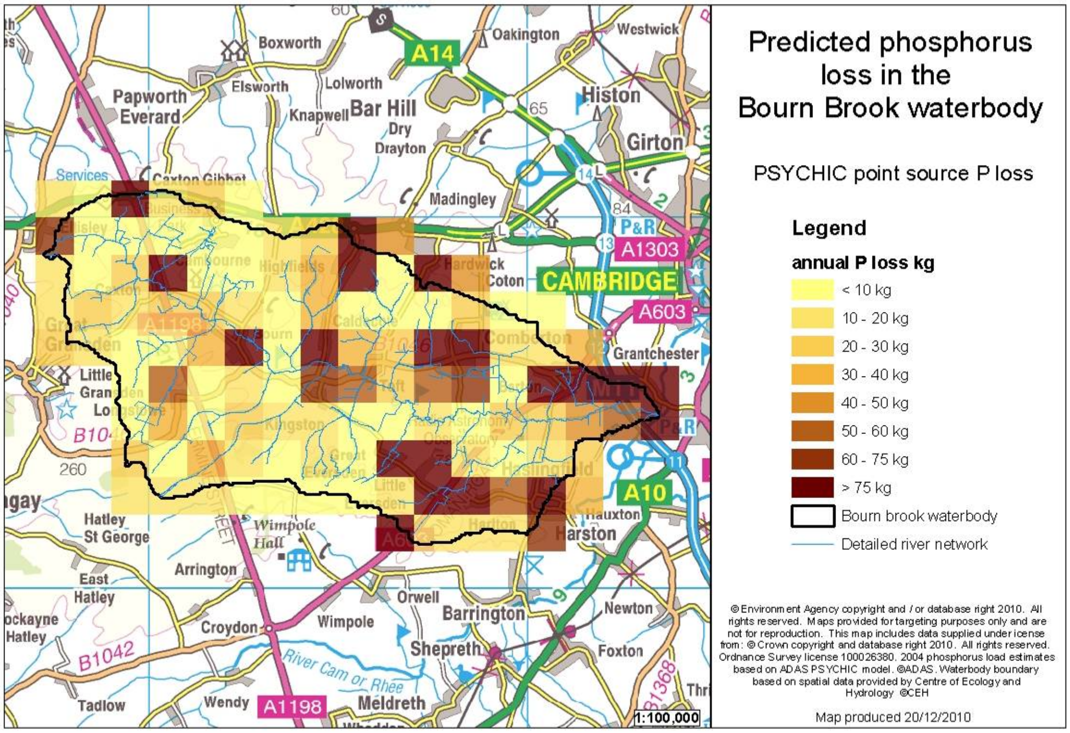


Figure 5.22 P losses from point sources in Bourn Brook catchment

Calculations made from the National SIMCAT model (WRc, 2006) suggest that Bourn STW contributes around 22 per cent of the total P load at its closest monitoring point, 32M04.

Like Harpers Brook, areas of hard standing in the catchment generally contribute less than 0.03 kg ha^{-1} a^{-1}, and almost all of this comes from beef cattle (Figure 5.23). Most areas with the highest contributions from hard standing are located in the upper part of the catchment. Contributions of P from fertilisers are generally less than 0.035 kg ha^{-1} a^{-1}, and are spread evenly across the catchment (Figure 5.24). No data was supplied on P losses from manures in the Bourn Brook catchment.

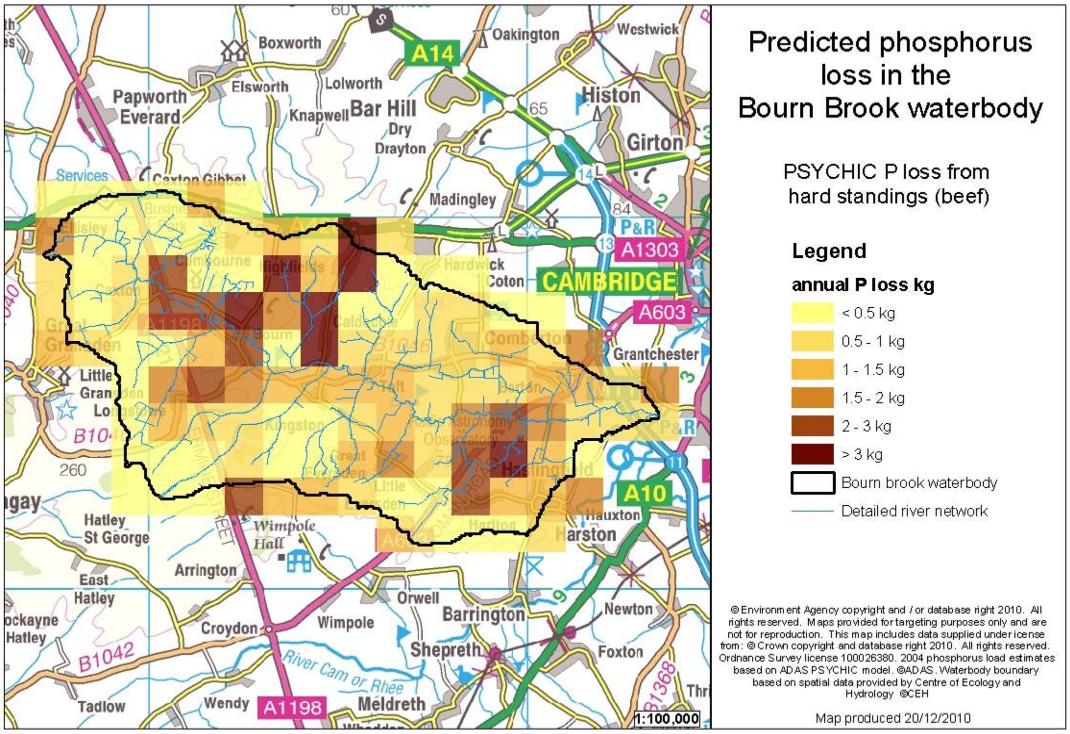


Figure 5.23 P losses from areas of hard standing in Bourn Brook catchment

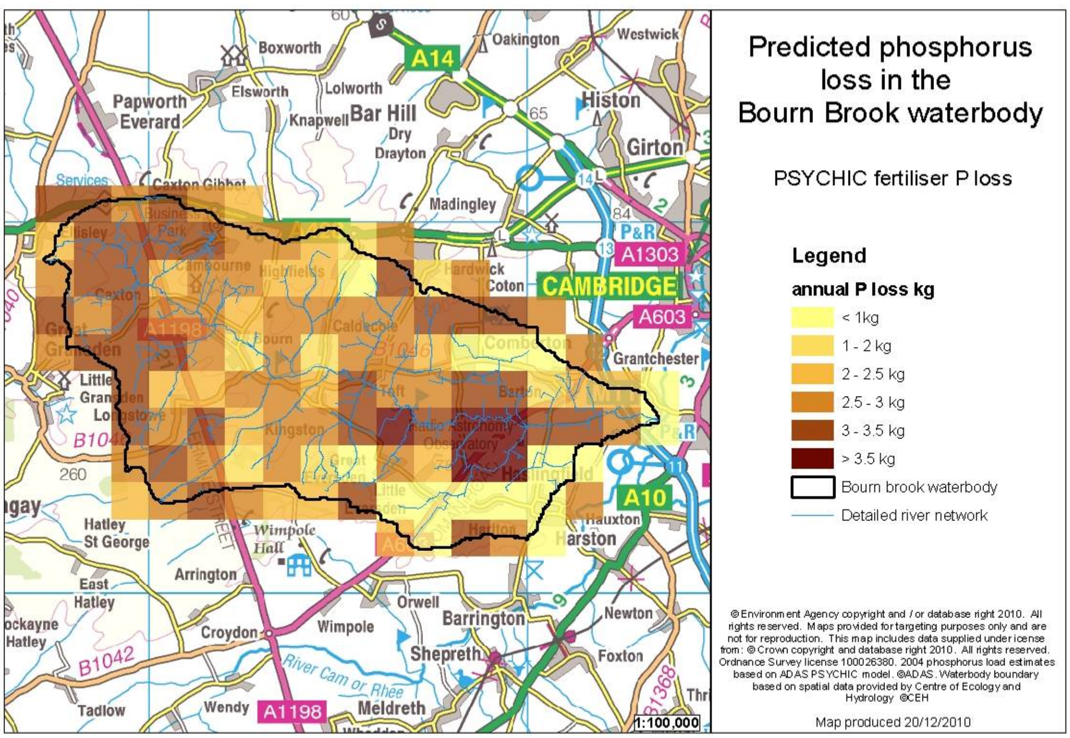
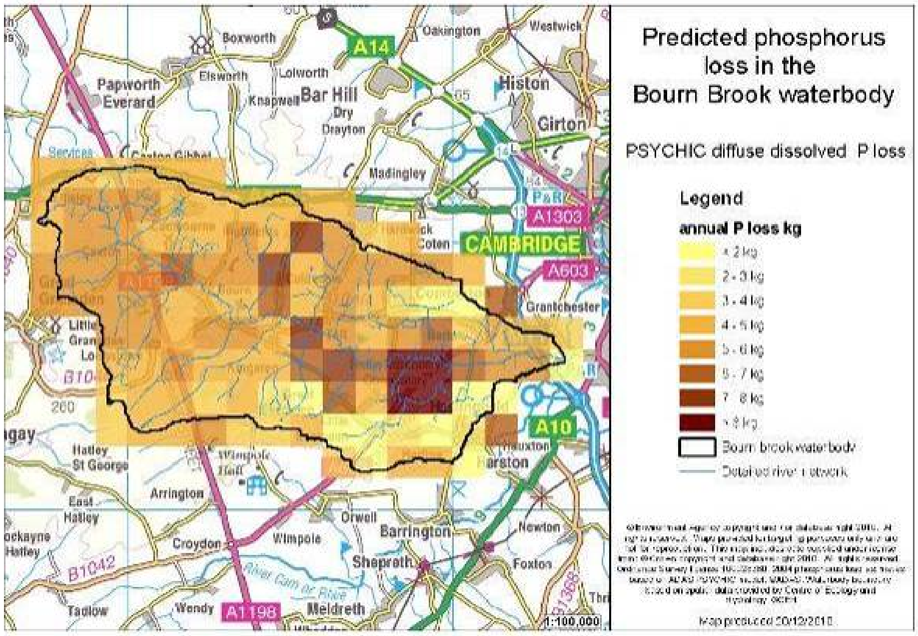


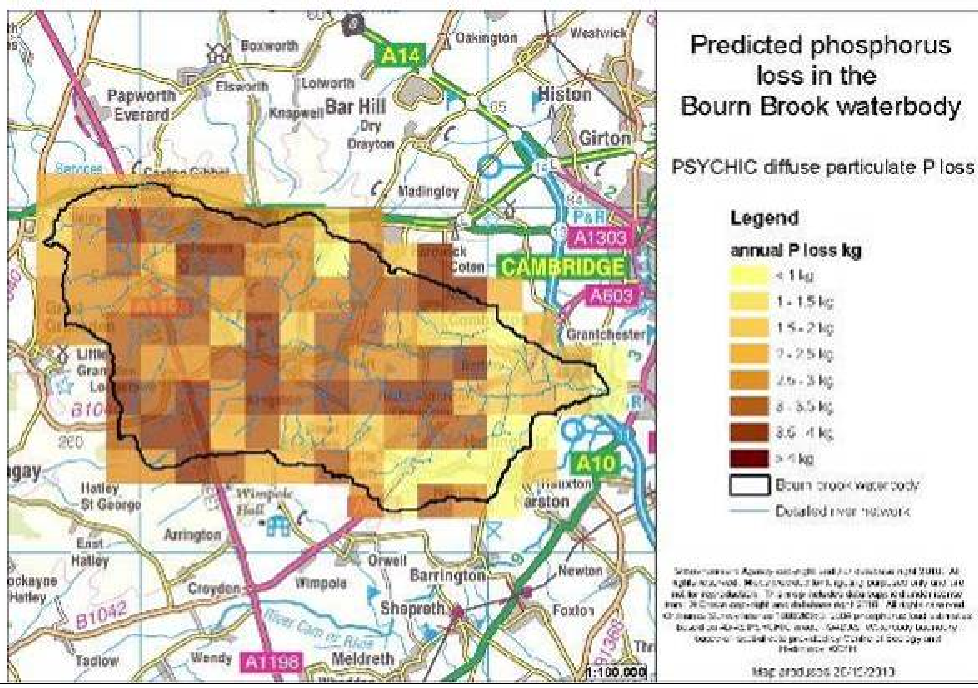
Figure 5.24 P losses from fertiliser in Bourn Brook catchment

Forms of P

As in Harpers Brook, concentrations of dissolved P in the catchment are around double the concentrations of particulate P, peaking at 0.08 kg ha^{-1} a^{-1} and 0.04 kg ha^{-1} a^{-1}respectively. Particulate forms of P are variable across the catchment, with the lowest levels at the downstream end, while dissolved forms are lower in the upper part of the catchment, at around 0.05 kg ha^{-1} a^{-1}, with a small area of very high concentrations in the lower catchment (Figure 5.25).



a) Dissolved P



b) Particulate P

Figure 5.25 Dissolved and particulate forms of diffuse P in Bourn Brook

Agricultural P in groundwater

Two groundwater monitoring boreholes are located in the Bourn Brook catchment, one of which (TL35075S) is located in the upper Bourn. However, as much of the catchment is underlain by clay geology, only a limited amount of water (and subsequently P) may be moving from the surface to the groundwater below the clay layer. Table 5.8 shows P concentrations in the two boreholes, which are low (0.020 mg l^{-1} is the detection limit, so true average values are likely to be much lower than this).

Holman *et al.* (2010) shows very high concentrations of P in the Woburn Sands aquifer, which underlies part of the Upper Bourn Brook catchment. In this area, agricultural or urban sources of P may be having a more significant impact on groundwater concentrations. The boreholes shown in Table 5.8 do not fall within this aquifer, so an accurate estimate of P concentrations cannot be obtained.

Table 5.8 Groundwater orthophosphate concentrations in the Bourn Brook catchment

ID	Type	Years	Mean (mg l^{-1})	95 percentile (mg l^{-1})	Maximum (mg l^{-1})
TL35075S	Borehole	2006-2010	0.020	0.020	0.020
TL35067S	Borehole	2005-2009	0.031	0.075	0.105

5.6 Water quality and ecological impacts

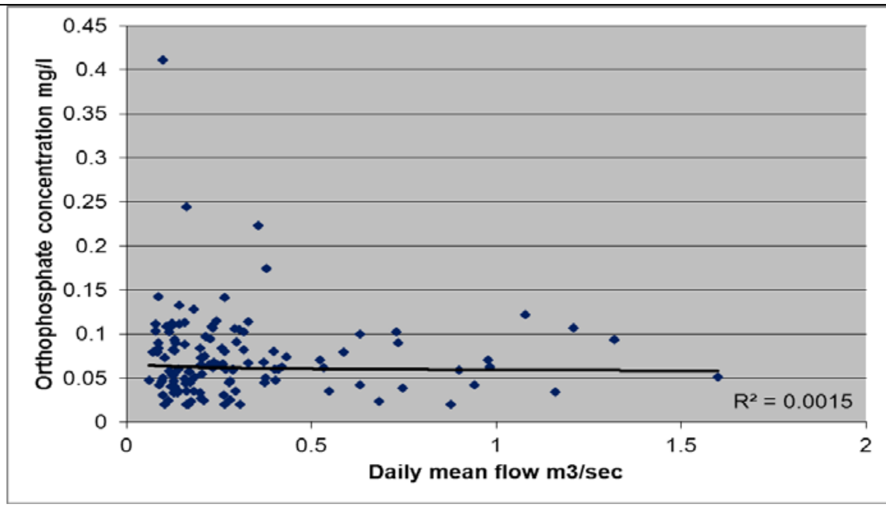
5.6.1 Harpers Brook

WFD status in Harpers Brook catchment is ‘moderate’ for phosphorus, falling only marginally below the ‘good’ target standard. Summary statistics for water quality can be found in Table 5.1 and seasonal variations can be seen in Figure 5.4.

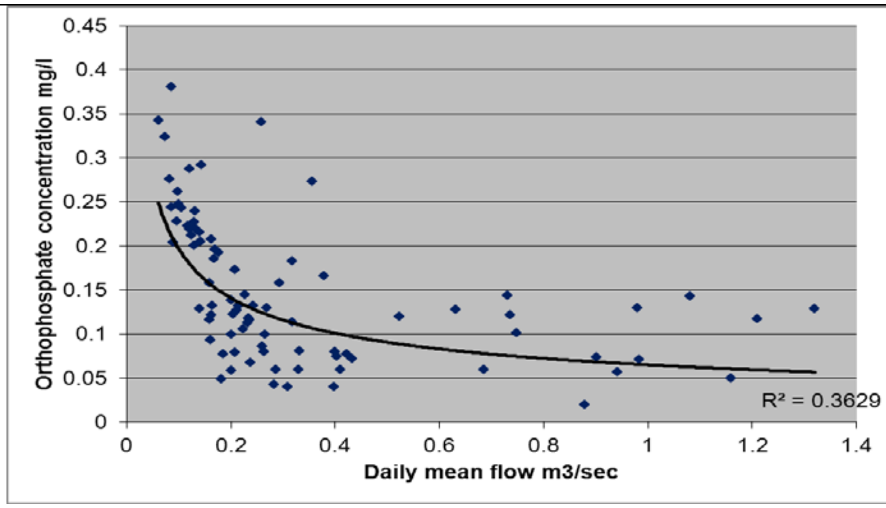
There has been little change in patterns of water quality over the monitoring period. The two sites downstream of the STW, HARP140B and HARP200A, show some seasonal variability, with the lowest concentrations seen in winter and the highest in summer. HARP 100L shows less variability and no consistent seasonal trend.

Figure 5.26 shows concentrations plotted against flow levels measured at the bottom of the catchment, close to monitoring site HARP200A.

a) HARP100L



b) HARP140B



c) HARP200A

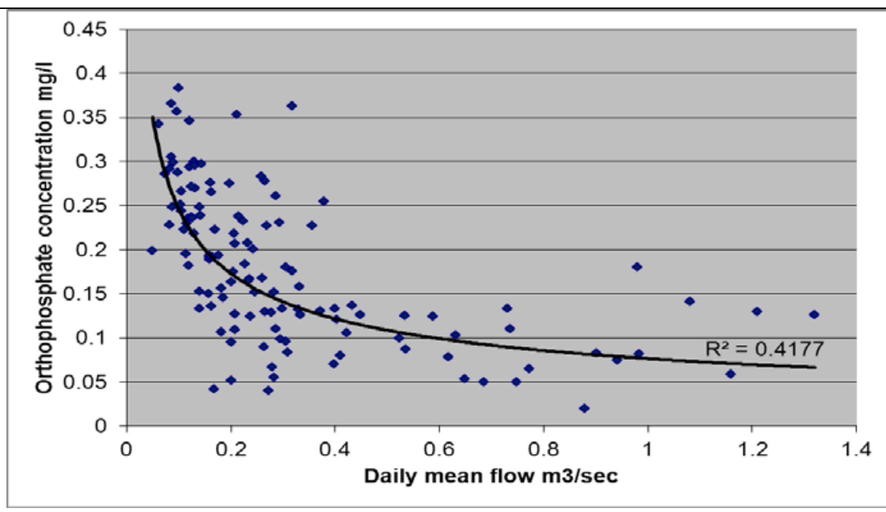


Figure 5.26 Orthophosphate concentrations plotted against flow in Harpers Brook catchment

At site HARP100L, which is upstream of Brigstock and Stanion STW, concentrations are consistent, and do not change with flow, indicating that there is no STW influence. At sites

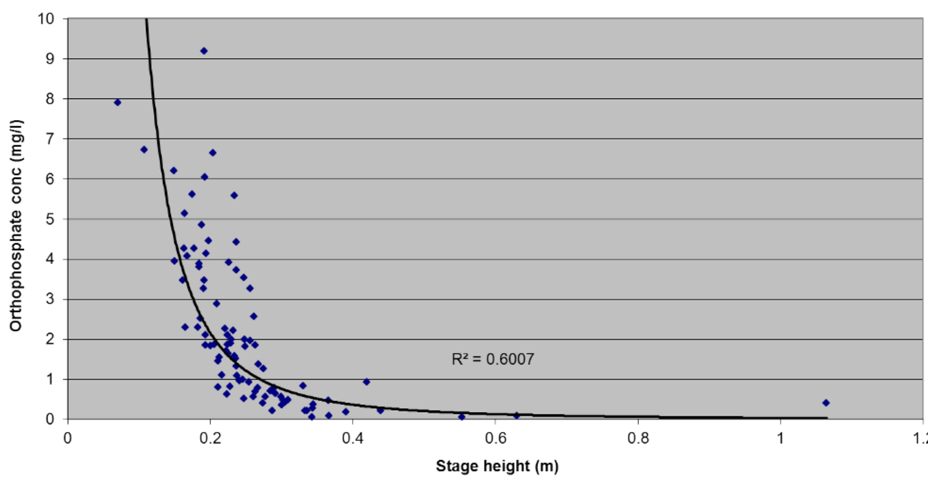
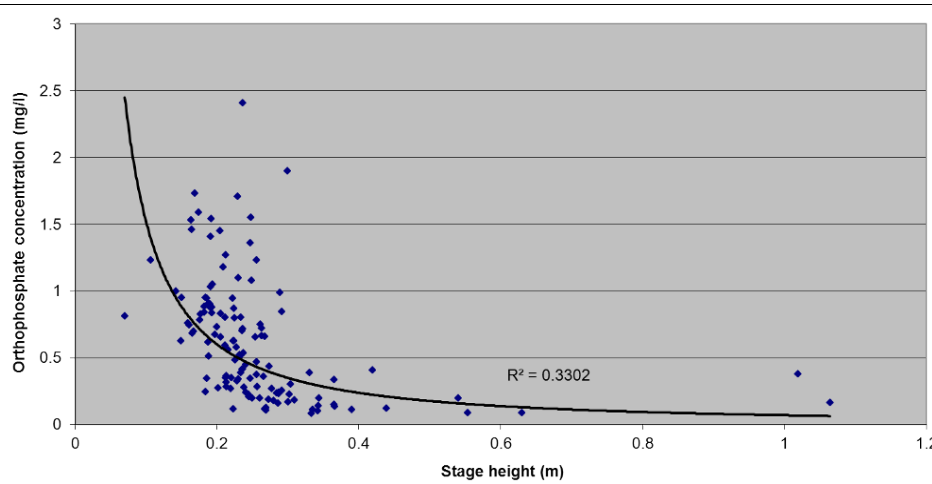
HARP140B (downstream of Stanion STW) and HARP200A (downstream of Brigstock STW), concentrations fall as flow levels increase. This implies a constant level of loading, which is likely to come from a point source.

No information on macrophytes or diatom communities within the catchment was available, so the ecological response of these communities to high P concentrations could not be assessed.

5.6.2 Bourn Brook

WFD status in the Bourn Brook catchment is ‘poor’, and is significant ly below the target ‘good’ standard. The water quality data in Table 5.4 and Figure 5.10 shows that P concentrations in the upper Bourn catchment are below those in the lower catchment, but still fall short of the required standard for the WFD. All sites also show a slight downward trend in concentrations, suggesting that some improvement is being made.

Figure 5.27 shows the relationships between concentrations and measured stage height values, which are used as a substitute for flow. This could not be done for site 32M05, as monitoring at this site stopped in April 1999, before the measurement of stage height began.

a) 32M04	
b) 32M02	

c) 32M01

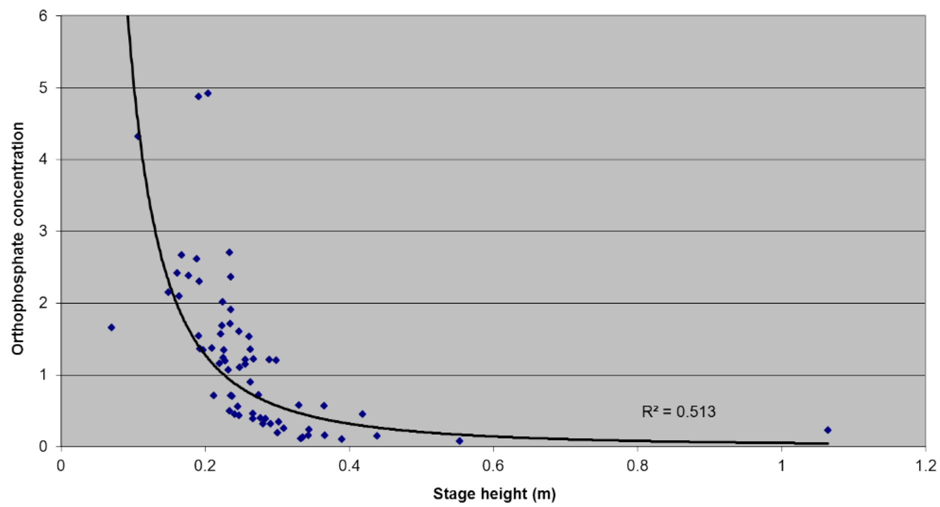


Figure 5.27 Orthophosphate concentrations plotted against stage height in Bourn Brook catchment

These plots show downwards trends in concentration as flow levels increase at all three sites. All of the monitoring sites (including 32M05) showed seasonal patterns in concentrations, with the peak concentrations at low flows in summer, and lowest concentrations at high flows in winter. As in Harpers Brook, this may indicate a constant input which is being diluted by higher flows in the winter months, and this is likely to come from a point source input.

As with Harpers Brook, no information on macrophytes or diatoms communities within the catchment was available, so the impact of high P concentrations on ecology could not be assessed.

6 Discussion and conclusions

6.1 Information gaps

Agricultural sources and pathways of delivery of P are relatively well understood, but the relative contribution of agriculture to P loads, in-stream P concentrations and ecological significance is poorly understood. The level of soil P surplus varies spatially, both across and between farms, and temporally. As fertilizer prices continue to rise leading to a fall in demand, more P from wastewater is recovered and spread to land, and nutrients in manure are recycled, the level of soil P surplus is likely to continue to decrease. Future changes in land use practices and soil and fertiliser management will affect both the surplus and the delivery of P to watercourses. Climate change is also likely to have an impact on water quality, especially in P loads from diffuse sources, as changes in rainfall patterns affect P mobilization and runoff, and in-river P concentrations as summer flow rates decline. P loads from impervious surfaces (such as farmyards) show high levels of variability in concentrations and further research is needed to fully appreciate these fluctuations, while leakage to groundwater is also poorly understood.

There are a number of gaps in our understanding of non-agricultural sources of P. While emissions from STW are well understood, the number and locations of homes which are not on the main sewerage system is not well known, and there is little information about the level of maintenance of these systems, which could affect effluent quality. The extent and impact of misconnections are also poorly understood, and this issue could have significant impacts on discharges to rivers. In addition, there is little understanding of levels of P in runoff from roads and urban areas, although the ecological impact of P is likely to be limited relative to the other diffuse contaminants in urban runoff such as heavy metals.

The most significant gap in our knowledge arises because routine Environment Agency water quality monitoring only measures one form of P (total reactive P), and so provides limited information about fluxes and cycling of P. Concentrations are only measured in the water column, and not in sediment, so it is not known how water column concentrations may be buffered by P in sediments. As river P standards are represented as an annual average concentration, seasonal variations in loading and sensitivity to nutrient enrichment are not considered. We recommend further work on river P monitoring approaches (sediment and water), the setting of standards, and biogeochemical processes and cycling of P within different river systems.

There are a number of uncertainties in our understanding of how river and lake environments respond to increased P concentrations. The cycling of P in rivers in relation to other eutrophication drivers, and the extent to which headwaters can buffer the impacts of P loadings, is poorly understood. Monitoring of macrophyte and diatom communities is presently limited, although this is improving with the implementation of the WFD. For lakes, there is no historical monitoring of ecological communities as no tools were available prior to WFD, although a number of hindcasting studies have estimated P loadings to lakes based on diatom species profiles in the lake sediments.

It is difficult to tell how rivers will respond to reductions in the external P loading and to reductions in ambient river P concentrations, due to the interaction of multiple stressors. In highly eutrophic waters, the limiting factor on algal growth may not be P, so it is difficult to predict how communities will respond to reduced P concentrations. Recent research suggests that river orthophosphate concentrations must be reduced to well below 0.1 mg l^{-1} to limit algal growth (Bowes *et al*., 2011). Studies carried out on lakes to assess the rate of recovery following reductions in P levels have shown recovery times to vary substantially, with shallow lakes taking the longest to recover over around 10-15 years.

A wide range of measures have been used in an effort to reduce diffuse water pollution from agriculture; however, the effectiveness of these has generally been assessed at field or farm level, and little is known about actual reductions in P loads at the catchment scale. A recent evaluation of Phase 2 of the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) using modelling indicates that improvements in management practices will result in significant reductions in pollutant losses. Reductions from the first four years of the ECSFDI are generally predicted to be between five and 10 per cent, but could be up to 36 per cent. These reductions translate into in-river decreases in pollutant concentrations of a similar magnitude.

Control measures for non-agricultural P emissions consist mainly of chemical and biological P removal technologies which are implemented at STW and regulated through the UWWTD. These methods are well regulated and understood, but current effluent standards are still too high for eutrophication control even after P stripping, and there is some uncertainty over how reductions in effluent P will affect river biology (Jarvie *et al*., 2004).

At a national level, there are a number of gaps in knowledge. These include P contributions to rivers from combined sewer overflows (CSOs), numbers of misconnections and the amount of P lost through them and P losses to rivers and groundwater from on-site wastewater treatment systems (OSWTS) and septic tanks. In addition to this, understanding of P behaviour in groundwater is limited.

At present no single database covers the measures being recommended for RBD, the level of implementation, and the locations where measures are used. Information of this type would substantially improve the possibility of assessing measures' effectiveness. We recommend setting up a single database of the measures being recommended within the Anglian RBD to improve water quality (both from point and diffuse sources), with supporting information on the timing, likelihood of uptake, and the location.

At a local level, gaps in knowledge can be more easily filled. In the pilot catchments, there is currently no monitoring of macrophyte and diatom communities, which would help to establish the ecological state of the river. Another significant gap is the size and level of maintenance of OSWTS and septic tanks, although this may be partially filled by the Environment Agency's registration programme depending on the future of this programme. Information on the level of interaction between groundwater and surface waters could also be useful to assess the long-term impacts of P pollution.

In the Bourn Brook catchment, there is currently no monitoring of flow or of water quality in the upper part of the catchment. As this is upstream of the STW, it is important to monitor these parameters to help establish the impact of the STW.

6.2 The contribution of agriculture to eutrophication

Agriculture is only one of a number of sources contributing P to surface waters and agriculture's contribution to P loads at national, river basin district and local catchment scales will therefore be highly dependent on the influx of other sources, notably wastewater. Sources of P enter rivers in different forms, at different times and at variable locations within the catchment. The downstream ecological impacts of these variable nutrient inputs are heavily dependent on their fluxes, concentrations and composition (forms) and how they are physically, chemically and biologically processed within the river channel. Additionally, eutrophication impacts are influenced by a number of ecosystem factors, including supplies of other nutrients carbon (C), nitrogen (N) and silica (Si) and environmental variables such as flow regime, water depth, water clarity, temperature, shading and macroinvertebrate grazing pressures. The extent to which P and/or N, and/or C, or other factors limit, or co-limit, the primary production that results in a cascade of eutrophication symptoms in rivers is not known, yet this is a vital requirement for developing sustainable and cost-effective catchment management strategies to control eutrophication.

Transfers of P from agriculture are highly variable temporally and from site to site. Farmed land, farmyards and farm tracks/roads are the major source areas and there is little doubt that export rates from agricultural land have increased substantially in recent decades due to the intensification of agriculture, with greater surpluses of P leading to soil P accumulation and changes to farming methods increasing the vulnerability of soils to runoff and erosion. Surpluses of P in agriculture are now declining but this will not greatly reduce P transfers in the short-term because of the large store of P that has built up in the soil. Careful management of soils and of the fertilizers and manures applied to land are now needed in critical source areas (those areas with high risk of P mobilization and delivery to the watercourse) to reduce the potential contribution of agriculture to eutrophication. In principle, a small P surplus managed poorly can lead to greater P loss to water than a large surplus managed well.

Phosphorus is delivered in both inorganic and organic forms, in soluble and particulate phases and along both surface and sub-surface pathways but the eutrophication impact of these forms/phases is likely to vary considerably due to the way they are processed in the river channel, and is poorly understood. Most P exported from arable and horticultural land is generally in particulate form, whereas soluble P is often (but not always) the main form of P transported from grassland. Where underdrainage exists, this will be the dominant pathway of P loss and is much more difficult to control without sacrificing the benefits that drainage brings in reducing surface wetness.

The impact of the P exported from agricultural land on river P concentrations is highly dependent on the flow; where flows are low small amounts of P loss will lead to sufficiently high P concentrations to trigger eutrophication. It is perhaps fortuitous that the largest exports of P occur from livestock farming but that livestock farming is concentrated in the wetter areas of western UK with high river flows, allowing dilution of the P input. Hence, catchment headwaters with intensive agriculture and little dilution from upland areas are most vulnerable to agriculturally-driven increases in river P levels. The contribution of agriculture to eutrophication is further complicated by a lack of understanding of the fate and bioavailability of the different forms transported. Although most catchment runoff is derived from farmed land, P export in this runoff is delivered to water bodies during winter storm events when biological activity is minimal. The ecological relevance of the P derived from farmed land is likely to be less than other rural sources (farmyard and road runoff, septic tank discharges) that are also variable in composition, but are delivered more continuously during the year. Point source discharges remain the predominant sources of bioavailable P in UK waters, even in rural areas, and are likely to have the greatest ecological impact. Estimates of agriculture's contribution to eutrophication based simply on modelled annual loads of total P are therefore misleading. This does not mean that agriculture's contribution to eutrophication will not be significant or need to be controlled as part of an integrated catchment approach that tackles all sources of P. A greater understanding of the fate of agriculturally derived P in rivers and the main drivers for eutrophication in rivers is, however, urgently needed to inform nutrient load reduction strategies in catchments and their likely ecological benefits.

At a national level, export coefficient modelling suggests that 70 per cent of the P exported to water is derived from point sources, with agriculture contributing on average about 0.5 kg ha^{-1}.

At an RBD level, export coefficient modelling suggests that 80 per cent of the P exported to water is derived from point sources, with agriculture contributing most of the remaining 20 per cent. The main drivers of P loss within the Anglian RBD are the relatively high soil P levels due to historic fertiliser and pig/poultry manure usage, whilst river P levels are relatively high due to the low drainage volumes and low river flows in an area with the lowest rainfall in the UK.

At a local level, export coefficient modelling suggests that 65 per cent of the P exported to water is derived from point sources, with agriculture contributing 35 per cent. The main drivers of agricultural P loss within the pilot catchments are farmyard areas in close proximity to streams, incidental losses in drainflow from manure applications and background sediment-associated P concentrations in drainflow from the large proportion of cultivated arable fields.

6.3 Potential to reverse eutrophication

Eutrophication is caused by elevated concentrations of nutrients and strategies to reverse eutrophication have generally focussed on reducing the loading of nutrients to the system. In freshwater systems, the nutrient of concern is phosphorus but other factors can influence eutrophication such as flow velocity, light, and invertebrate grazing pressures.

The evidence suggests that reduction in P concentrations in rivers and lakes can be readily achieved through P removal processes at STW that reduce concentrations in effluents. There is less evidence that diffuse P control measures reduce P concentrations in surface freshwaters, due to difficulties in monitoring the diffuse agricultural pollutant load. There is limited evidence to suggest that the reduction in phosphorus concentrations is transferring into detectable ecological improvements. The reasons for this include: P reductions do not decrease P concentrations sufficiently to be limiting; the timescales for observing the ecological response are longer than those of most studies and the monitoring for the detection of ecological responses has been insufficiently targeted or has ignored confounding factors.

Timescales for the recovery of rivers from eutrophication remain unknown but are likely to vary with river type. Larger, slower flowing eutrophic rivers are likely to take longer to recover due to the greater potential for substantial stores of P to accumulate in river sediments and act as an internal P loading, even when contributions from external source have been reduced. Rehabilitation of lakes following eutrophication has been more widely reported. For example, deep lakes (those which stratify permanently throughout the summer season) typically respond rapidly to reductions in nutrient loads, whereas shallow lakes can take decades to recover. Recovery timescales for shallow lakes are in the order of 10-15 years with sustained P loading from the sediment delaying ecological manifestations of reduced external P loads. In some lakes, high external P loading can lead to a change from a clear water state dominated by submerged higher plants to an alternative turbid state dominated by phytoplankton. This change may be maintained by a positive feedback loop whereby a combination of high turbidity, benthic algal mats and changes in sediment composition inhibit the return to a macrophyte-dominated state.

P reduction strategies need, therefore, to take into account the type of water body affected and be targeted at reducing external loadings that result in sustained reductions in SRP concentrations during the period of greatest eutrophication risk. In rivers, this tends to be during the biological growing season. Monitoring of ecological recovery needs to include measurements of P concentrations and a number of indicators of ecological response – phytobenthos, macrophytes and, in lakes, phytoplankton biomass (chlorophyll a) and composition – and be sustained for a long period (up to 15 years for shallow lakes).

6.4 Recommendations for pilot catchment monitoring

If an assessment of the impact of measures to address P and changes in agricultural practice is to be successful, the quality and quantity of monitoring data is vitally important. At present, the level of monitoring in the two pilot catchments is variable. The following recommendations are made for future monitoring.

Harpers Brook

- Retain all current physico-chemical monitoring sites and the single flow monitoring site to ensure a long enough baseline to analyse the effects of changes in practice.
- Undertake 'event monitoring' at key sites through the deployment of auto-samplers which sample automatically at high flows.
- Undertake regular monitoring of macrophyte and diatom communities to measure any improvement in response to P reductions. Diatoms are sensitive to seasonal changes in rivers, so sampling should be undertaken at least four times per year.

Bourn Brook

- Reinstate monitoring site 32M05 in the Upper Bourn Brook for physico-chemical monitoring and retain other downstream monitoring sites. This should be done as soon as possible to ensure that there is some baseline data from which the effects of any changes can be determined.
- Undertake 'event monitoring' at key sites through the deployment of auto-samplers which sample automatically at high flows.
- Install flow monitoring at the water body outflow point.
- Undertake regular monitoring of macrophyte and diatom communities to measure any improvement in response to P reductions. Diatoms are sensitive to seasonal changes in rivers, so sampling should be undertaken at least four times per year.

Self-sampling by farmers may also be a useful tool in monitoring changes in water quality levels if used appropriately. An organised monitoring programme is needed to measure P levels in drains and runoff as well as in rivers.

6.5 Conclusions

Phosphorus is a naturally occurring substance in freshwaters and is an important nutrient for the growth of algae and higher plants. Rivers and lakes in England and Wales are naturally deficient in biologically available phosphorus; natural concentrations of SRP in rivers are low (below 0.03

mg l^{-1}).Phosphorus is widely acknowledged to be the major limiting nutrient in natural systems, and current research suggests that the P concentration at which algal growth becomes limited is below 0.1 mg SRP $l^{-1}$and that above this level, any additional P has no extra effect, because other factors become limiting to growth.

High P levels can therefore trigger changes to the ecologies of these systems, known as eutrophication. The primary receptors in these ecosystems are algae and macrophytes, where increased exposure to bioavailable P leads to accelerated growth and enhanced productivity. This alters the competitive balance between species, which then affects the composition and diversity of these communities, also affecting water quality.

The following conclusions can be drawn from the evidence base described in this report with respect to the stages in the DPSIR framework.

The size and extent of the reservoir of P in agricultural soils that is potentially available for transfer to water is driven by inputs of P from fertilisers and manure and the proportion that is removed in crops. The national soil P surplus is now lower than that calculated for 1935 and further downward trends can be expected if the price of fertilizers and feeds continues to increase. Recovery and recycling of wastewater P is also forecast to increase in the future, reducing dependence on imports of highly soluble fertilizers.

Loss of P from agricultural land is principally a storm-driven process, with the exception of irrigation of high-value crops and the hosing down of concrete areas on farms which can have preferential routes to water courses through drainage systems. Typically. most P is transported during a few major events and seasonality is largely dominated by the pattern of storm events during the year.

Sources arising from impervious and pervious surfaces deliver both soluble and particulate forms of P to water courses at different rates.

Land use patterns and farming practice largely control the principal pathways of P loss from agricultural land to water. Farming practices that increase soil compaction can increase the transport of P along surface flow pathways.

The main driver for increased losses of non-agricultural P is growth in population leading to higher levels of sewage. Drivers for improvements include legislation such as the Urban Waste Water Treatment Directive, source control measures such as reduction of P levels in detergents, and concerns over the exhaustion of raw P supplies, which will encourage P recovery and recycling.

Non-agricultural P comes from both natural and anthropogenic sources, but losses from background sources are generally low. Anthropogenic sources include domestic sewage from sewage treatment plants and septic tanks, industrial discharges, and diffuse urban and road runoff. Misconnections are also an issue, as they allow domestic sewage to flow directly to rivers without undergoing any treatment.

The most common pathway for non-agricultural P loads is through direct discharges, such as CSOs and sewer outfalls, while effluent from septic tanks may also travel through soakaways and field drains. Urban and road sources come from runoff from impervious surfaces and are storm-dependent.

It has been estimated that P from STW effluent makes up around 92 per cent of all non-agricultural P losses, with a further three per cent from misconnections, one per cent from industrial discharges and four per cent from detergents. These figures come from the Non-agricultural Diffuse Water Pollution (NADWP) WFD preliminary cost-effectiveness group. The detergent figure includes: washing activities (such as cars), misuse of the drainage system, and small package treatment plants/septic tanks (ICF International, 2007).

Phosphorus takes many chemical forms, which confounds understanding of water nutrient status and its relationship to ecology. Bioavailable P consists mainly of inorganic soluble forms, organic forms and labile P which rapidly desorbs from particles. Within the river, P undergoes numerous transformations between particulate and dissolved phases, and between the water column and bed sediments. The large spatial and temporal variability in P mobilization and delivery across a catchment also makes source apportionment difficult.

These processes vary in time and space depending on stream size, catchment geology and, channel hydromorphology. In small, headwater streams, uptake of SRP may restrict downstream P supply, whereas in larger rivers SRP uptake from the water column by phytoplankton and retention of phosphorus in bed sediments have an impact. Catchments with a clay or silt geology with high concentrations of iron and aluminium oxides have a high capacity for phosphorus sorption. River hydrology and geomorphology control flow velocity and contact time between water and P exchange surfaces.

The nutrient status of rivers and lakes is assessed by routine monitoring of phosphorus concentrations in the water column, which has taken place since 1990. Status is assessed at different concentrations of soluble reactive phosphorus (SRP) depending on the sensitivity of the river to eutrophication. For lakes, total P is used instead of SRP.

Although high concentrations of dissolved phosphorus are thought to indicate eutrophication problems, river and lake sensitivity is variable and influenced by water chemistry and hydrology. In addition, the interaction of multiple stressors makes it difficult to predict how an individual river or lake will respond to high P concentrations and means that simply reducing inputs may not be enough to reverse ecological changes.

The impact of P inputs depends on the timing of loading. Point source P inputs from STW are continuous or semi-continuous, while runoff from agricultural land is storm-dependent, and consequently contributes more in the wet winter months. At these times river dilution is higher, and the ecological impact of the P inputs is likely to be less. When flow levels are low during the summer, continuous P inputs can have a major impact on ecological quality. The form of P discharged also affects ecological quality. Point sources discharge most of their P as highly bioavailable SRP, while agricultural runoff contains higher levels of PP.

Strategies to reverse eutrophication focus on reducing the loading of nutrients to the system, but other limiting factors can affect growth, which prevent such reductions from improving biological composition. Even if P concentrations are reduced to natural levels, this may not cause an immediate biological response if, for example, a large P reservoir has accumulated in river or lake sediments, as it can be released back into the water column.

A number of measures can help reduce the transfer of P from agricultural land to water. If mitigation options are to be successful, they must (a) be iterative, (b) involve stakeholder-driven programmes and (c) address the inherent complexity of all P sources within catchments.

Whilst there is a general consensus that combinations of measures rather than individual actions are required to control inputs of P from the large number of widely distributed sources and areas on farms, there remains, in reality, a high level of uncertainty as to whether adopted measures will have any impact in reducing P export or improving the ecological quality of the receiving water. This is due to a number of reasons:

(1) The origin of diffuse P on the farm it not always visible or clear and there is usually no monitoring data to assess the relative importance of different farm sources.
(2) Whether an individual measure works at farm scale is heavily dependent on landscape type, farm management skill and weather conditions at individual sites. Effectiveness is therefore likely to vary considerably from one year to the next, even on the same site.
(3) At a catchment scale, additional factors become important, such as the intensity with which a particular measure, or suite of measures, is implemented within the wider catchment area, the extent of dilution by upstream waters, in-stream processing and contributions from other sources.
(4) There is a lack of understanding of the precise impact of different forms of P on stream ecology (particularly particulate and organic P) and the resilience of aquatic ecosystems to alter ecological status means that improvements may not manifest themselves for some years.
(5) The ecological significance of sources of P other than agriculture may have been underestimated by modelling approaches.

Phosphorus in STW effluent arises mainly from human waste and the use of detergents which contain phosphate for laundry and dishwashers. Chemical and biological processes can be used to remove P from effluent within the treatment regime. Most STW in England and Wales use chemical P removal as it can be retrofitted to existing plants, and is simple and inexpensive.

Other technologies to remove P include waste stabilisation ponds and the use of beds of media that bind phosphorus (such as clay). These options are used in smaller STW where it may not be cost-effective to install chemical or biological P removal. Source reduction can also be used to reduce P concentrations going into STW, for example by replacing P in laundry detergents with zeolite.

It is estimated that 78 per cent of the P load to the water environment comes from households, with around 16 per cent of this coming from dishwasher and laundry detergents (White and Hammond, 2009; Defra, 2009b). There is currently no legislation to ban the use of P in these detergents, but a consultation process was undertaken in 2008, and P use in laundry detergents will be banned from 2015.

Tier 1: England & Wales

In England and Wales, 65-73 per cent of the annual total P load in waters derives from point sources, mostly sewage treatment works (STW); 18-28 per cent is derived from agricultural sources, and background is approximately six per cent. For soluble reactive P, the proportion from agriculture falls, making up around a fifth, while the household contribution rises to 72 per cent. The largest sources of agricultural P are beef and dairy farms, making up 19 and 15 per cent respectively, while fertiliser contributes around a tenth of the total.

Agricultural P losses occur in three main ways: dissolved in runoff after a rainfall event, in soil water where soil P levels are high, and bound to soil as particulate P when soil is eroded. Monitoring has shown that particulate P usually makes up the largest proportion of TP exported from arable farms, whereas SRP is often the more dominant fraction from grassland farms (Withers and Lord, 2002; Jordan *et al*., 2005; Withers and Hodgkinson, 2009), but this is by no means universal. Runoff from impervious surface on farms contains higher concentrations of P than runoff from fields, and also contains a high proportion of soluble P. These sources are likely to release P continually throughout the year, and will be more environmentally damaging.

Monitoring has shown that 37 per cent of English rivers and eight per cent of Welsh rivers contain over 0.10 mg SRP l^{-1}. Higher concentrations are generally seen in the lowlands of central, eastern and southern England, due to variations in geology and population. Under the UWWTD and Nitrates Directive, five per cent of rivers are classified as suffering from eutrophication, while two per cent of lakes and 15 transitional and coastal waters are designated as “eutrophic” and a further 44 river water bodies are being considered for eutrophic designation at present.However, half of the river length in England fails the WFD P standard as does 64 per cent of lakes monitored to date. The majority of these water bodies have extended deadlines under WFD, with investigations ongoing into whether there is a eutrophication problem to solve.

A wide range of regulatory measures are being introduced under the WFD such as further STW improvements, a ban on P in laundry detergents, and controls on manure management under the Nitrates Directive. In addition, voluntary schemes such as Catchment Sensitive Farming and Environmental Stewardship will help farmers implement and fund measures to reduce sources, mobilisation and transport of agricultural P. At present, the effectiveness of these measures in reducing P loads is uncertain, but agricultural loads are expected to be reduced by up to 60 per cent in some areas.

Tier 2: Anglian RBD

At present, 18 per cent of water bodies in Anglian RBD are at ‘good’ ecological status (compliant for all chemical parameters and biological elements), and status for P is much lower than other determinands, with less than 35 per cent at ‘high’ or ‘good’ status. Although the RBD is predominantly rural, the main source of P pollution is point sources. Agricultural inputs of P come from three main sources: fertilisers, manures and hard standings (dairy). Modelling using the PSYCHIC model predicted that P loss from fertiliser was widespread across the RBD, with P losses from areas of hard standing and manure spreading being more concentrated in small areas. The majority of the water bodies in the RBD have been assessed as ‘probably at risk’ from agricultural sources of P by the Environment Agency (Environment Agency, 2010c).

While agricultural measures can help to reduce P losses in the RBD, the estimated net cost of reductions in this area is high, at between £16 and £204 per kg. Anthony *et al.* (2009) estimated that reductions of up to 30 per cent could be achieved through measures which offer cost savings or have no net cost.

Tier 3: Pilot catchments

The pilot catchments were selected to provide evidence of the agricultural contribution to P pollution. The Harpers Brook and Upper Bourn Brook catchments were selected as they were not covered by any other catchment initiatives and failed the WFD only for P.

In the Harpers Brook catchment, the highest P concentrations came from point sources, and were found around the catchment’s two STW. Around 55 per cent of the total P load in the catchment was estimated to come from these STW. Diffuse P losses were highest in the upper and central parts of the catchment, and came from three main sources: areas of hard standing, manures, and fertilisers. The PSYCHIC model predicted that concentrations of dissolved forms of P were around twice as high as those for particulate forms from diffuse sources.

P concentrations in the Bourn Brook were at similar levels to Harpers Brook, with the highest concentrations at Bourn STW, although similar concentrations were observed from other, non-STW, point sources. Diffuse P losses are spread fairly evenly across the catchment. Again similarly to Harpers Brook, concentrations of dissolved P from diffuse sources are much higher than particulate P. The decrease in P concentration seen in both catchments as river flow levels rise supports the modelling predictions that point sources are dominant.

References

ADAS 2002a. Development of a database of agricultural drainage. Final Report for Project ES0111. Department for Environment, Food and Rural Affairs, 48 pp.

ADAS 2002b. Methods and measures to minimise the diffuse pollution of water from agriculture – a critical appraisal. Report to Defra.

ADAS 2007a. Identifying the gap to meet WFD and best policies to close the gap. Defra Project WT0719CSF.

ADAS 2007b. Cost and effectiveness of policy instruments for reducing diffuse agricultural pollution: Part One – Farm scale modelling. Defra Projects WQ0106 and ES0205.

Ahuja, L.R. 1986. Characterization and modelling of chemical transfer to run off. Advances in Soil Science, 4, 149-188.

AIC 2009. Fertiliser statistics. Agricultural Industries Corporation.

Anthony, S., Duethman, D., Gooday, R., Harris, D., Newell-Price, P., Chadwick, D. and Misselbrook, T. 2009. Quantitative assessment of scenarios for managing trade-off between the economic performance of agriculture and the environment and between different environmental media. Final Report, Defra Project WQ0106 (Module 6), 95 pp.

Aronsson, H., Stenberg, M. and Ulén, B. 2011. Leaching of N, P and glyphosate from two soils after herbicide treatment and incorporation of a ryegrass catch crop. Soil Use and Management, 27, 54-68.

Babatunde, A.O., Zhao, Y.Q., O'Neill, M. and O'Sullivan, B. 2008. Constructed wetlands for environmental pollution control: A review of developments, research and practice in Ireland. Environment International, 34, 116-126.

Bannerman, R., Owens, D., Dodds, R. and Hornewer, N. 1993. Sources of pollutants in Wisconsin stormwater. Water Science and Technology, 28 (3-5), 241-259.

Baxter, S.J., Oliver, M.A. and Archer, J.R. 2006. The Representative Soil Sampling Scheme of England and Wales: the spatial variation of topsoil nutrient status and pH between 1971 and 2001. Soil Use and Management, 22, 383-392.

Bechmann, M.E., Kleinman, P.J.A., Sharpley, A.N. and Saporito, L.S. 2005. Freeze-thaw effects on phosphorus loss in run-off from manures and catch-cropped soils. Journal of Environmental Quality, 34, 2301-2309.

Beechie, T., Pess, G., Roni, P. and Giannico, G. 2008. Setting river restoration priorities: a review of approaches and a general protocol for identifying and prioritizing actions. North American Journal of Fisheries Management, 28, 891-905.

Bilotta, G.S., Brazier, R.E. and Haygarth, P.M. 2007. The impacts of grazing animals on the quality of soils, vegetation and surface waters in intensively managed grasslands. Advances in Agronomy, 94, 237-280.

Bowes, M.J., Ings, N.L., McCall, S.J., Warwick, A., Barrett, C., Wickham, H.D., Harman, S.A., Armstrong, L.K., Scarlett, P.M., Roberts, C. and Lehmann K. 2011. Nutrient and light limitation of periphyton in the River Thames: implications for catchment management. Science of the Total Environment DOI 10.1016/j.scitotenv.2011.09.082.

Bowes, M.J., Neal, C., Jarvie, H.P., Smith, J.T. and Davies, H.N. 2010. Predicting phosphorus concentrations in British rivers resulting from the introduction of improved phosphorus removal from sewage effluent. Science of the Total Environment, 408, 4239-4250.

Bowes, M.J., Smith, J.T., Jarvie, H.P. and Neal C. 2008. Modelling of phosphorus inputs to rivers from diffuse and point sources. Science of the Total Environment, 395, 125-138.

Bowes, M.J., Smith, J.T., Hilton, J. and Armitage, P.D. 2007. WFD objectives are explicitly ecological and the outcomes are defined on the basis of biological parameters. Canadian Journal of Fisheries and Aquatic Sciences, 64, 227-238.

Bowes, M.J., Hilton, J., Irons, G.P. and Hornby, D.D. 2005. The relative contribution of sewage and diffuse phosphorus sources in the River Avon catchment, southern England: implications for nutrient management. Science of the Total Environment, 344, 67-81.

Buczko, U. and Kuchenbuch, R.O. 2007. Phosphorus indices as risk-assessment tools in the U.S.A. and Europe: A review. Journal of Plant Nutrition and Soil Science, 170, 445-460.

Carter, M.R. 1998. Conservation tillage practices and diffuse pollution. In: Petchey, T., D'Arcy, B., Frost, A. (eds.), Non-Point Pollution II. The Scottish Agricultural College, Aberdeen, UK. pp. 51-60.

Carvalho, L. and Moss, B. 1995. The current status of a sample of English Sites of Special Scientific Interest subject to eutrophication. Aquatic Conservation: Marine and Freshwater Ecosystems, 191-204.

CEFIC/CEEP. 1998. International conference on phosphorus recovery from sewage and animal wastes. Summary of conclusions and discussions. CEFIC, Belgium.

Chapman, A.S., Foster, I.D.L., Lees J.A. and Hodgkinson, R. A. 2005. Sediment delivery from agricultural land to rivers via subsurface drainage. Hydrological Processes, 19(15), 2875-2897.

Chardon, W.J., Aalderink, G.H. and van der Salm, C. 2007. Phosphorus leaching from cow manure patches on soil columns. Journal of Environmental Quality, 36, 17-22.

Chardon, W.J. and Schoumans, O.F. 2007. Soil texture effects on the transfer of phosphorus from agricultural land in the river deltas of Northern Belgium, The Netherlands and North-West Germany. Soil Use and Management, 23 (Supplement 1), 16-24.

CCW (Countryside Council for Wales) 2009. Sites of Special Scientific Interest. [Online] Available at: http://www.ccw.gov.uk/landscape--wildlife/protecting-our-landscape/special-landscapes--sites/protected-landscapes/sssis.aspx. [Accessed June 2010].

Cordell, D., Drangert, J. and White, S. 2009. The story of phosphorus: global food security and food for thought. Global Environmental Change, 19, 292-305.

Crabtree, B., Kelly, S., Green, H., Squibbs, G. and Mitchell, G. 2009. Water Framework Directive catchment planning: a case study apportioning loads and assessing environmental benefits of programme of measures. Water Science and Technology, 59.3.

Cranfield University 2007. Scoping study to assess soil compaction affecting upland and lowland grassland in England and Wales. Defra Project BD2304.

Cuttle, S.P., Macleod, C. J. A., Chadwick, D. R., Scholefield, D., Haygarth, P. M., Newell-Price, P., Harris, D., Shepherd, M. A., Chambers, B. J. and Humphrey, R. 2006. An Inventory of Methods to Control Diffuse Water Pollution from Agriculture (DWPA): User Manual. Defra Project ES0203.

Davison, P.S., Withers, P.J.A., Lord, E.I., Betson, M.J. and Stromqvist, J. 2008. PSYCHIC – A process-based model of phosphorus and sediment mobilisation and delivery within agricultural catchments. Part 1: Model description and parameterisation. Journal of Hydrology, 350, 290-302.

Dawe, V. 2010. Non-agricultural diffuse water pollution. [Online] Available at: http://www.euwfd.com/SH100226_Defra_Non-Agri_Diff_Poll.pdf [Accessed March 2010]

Dayton, E.A. and Basta, N.T. 2005. Use of drinking water treatment residuals as a potential best management practice to reduce phosphorus risk index scores. Journal of Environmental Quality, 34, 2112-2117.

Deasy, C., Quinton, J.N., Silgram, M., Bailey, A.P., Jackson, B. and Stevens, C.J. 2009. Mitigation options for sediment and phosphorus loss from winter-sown arable crops. Journal of Environmental Quality, 38, 212-213.

Defra 2003.Cost curve assessment of phosphorus mitigation options relevant to UK agriculture. Defra Project PE0203.

Defra, 2008a. Future Water – the government's water strategy for England. [Online] Available at: http://archive.defra.gov.uk/environment/quality/water/strategy/pdf/future-water.pdf. [Accessed June 2010]

Defra 2008b. Consultation on options for controls on phosphates in domestic laundry cleaning products in England. [Online] Available at: http://www.endseurope.com/docs/80215a.pdf [Accessed June 2010].

Defra 2009a. Key facts about inland water quality and use: Phosphate concentrations in rivers 1995-2008. [Online] Available at: http://www.defra.gov.uk/evidence/statistics/environment/inlwater/kf/iwkf09.htm [Accessed June 2010].

Defra 2009b. Impact assessment of ban on phosphorus in domestic laundry cleaning products. [Online] Available at: http://www.defra.gov.uk/corporate/consult/detergents-regs2005/ai-phosphate-detergent.pdf [Accessed June 2010].

Defra 2010a. Agriculture in the United Kingdom. 2009. Stationary Office, London.

Defra 2010b.The Fertiliser Manual (RB209).Eighth Edition. Department for Environment, Food and Rural Affairs. The Stationary Office, London.

Dils, R.M. and Heathwaite, A.L. 1996. Phosphorus fractionation in hillslope hydrological pathways contributing to agricultural runoff. In: Andersen, M.G. and Brooks, S.M. (eds.). Advances in Hillslope Processes, Volume 1. John Wiley and Sons, Chichester, UK. pp 229-251.

Dils, R., Leaf, S., Robinson, R. and Sweet, N. 2001. Phosphorus in the environment – why should recovery be a policy issue? Environment Agency. CEEP 12-13 March 2001, [Online] Available at: http://www.nhm.ac.uk/research-curation/projects/phosphate-recovery/Nordwijkerhout/Sweet0204.doc. [Accessed June 2011]

Domburg, P., Edwards, A.C., Sinclair, A.H., and Chalmers, N.A.. 2000. Assessing nitrogen and phosphorus efficiency at farm and catchment scale using nutrient budgets. Journal of Science Food and Agriculture, 80, 1946-1952.

Donohue, I., Styles, D., Coxon, C. and Irvine, K. 2005. Importance of spatial and temporal patterns for assessment of risk of diffuse nutrient emissions to surface waters. Journal of Hydrology, 304, 183-192.

Dorioz, J.M., Wang, D., Poulenard, J. and Trevisan, D. 2006. The effect of grass buffer strips on phosphorus dynamics — A critical review and synthesis as a basis for application in agricultural landscapes in France. Agriculture, Ecosystems and the Environment, 117, 4-21.

Dou, Z., K.F. Knowlton, R.A. Kohn, Z. Wu, L.D. Satter, G. Zhang, J.D. Toth, and J.D. Ferguson. 2002. Phosphorus characteristics of dairy faeces affected by diets. Journal of Environmental Quality, 31, 2058–2065.

Dudley, B. and May, L. 2007. Estimating the phosphorus load to water bodies from septic tanks. Centre for Ecology and Hydrology, pp45.

Dunne, E.J., Culleton, N., O'Donovan, G., Harrington, R. and Olsen, A.E. 2005. An integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water. Ecological Engineering, 24, 337-354.

ECSFDI. 2008. Evaluation Report: Final Report. [Online] Available at: http://www.defra.gov.uk/foodfarm/landmanage/water/csf/documents/ecsfdi-evaluation-report.pdf [Accessed March 2011]

Edwards, A.C. and Hooda, P.S. 2007. Farmyard point discharges and their influence on nutrient and labile carbon dynamics in a second order stream draining through a dairy unit. Journal of Environmental Management, 87, 591-599.

Edwards, A.C. and Withers, P.J.A. 2007.Linking phosphorus sources to impacts in different types of water body. Soil Use and Management, 23 (Supplement 1), 133-143.

Edwards, A.C. and Withers, P.J.A. 2008. Transport and delivery of suspended solids, nitrogen and phosphorus from various sources to freshwaters in the UK. J Hydrology, 350, 144-153.

Edwards, A.C., Kay, D., McDonald, A., Francis, C., Watkins, J., Wilkinson, R.J. and Wyer, M.D. 2007. Farmyards, an overlooked source for highly contaminated runoff. Journal of Environmental Management, 87, 551-559.

Elliott, H.A., O'Connor, G.A., Lu, P. and Brinton, S. 2002. Influence of water treatment residuals on phosphorus solubility and leaching. Journal of Environmental Quality, 31, 1362-1369.

Emmett, B.A., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C. and Woods, C. 2010. Countryside Survey Soil Report for 2007. CS Technical Report No. 9/07. Centre for Ecology and Hydrology, Wallingford.

Environment Agency. 2007. Water Framework Directive Programme Approach to risk assessment (technical). http://www.environment-agency.gov.uk/static/documents/Research/Risk_Assessment_Approach_technical_final.pdf

Environment Agency 2009a. Discharge Consents – Who needs one? [Online] Available at:http://www.environment-agency.gov.uk/business/regulation/32038.aspx [Accessed Jan 2011].

Environment Agency 2009b. Diffuse pollution programme draft briefing paper: tackling contributions of pollutants from small sewage discharges.

Environment Agency 2010a. Diffuse Pollution Programme Strategic Assessment Report – Phosphorus. (Version 12: 02/02/10).

Environment Agency 2010b. Water Quality. [Online] Available at: http://www.environment-agency.gov.uk/research/planning/34383.aspx [Accessed Jan 2011]

Environment Agency 2010c. Agricultural impacts in Anglian region: An evidence base. (Version 2, March 2010).

Environment Agency 2010d. Anglian River Basin District Phosphorus Review Evidence Base.

Environment Agency 2011. Catchment Sensitive Farming, ECSFDI Phase 1 & 2 Full Evaluation Report.

European Commission 2010. Links between the Water Framework Directive (WFD 2000/60/EC) and Nature Directives (Birds Directive 2009/147/EC and Habitats Directive 92/43/EEC). Frequently Asked Questions. [Online]. Available at: http://circa.europa.eu/Public/irc/env/wfd/library?l=/framework_directive/implementation_conventio/biodiversity_legislation&vm=detailed&sb=Title [Accessed July 2011]

Foresight Land Use Futures Project 2010. Final Project Report. The Government Office for Science, London.

Foy, R.H. and Bailey-Watt, A.E. 1998. Observations on the spatial and temporal variation in the phosphorus status of lakes in the British Isles. Soil Management, 14, 131-138.

Foy, R.H. and Lennox, S.D. 2006.Evidence for a delayed response of riverine phosphorous exports from increasing agricultural catchment pressures in the Lough Neagh catchment. Journal of Oceanography,51, 665-663.

Foy, R.H. and Withers, P.J.A. 1995. The contribution of agricultural phosphorus to eutrophication. Proceedings of the Fertiliser Society No. 365. The Fertiliser Society, Peterborough.

Foy, R.H., Bailey, J.S. and Lennox, S.D. 2002. Mineral balances for the use of phosphorus and other nutrients by agriculture in Northern Ireland from 1925 to 2000 – methodology, trends and impacts of losses to water. Irish Journal of Agricultural and Food Research, 41, 247-263.

Franklin, P., Dunbar, M. and Whitehead, P. 2008. Flow controls on lowland river macrophytes: a review. Science of the Total Environment, 400, 369-378.

Green, F.H.W. 1980. Field under-drainage before and after 1940. Agricultural History Review, 28(2), 120-123 www.bahs.org.uk/28n2a5.pdf

Hart, M.R., Quin, B.F. and Nguyen, M.L. 2004. Phosphorus runoff from agricultural land and direct fertilizer effects: A review. Journal of Environmental Quality, 33, 1954-1972.

Haygarth, P.M. and Jarvis, S.C. 1999. Transfer of phosphorus from agricultural soils. Advances in Agronomy, 66, 195-249.

Haygarth, P.M., Simon, H., Betson, M., Harris, D., Hodgkinson, R. and Withers, P.J.A. 2009. Mitigating diffuse phosphorus transfer from agriculture according to cost and efficiency. Journal of Environmental Quality, 38, 2012-2022.

Haygarth, P.M., Hepworth, L. and Jarvis, S.C. 1998. Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. European Journal of Soil Science, 49, 65-72.

Heathwaite, A.L. 2010. Multiple stressors on water availability at global to catchment scales: understanding human impact on nutrient cycles to protect water quality and water availability in the long term. Freshwater Biology, 55 (Suppl.), 241-257.

Heckrath, G., Brookes, P.C., Poulton, P.R. and Goulding, K.W.T. 1995. Phosphorus leaching from soils containing different P concentrations in the Broadbalk experiment. Journal of Environmental Quality, 24, 904-910.

Hecky, R.E. and Kilham, P. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment. Limnology and Oceanography, 33, 796-822.

Hilton, J., O'Hare, M., Bowes, M.J. and Jones, I. 2006. How green is my river? A new paradigm of eutrophication in rivers. Science of the Total Environment, 360, 66-83.

Hively, W.D., Bryant, R.B. and Fahey, T.J. 2005. Phosphorus concentrations in overland flow from diverse locations on a New York dairy farm. Journal of Environmental Quality, 34, 1224-1233.

Hodgkinson, R.A. and Withers, P.J.A. 2007. Sourcing, transport and control of phosphorus loss in two English headwater catchments. Soil Use and Management, 23 (Supplement 1), 92-103.

Holman, I.P., Howden, N.J.K., Bellamy, P., Willby, N., Whelan, M.J. and Rivas-Casado, M. 2010. An assessment of the risk to surface water ecosystems of groundwater P in the UK and Ireland. Science of the Total Environment, 408, 1847-1857.

Holmes, N.T.H., Newman, J.R., Chadd, S., Rouen, K.J., Saint, L. and Dawson, F.H. 1999. Mean Trophic Rank: a User's Manual. Environment Agency R&D Technical Report E38. 134 pp.

House, W.A. and Casey, H. 1989. Transport of phosphorus in rivers. Regional SCOPE workshop 1: Europe. Saskatchewan Institute of Pedology, Canada, p253-282.

Hulme, M., Jenkins, G.J., Lu, X., Turnpenny, J.R. 2002.Climate change scenarios for the UK. The UKCIP02 Scientific Report. Tyndall Centre for Climate Change Research, School of Environmental Sciences, University of East Anglia, Norwich.

ICF International 2007. Preliminary cost effectiveness analysis of the Water Framework Directive: Revised after stakeholder review. December 2007.

James, C., Fisher, J., Moss, B. 2003. Nitrogen-driven lakes: the Shropshire and Cheshire meres? Arch Hydrobiol, 158, 249-266.

Jarvie, H.P., Neal, C. and Williams, R.J. 2004. Assessing changes in phosphorus concentrations in relation to in-stream plant ecology in lowland permeable catchments: bringing ecosystem functioning into water quality monitoring. Water, Air, and Soil Pollution: Focus, 4, 641-655.

Jarvie, H.P., Neal, C. and Withers, P.J. A. 2006. Sewage effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus? Science of the Total Environment, 360, 246-253.

Jarvie, H.P., Neal, C., Withers, P.J.A., Baker, D.B., Richards, R.P. and Sharpley, A.N. 2010. Quantifying phosphorus retention and release in rivers and watersheds using Extended Endmember Mixing Analysis (E-EMMA). Journal of Environmental Quality, 40, 492-504.

Jarvie, H.P., Haygarth, P.M., Neal, C., Butler, P., Smith, B. and Naden, P.S. 2008. Stream water chemistry and quality along an upland-lowland rural land-use continuum, south west England. Journal of Hydrology, 350, 215-231.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, .F., Deneke, R., Dokulil, M.T., Foy, B., Gerdeaux, D., Hampton, S.E., Hilt, S., Kangur, K., Köhler, J., Lammens, E.H.H.R., Lauridsen, T.L., Manca, M., Miracle, M.R., Moss, B., Nõges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C.L., Straile, D. Tatrai, I., Willén, E. and Winder, M. 2005. Lake responses to reduced nutrient loading – an analysis of contemporary long-term data from 35 case studies. Freshwater Biology, 50, 1747-1771.

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

Johnston, A.E. and Dawson, C.J. 2005. Phosphorus in agriculture and in relation to water quality. A Report to Association of Independent Consultants, AIC Peterborough.71 pp.

Jones, I.J. and Sayer, C.D. 2003. Does the fish-invertebrate-periphyton cascade precipitate plant loss in shallow lakes? Ecology, 84, 2155–2167.

Jones, I.J., Young, J.O., Eaton, J.W., Moss, B. 2002. The influence of nutrient loading, dissolved inorganic carbon and higher trophic levels on the interaction between submerged plants and periphyton. J Ecology, 90, 12-24.

Jordan, P., Menary, W., Daly, K., Kiely, G., Morgan, G., Byrne, P. and Moles, R. 2005. Patterns and processes of phosphorus transfer from Irish grassland soils to rivers – integration of laboratory and catchment studies. Journal of Hydrology, 304, 20-34.

Kay, P., Edwards, A.C. and Foulger, M. 2008. A review of the efficacy of contemporary agricultural stewardship measures for ameliorating water pollution problems of key concern to the UK water industry. Agricultural Systems, 99, 67-75.

Kayhanian, M., Suverkropp, C., Ruby, A., and Tsay, K. 2007. Characterization and prediction of highway runoff constituent event mean concentration. Journal of Environmental Management, 85, 279-295.

Kelly, M.G. and Whitton, B.A. 1995. The Trophic Diatom Index: a new index for monitoring eutrophication in rivers. Journal of Applied Phycology, 7, 433-444.

Kelly, M.G., Juggins, S., Guthrie, R., Pritchard, S., Jamieson, B.J., Rippey, B., Hirst, H., Yallop, M.L., 2008. Assessment of ecological status in UK rivers using diatoms. Freshwater Biology 53, 403-422.

Kinniburgh, J.H. and Barnett, M. 2010. Orthophosphate concentrations in the River Thames: reductions in the past decade. Water and Environment Journal, 24, 107-115.

Kleinman, P.J.A., Bryant, R.B., Reid, W.S., Sharpley, A.N. and Pimentel, D. 2000. Using soil phosphorus behavior to identify environmental thresholds. Soil Science, 165, 943-950.

Kleinman, P., Sullivan, D., Wolf, A., Brandt, R., Dou, Z., Elliott, H., Kovar, J., Leytem, A., Maguire, R., Moore, P., Saporito, L., Sharpley, A., Shober, A., Sims, T., Toth, J., Toor, G., Zhang, H. and Zhang, T. 2007. Selection of a water-extractable phosphorus test for manures and biosolids as an indicator of runoff loss potential. Journal of Environmental Quality, 36, 1357-1367.

Koskiaho, J., Ekholm, P., Raty, M., Riihimaki, J. and Puustinen, M. 2003. Retaining agricultural nutrients in constructed wetlands: experiences under boreal conditions. Ecological Engineering, 20, 89-103.

Kronvang, B., Bechmann, M., Lundekvam, H., Behrendt, Heckrath, G.H., Rubaek, G., Schoumans, O., Syversen, N., Andersen, H. and Hoffmann, C.C. 2005. Phosphorus losses from agricultural areas in river basins: effects and uncertainties of targeted mitigation measures. Journal of Environmental Quality, 24, 2129–2144.

Langton, S. 2009. Change in the area and distribution of uncropped land in England: January 2009 update. Defra Agricultural Change and Environment Observatory Report No. 13.

Leader, J.W., Dunne, E.J. and Reddy, K.R. 2008. Phosphorus sorbing materials: sorption dynamics and physicochemical characteristics. Journal of Environmental Quality, 37, 174-181.

Lough Neagh Advisory Committee 2008. Lough Neagh Management Strategy. [Online] Available at: http://www.loughneagh.com/publications.html [Accessed June 2010].

Lutz, F.G. 2010. Detecting silica deposition changes in diatoms due to variable nutrient concentrations using a fluorescent silica tracer in the waters of Barkley Sound, Canada. [Online] Available at: https://digital.lib.washington.edu/researchworks/handle/1773/16121 [Accessed June 2010]

Maberly, S.C., King, L., Dent, M.M., Jones, R.I., Gibson, C.E. 2002. Nutrient limitation of phytoplankton and periphyton growth in upland lakes. Freshwater Biology, 47, 2136-2152.

Maguire, R.O., G.H. Rubæk, B.E. Haggard, and R.H. Foy. 2009. Critical evaluation of the implementation of mitigation options for phosphorus from field to catchment scales. Journal of Environmental Quality, 38, 1989-1997.

Maguire, R.O., Crouse, D.A. and Hodges, S.C. 2007. Diet modification to reduce phosphorus surpluses: A mass balance approach. Journal of Environmental Quality, 36, 1235-1240.

Maguire, R.O., Chardon, W.J. and Simard, R.R. 2005. Assessing potential environmental impacts of soil phosphorus by soil testing. In: Sims, J.T. and Sharpley, A.N., (eds.). Phosphorus: Agriculture and the Environment. American Society of Agronomy Monograph 46, Madison, Wisconsin. pp. 145-180.

Makarewicz, J.C., Lewis, T.W., Bosch, I., Noll, M.R., Harendeen, N., Simon, R.D., Zollweg, J. And Vodacek, A. 2009. The impact of agricultural best management practices in downstream systems: Soil loss and nutrient chemistry and flux to Conesus Lake, New York, USA. Journal of Great Lakes Research, 35, 23-36.

Mainstone, C.P. and Parr, W. 2002. Phosphorus in rivers – ecology and management. Science of the Total Environment, 282-283, 25-47.

Mainstone, C.P., Dils, R.M., Withers, P.J.A. 2008. Controlling sediment and phosphorus transfer to receiving waters - A strategic management perspective for England and Wales. Journal of Hydrology, 350, 131-143.

Mainstone, C.P., Parr, W. and Day, M. 2000. Phosphorus and River Ecology: Tackling sewage inputs. English Nature, Peterborough.

May, L., Spears, B., Place, C. and O'Malley, M. 2010. The impact of phosphorus inputs from small discharges on designated freshwater sites. Final report to Natural England.

McDowell, R.W., Littlejohn, R.P. and Blennerhassett, J.D. 2010. Phosphorus fertilizer form affects phosphorus loss to waterways: a paired catchment study. Soil Use and Management, 26, 365-373.

McDowell, R.W., Nash, D., George, A., Wang, Q.J. and Duncan, R. 2009. Approaches for quantifying and managing diffuse phosphorus exports at the farm/small catchment scale. Journal of Environmental Quality, 38, 1968-1980.

McDowell, R.W., Sharpley, A.N. and Bourke, W. 2008. Treatment of drainage water with industrial by-products to prevent phosphorus loss from tile-drained land. Journal of Environmental Quality, 37, 1575-1582.

McDowell, R.W., Nash, D.M. and Robertson, F. 2007. Sources of phosphorus lost from a grazed pasture receiving simulated rainfall. Journal of Environmental Quality, 36, 1281-1288.

McDowell, R.W. and Wilcock, R.J. 2007. Sources of sediment and phosphorus in stream flow of a highly productive dairy farmed catchment. Journal of Environmental Quality, 36, 540-548.

McDowell, R.W. and Catto, W. 2005. Alternative fertilizers to prevent incidental phosphorus loss. Environmental Chemistry Letters, 2, 169-174.

McDowell, R.W., Biggs, B.J.E., Sharpley, A.N. and Nguyen, L. 2004. Connecting phosphorus loss from agricultural landscapes to surface water quality. Chemistry and Ecology, 20, 1-40.

McDowell, R.W., Drewery, J.J., Muirhead, R.W. and Paton, R.J. 2003. Cattle treading and phosphorus and sediment loss in overland flow from grazed cropland. Australian Journal of Soil Research, 41, 1521-1532.

Michigan Department of Transport 1998. Highway Stormwater Runoff Study. [Online] Available at: http://www.michigan.gov/documents/MDOT_MS4_MDOT_Hwy_SW_Runoff_Study_91946_7.pdf [Accessed March 2011]

Mitchell, G. 2001. The quality of urban stormwater in Britain and Europe: Database and recommended values for strategic planning models. Technical Report, School of Geography, University of Leeds, Leeds, UK.

Monaghan, R.M. and Smith, L.C. 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils II. The contribution of preferential flow of effluent to whole-farm pollutant losses in sub-surface drainage from a West Otago dairy farm. New Zealand Journal of Agricultural Research, 47, 417-429.

Monaghan, R.M., Wilcock, R.J., Smith, L.C., Tikkisetty, B., Thorrold, B.S. and Costall, D. 2007. Linkages between land management activities and water quality in an intensively farmed catchment in southern New Zealand. Agriculture Ecosystems and the Environment, 118, 211-222.

Moss, B. 2008. Water pollution by agriculture. Philosophical Transactions of the Royal Society B-Biological Sciences, 363, 659-666.

Nair, V.D., Portier, K.M., Graetz, D.A. and Walker, M.L. 2004. An environmental threshold for degree of phosphorus saturation in sandy soils. Journal of Environmental Quality, 33, 107-113.

Natural England 2006. Target 2010: The condition of England's Sites of Special Scientific Interest in 2005. [Online] Available at: http://naturalengland.etraderstores.com/NaturalEnglandShop/ST120 [Accessed June 2010].

Natural England 2009. Sites of Special Scientific Interest. [Online] Available at: http://www.naturalengland.org.uk/ourwork/conservation/designatedareas/sssi/default.aspx [Accessed June 2010].

Neal, C., Jarvie, H.P., Withers, P.J.A., Whitton, B.A., Neal, M. 2010. The strategic significance of wastewater sources to pollutant phosphorus levels in English rivers and to environmental management for rural, agricultural and urban catchments, Science of the Total Environment, 408, 1485-1500.

Neal, C., Jarvie, H.P., Neal, M., Love, A.J., Hill, L. and Wickham, H. 2005. Water quality of treated sewage effluent in a rural area of the upper Thames Basin, southern England, and the impacts of such effluents on riverine phosphorus concentrations. J Hydrology, 304, 103-117.

Needelman, B.A., Gburek, W.J., Petersen, G.W., Sharpley, A.N. and Kleinman, P.J.A. 2004. Surface runoff along two agricultural hillslopes with contrasting soils. Soil Science Society of America Journal, 68, 914-923.

NERC 1998. Hydrometric Register and Statistics 1991-95. Natural Environment Research Council, Oxon. 207 pp.

Newell Price, J.P., Harris, D., Taylor, M., Williams, J.R., Anthony, S.G., Duethmann, D., Gooday, R.D., Lord, E.I., Chambers, B.J., Chadwick, D.R. and Misselbrook, T.H. 2011. An inventory of methods and their effects on diffuse water pollution, greenhouse gas emissions and ammonia emissions from agriculture. Defra project WQ0106. Work Package 5. Final Report.

Odum, E.P., Finn, J.T. and Franz, E.H. 1979. Perturbation-theory and the subsidy-stress gradient. Bioscience, 29, 349-352.

Office for National Statistics 2010. Population change. [Online] Available at: http://www.statistics.gov.uk/cci/nugget.asp?id=950 [Accessed June 2011]

Ofwat 2004. Environment, Food and Rural Affairs Committee's inquiry into sites of special scientific interest: Memorandum by the Office of Water Services. [Online] Available at:http://www.ofwat.gov.uk/legacy/aptrix/ofwat/publish.nsf/AttachmentsByTitle/SSSIevidence260704.doc/$FILE/SSSIevidence260704.doc [Accessed June 2010].

Oliver, M.A., Archer, J.R., Baxter, S.J., Todd, A.D. and Skinner, R.J. 2006. The Representative Soil Sampling Scheme of England and Wales: a statistical analysis of topsoil pH and nutrient status between 1971 and 2001. Soil Use and Management, 22, 372-382.

Orr, H.G., Wilby, R.L., Mckenzie Hedger, M. and Brown, I. 2008. Climate change in uplands: a UK perspective on safeguarding regulatory ecosystem services. Climate Research, 37, 77-98.

Page, T., Haygarth, P., Beven, K.J., Joynes, A., Butler, T., Keeler, C., Freer, J., Owens, P. and Wood, G., 2005. Spatial variability of soil phosphorus in relation to the Topographic Index and

Critical Source Areas: sampling for assessing risk to water quality. Journal of Environmental Quality, 34, 2263-2277.

Palmer-Felgate, E.J., Mortimer, R.J.G., Krom, M.D. and Jarvie, H.P. 2010. Impact of point-source pollution on phosphorus and nitrogen cycling in stream-bed sediments. Environmental Science and Technology, 44, 908-914.

Phillips, G., Kelly, A., Pitt, J., Sanderson, R. and Taylor, E. 2005. The recovery of a very shallow eutrophic lake, 20 years after the control of effluent derived phosphorus. Freshwater Biology, 50, 1628-1638.

Pionke, H.B., Gburek, W.J. and Sharpley, A.N. 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. Ecological Engineering, 14, 325-526.

Pretty, J.N., Mason, C.F., Nedwell, D.B., Hine, R.E., Leaf, S. and Dils, R. 2003. Environmental costs of freshwater eutrophication in England and Wales. Environmental Science and Technology, 32, 201-208.

Ranganath, S.C., Hession, W.C. and Wynn, T.M. 2009. Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. Journal of Soil and Water Conservation, 64, 33-42.

Reijneveld, J.A., Ehlert, P.A.I., Termorshuizen, A.J. and Onema, O. 2010. Changes in soil phosphorus status of agricultural land in the Netherlands during the 20th century. Soil Use and Management, 26, 399-411.

Reynolds, C.S. and Davies, P.S. 2001. Sources and bioavailability of phosphorus fractions in freshwaters: a British perspective. Biological Reviews of the Cambridge Philosophical Society, 76, 27-64.

Reynolds, C.S., Irish, A.E., Elliott, J.A. 2001. The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). Ecological Modelling, 140, 271291.

Rogner, M. 2009. Phosphorus Recovery from Sewage Treatment Works: Executive Summary of MSc Thesis. [Online] Available at: https://workspace.imperial.ac.uk/environmentalpolicy/public/Rogner_Exec_Summary_09.pdf [Accessed June 2011]

Rounsevell, M.D.A., Ewert, F., Reginster, I., Leemans, R., Carter, T.R., 2005. Future scenarios of European agricultural land use. II: projecting changes in cropland and grassland. Agriculture, Ecosystems and the Environment, 107, 117-135.

RPA, 2003. Water Framework Directive – Indicative Costs of Agricultural Measures Assessment. Report to Defra, Risk and Policy Analysts Limited, Loddon, Norfolk.

RPA, 2010. Single Payment Scheme. Cross Compliance Guide for Soil Management. 2010 edition. Available at: http://rpa.defra.gov.uk/rpa/index.nsf/0/2ba694d4a8a991478025768e005e67c0/$FILE/Cross%20Compliance%20Guide%20to%20Soil%20Management%202010%20edition.pdf

Schärer, M., Page, T. and Beven, K. 2006. A fuzzy decision tree to predict phosphorus export at the catchment scale. Journal of Hydrology, 331, 484-494.

Schelde, K., de Jonge, L.W., Kjaergaard, C., Laegdsmand. M. and Rubaek, G.H. 2006. Effects of manure application and plowing on transport of colloids and phosphorus to tile drains. Vadose Zone Journal, 5 (1), 445-458.

Schmit, C., Rounsevell, M.D.A., RPA (Rural Payments Agency). 2010. Protection of hedgerows and watercourses (GAEC 14). [Online] Available at: http://rpa.defra.gov.uk/rpa/index.nsf/293a8949ec0ba26d80256f65003bc4f7/e2ef65c93d5a9199802573aa0053e7fb!OpenDocument [Accessed July 2011].

Schulte, R.P.O., Melland, A.R., Fenton, O., Herlihy, M., Richards, K. and Jordan, P. 2010. Modelling soil phosphorus decline: Expectations of Water Framework Directive policies. Environmental Science and Policy, 13, 472-484.

Shand, P., Edmunds, W.M., Lawrence, A.R., Smedley P.L. and Burke, S. 2007. The natural (baseline) quality of groundwater in England and Wales. Environment Agency Science Group: Air, Land and Water, Technical Report NC/99/74/24

Sharpley, A.N., Foy, B. and Withers, P.J.A. 2000. Practical and innovative measures for the control of agricultural phosphorus losses to water: An overview. Journal of Environmental Quality, 29, 1-10.

Sharpley, A.N., Kleinman, P.J.A., Jordan, P., Bergstrom, L. and Allen, A.L. 2009. Evaluating the success of phosphorus management from field to watershed. Journal of Environmental Quality, 38, 1981-1988.

Sharpley, A.N., Kleinman, P J.A., Heathwaite, A.L., Gburek, W.L., Folmar, G.L. and Schmidt, J.P. 2008. Phosphorus loss from an agricultural watershed as a function of storm size. Journal of Environmental Quality, 37, 362-368.

Shepherd, M.A. and Withers, P.J.A. 1999. Application of poultry litter and triple superphosphate fertilizer to a sand soil: effects on soil phosphorus status and profile distribution. Nutrient Cycling in Agroecosystems, 54, 233-242.

Shepard, R. 2005. Nutrient management planning: Is it the answer to better management? Journal of Soil and Water Conservation, 60, 171-176.

Scholz, M., Harrington, R., Carroll, P. And Mustafa, A. 2007. The integrated constructed wetlands (ICW) concept. Wetlands, 27, 337-354.

Sims, J.T. and Kleinman, P.J.A. 2005. Managing agricultural phosphorus for environmental protection. In: Sims J.T. and Sharpley, A.N. (eds.), Phosphorus; Agriculture and the Environment. American Society of Agronomy Monograph No. 46, American Society of Agronomy, Madison, WI. pp. 1021-1068.

Skinner, R.J. and Todd, A.D. 1998. Twenty-five years of monitoring pH and nutrient status of soils in England and Wales. Soil Use and Management, 14, 162-169.

Smith, K.A., Jackson, D.R. and Withers, P.J.A. 2001. Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus. Environmental Pollution, 112, 53-60.

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

Stephens, C.J. and Quinton, J.N. 2010. Diffuse pollution swapping in arable agricultural systems. Critical Reviews in Environmental Science and Technology, 39(6), 478-520.

Stephens, C.J., Quinton, J.N., Bailey, A.P., Deasy, C., Silgram, M. and Jackson, D.R. 2009. The effects of minimal tillage, contour cultivation and in-field vegetative barriers on soil erosion and phosphorus loss. Soil and Tillage Research, 106, 145-151.

Stutter, M. and Demars, B. no date. Understanding and managing the range of phosphorus forms contributing to eutrophication. [Online] Available at: http://www.life-m3.eu/fileadmin/M3-life/downloads/presentations-workshop2010/M3-session3-Stutter.pdf [Accessed March 2011].

Stutter, M.I., Langan, S.J. and Lumsdon, D.G. 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: a biogeochemical assessment of the mechanisms. Environmental Science and Technology, 43, 1858-1863.

Tanner, C.C., Nguyen, M.L. and Sukias, J.P.S. 2005. Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. Agriculture, Ecosystems and the Environment, 105, 145-162.

Taylor Lovell, S. and Sullivan, W.C. 2006. Environmental benefits of conservation buffers in the United States: Evidence, promise, and open questions. Agriculture, Ecosystems and the Environment, 112, 249-260.

Torbert, H.A., Potter, K.N., Hoffman, D.W., Gerik, T.J. and Richardson, C.W. 1999. Surface residue and soil moisture affect fertilizer loss in simulated runoff on a heavy clay soil. Agronomy Journal, 91, 606-612.

UKTAG 2008a. UK Environmental Standards and Conditions (Phase 1) Final Report. [Online] Available at: http://www.wfduk.org/UK_Environmental_Standards/LibraryPublicDocs/UKTAG%20ReportAug%202006UKEnvironmentalStandardsandConditionsFinalReport [Accessed June 2010].

UKTAG 2008b. UK Environmental Standards and Conditions (Phase 2) Final Report. [Online] Available at: http://www.wfduk.org/UK_Environmental_Standards/LibraryPublicDocs/UKTAG_Report_Surface_Water_Standards_and_Conditions [Accessed June 2010].

UKTAG 2008c. UKTAG Lake Assessment Methods, Macrophytes and Phytobenthos: Diatom Assessment of Lake Ecological Quality (DARLEQ). [Online] Available at: http://www.wfduk.org/bio_assessment/lakes_phytobenthos_darleq [Accessed March 2010].

Ulén, B. and Jakobsson, C. 2005. Critical evaluation of measures to mitigate phosphorus losses from agricultural land to surface waters in Sweden. Science of the Total Environment, 344, 37-50.

Ulén, B., Bechmann, M. Fölster, J., Jarvie, H.P. and Tunney, H. 2007. Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. Soil Use and Management, 23 (Supplement 1), 5-15.

Ulén, B., Aronsson, H., Bechmann, M., Krogstad, T., Øygarden, L. and Stenberg, M. 2010. Soil tillage methods to control phosphorus loss and potential side-effects: a Scandinavian review. Soil Use and Management, 26, 94-107.

US Department of Transportation 1999. Is highway runoff a serious problem? [Online] Available at: http://www.tfhrc.gov/hnr20/runoff/runoff.htm [Accessed March 2011].

Uussi-Kamppa, J., Braskerut, B., Jansson, H., Syverson, N. and Uusitalo, R. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. Journal of Environmental Quality, 29, 151-158.

Van der Salm, C., Chardon, W.J., Koopmans, G.F., van Middlekoop, J.C. and Ehlert, P.A.I. 2009. Phytoextraction of phosphorus-enriched grassland soils. Journal of Environmental Quality, 38, 751-761.

Watson, C.J., Smith, R.V. and Matthews, D.I. 2007. Increase in phosphorus losses from grassland in response to Olsen-P accumulation. Journal of Environmental Quality, 36, 1452-1460.

White, P.J. and Hammond, J.P. 2009. The sources of phosphorus in the waters of Great Britain. Journal of Environmental Quality, 38, 13-26.

Wiederholt, R. and Johnson, B. 2005. Phosphorus behaviour in the environment. North Dakota State University Extension Service leaflet NM-1298.

Willby, N.J., Pitt, J and Phillips, G. L. 2009. Development of a system for the classification of rivers and lakes in the UK using aquatic macrophytes. Part 2 Rivers. Environment Agency Science Report.

Withers, 2008. Effects of farming practices on phosphorus transfer in rural catchments. PhD Thesis, January 2008.

Withers, P.J.A. 2012. Agronomic and environmental impacts of phosphorus in biosolids applied to agricultural land – a review of UK research. Critical Reviews in Environmental Science and Technology (submitted).

Withers, P.J.A. and Bailey, G.A. 2003. Sediment and phosphorus transfer in overland flow from a maize field receiving manure. Soil Use and Management, 19, 28-35.

Withers, P.J.A. and Hodgkinson, R.A. 2009. The effect of farming practices on phosphorus transfer to a headwater stream in England. Agriculture, Ecosystems and the Environment, 131, 347-355.

Withers, P.J.A. and Jarvie H.P. 2008. Delivery and cycling of phosphorus in rivers: a review. Science of the Total Environment, 400, 379-395.

Withers, P.J.A. and Jarvis, S.C. 1998. Mitigation options for diffuse phosphorus loss to water. Soil Use and Management, 14, 186-192.

Withers, P.J.A. and Lord, E.I. 2002. Agricultural nutrient inputs to rivers and groundwaters in the UK: policy, environmental management and research needs. Science of the Total Environment, 282-283, 9-24.

Withers, P.J.A., Edwards, A.C. and Foy, R.H. 2001. Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. Soil Use and Management, 17, 139-149.

Withers, P.J.A., Jarvie, H.P. and C. Stoate, C. 2011. Quantifying the impact of septic tank systems on eutrophication risk in rural headwaters. Environment International, 37, 644-653.

Withers, P.J.A., Hodgkinson, R.A., Bates, A. and Withers, C.L. 2007. Soil cultivation effects on sediment and phosphorus mobilisation in surface runoff from three contrasting soil types in England. Soil and Tillage Research, 93, 438-451.

Withers, P.J.A., Ulén, B., Stamm, C. and Bechmann, M. 2003. Incidental phosphorus loss – is it significant and can it be predicted? Journal of Soil Science and Plant Nutrition, 166, 459-468.

Withers, P.J.A., Hartikainen, H., Barberis, E., Flynn, N.J. and Warren, G.P. 2009b. The effect of soil phosphorus on particulate phosphorus in land runoff. European Journal of Soil Science, 60, 994-1004.

Withers, P.J.A., Jarvie, H.P., Hodgkinson, R.A., Palmer-Felgate, E.J., Bates, A., Neal, M., Howells, R., Withers, C.M. and Wickham, H.D. 2009a. Characterization of phosphorus sources in rural watersheds. Journal of Environmental Quality, 38, 1998-2011.

Withers, P.J.A., Hodgkinson, R.A., Jarvie, H.P., Sutton, E.J., Harper, D., Kwiatkowska, K., Wasiak, P., Stoate, C. and Foy, R.H. 2009c. Sources of nutrients in rural lowland headwater catchments and their ecological significance. In: Crighton, K and Audsley, R. (eds)., Land Management in a Changing Environment, SAC and SEPA Biennial Conference, 26-27 March 2008, pp. 109-113.

WRc. 2006. Production of SIMCAT model structures for England and Wales: Final Report. WRc Report UC7189 to the Environment Agency.

WRc 2007. Cost-benefit analysis of potential solutions for point and diffuse source pollution to achieve WFD ‘Good’ status: Ribble SIMCAT pilot study. WRc Report WW17C205.

WRc 2009. Effectiveness of phosphate removal by sewage treatment works. WRc Report No.SC060107/SR to the Environment Agency.

WRc 2010. Review of evidence for use of phosphate-free dishwasher detergents in UK. WRc Report No. Defra 8252.02 to Defra.

Zhang, G.S., Chan, K.Y., Oates, B.A., Heenan, D.P. and Huang, G.B. 2007. Relationship between soil structure and runoff/soil loss after 24 years of conservation tillage. Soil and Tillage Research, 92, 122-128.

Appendix A – Catchment Walkovers

Three catchments (Linnet, Harpers Brook and Bourn Brook) in East Anglia with potentially significant export of agricultural diffuse phosphorus (P) were visited over a two- to three-day period to identify potential sources of P and, in conjunction with information on landscape attributes (topography, soil type), river flows and P data and catchment modelling (PSYCHIC) outputs, help assess the contribution of farming activities to riverine P loads. The overall purpose of the visits was to determine whether a targeted campaign to adopt more sensitive farming practices and P mitigation practices in the catchment would reduce ambient river P concentrations and help meet P targets set by the Water Framework Directive (WFD). The visits were made in September 2010 (Linnet and Harpers Brook) and in January 2011 (Bourn Brook).

Starting at the catchment outlet, upstream sections of each river were surveyed to assess the type of farming (crops, cultivation type), physical landscape features (slope, presence of buffer features), proximity of impervious surfaces (farmyards, tracks, roads), proximity and predominance of urban development (villages, towns, sewage treatment works, septic tanks) and evidence of runoff pathways (surface runoff and drainflow). In each catchment, at least one farmer was interviewed for more precise details of catchment characteristics, the historical development of farming activities (crops and livestock) and on current soil P levels and P inputs.

Linnet (15-16 September 2010)

The Linnet is a largely chalkland catchment close to Bury St Edmunds with workable soils, some steep slopes and low stream density. Small areas of underdrained soils also exist. The catchment walk was undertaken in dry weather and although the catchment outlet in the town of Bury St Edmunds had flow, a significant portion of the stream just upstream of the town was completely dry. There appeared to be no surface connectivity between the upper catchment area and the catchment outlet. Interviews with two farmers and local residents suggested the stream regularly runs dry and flow rates in the upper catchment area were very low at the time of the visit. A brewery in Bury St Edmunds abstracts water from the groundwater. The Linnet feeds into the River Lark.

Historically the catchment area has been dominated by livestock farming, particularly dairy and indoor and outdoor pigs. One farmer interviewed still keeps indoor pigs but in lower numbers than previous years and all the dairy farms have now gone out of production. Soil P levels are high in some areas (up to P Index 5) due to historical inputs of P. On the pig farm, slurry is recycled on a rotational basis. Currently, fields are cropped to grass, cereals and maize. Many fields in the river corridor are in grass. On cultivated land, tramlines invariably run downslope, even on steeper land, but we found no visible signs of erosion and most fields had 6-10 m buffer strips along the flatter riparian margins. Adjacent tracks and roads are considered the main source of surface runoff entering the stream.

A small STW (Chedburgh) discharges (P concentration ofaround 5 mg l^{-1} based on Environment Agency data) into the headwater of the Linnet catchment and under the low flow conditions encountered during the visit, is the main source of P to the stream. Few houses have septic tanks directly next to the river. Small industrial units (fertiliser factory, clothing factory) were operating in the past but these are not thought to be discharging now.

The location of the STW at the head of the catchment and the low summer flow conditions suggest point sources are the dominant source controlling stream P concentrations. Surface runoff is the main pathway of diffuse P loss and runoff would be P enriched due to P release from the highly fertile soils. However, the proportion of field surface runoff entering the Linnet is considered small due to the free-draining nature of the soils and the presence of wide riparian buffer strips along most of the river corridor. The main source of surface runoff into the stream is likely to be road and track runoff. The source of the water being monitored at the catchment outlet in Bury St Edmunds is not clear.

Harpers Brook (16-18 September 2010)

Harpers Brook is a clay catchment overlying limestone close to the town of Corby with a dense network of ditches draining the land area. The catchment walk was undertaken in dry weather but many of the ditches were still running. There are a large number of natural springs (often located in woodland) and interviews with three farmers in the locality suggests the Brook responds rapidly to rainfall. The Brook has been hydrologically modified in the past and at least one farmer considered surface runoff from recent urban development in Corby to be responsible for much quicker flows and response than previously. The central part of the catchment has historically had opencast and mined ironstone workings although this stopped in about 1940. A large part of the catchment has been underdrained. A number of STW are present but chubb, trout and crayfish are fairly abundant in the Brook and water quality is generally considered by the people interviewed to have improved in recent years. Regular flushing during storm events helps to maintain a gravel bottom in many river sections. The Brook feeds into the Titchmarsh nature reserve before entering the River Nene. An Environment Agency gauging station is present close to the catchment outlet.

The catchment is dominated by arable farming with cereals, oilseed rape and beans. There are only a few livestock (dairy/beef/sheep) farms, and at least one of these is now in organic production. Dairy farming was more extensive in the past. Only small amounts of manure are therefore recycled. Soil P levels are not considered to be high (P Index 1/2) but little information is available to confirm this. Topography is slightly undulating. Most fields in the river corridor are cultivated, with tramlines running both across slope and down slope. Minimal cultivation is widely practised. Riparian buffer strips varying from 2-10 m are widespread (mostly 6 m). At one point, there was a tractor entry point into the Brook.

Several small STW (e.g. Brigstock, Stanion) discharge (P concentration up to 15 mg l^{-1}based on Environment Agency data) into the Brook and only a few isolated houses are probably on septic tank systems. Other small consented discharges are present.

Drainflow is the dominant pathway of diffuse P loss in the catchment and drains run turbid during storm events, especially in autumn. Given the high ditch density, farming activities have the potential to influence ambient P concentrations within the river but no information is available on the likely range in P concentrations being discharged from land drains. High flow volumes during storm events help to flush out any sediment accumulated within the stream channel and will dilute inputs of P from STW and other consented discharges. The urban development at Corby appears to be contributing to the stream.

Bourn Brook (10-13 January 2011)

The upper catchment area surrounding the villages of Bourn and Caxton and upstream of their STW was the main survey area, although the wider catchment area down to Grantchester was also investigated. The catchment walk was undertaken in mostly wet weather. Stream density is high with numerous first order tributaries flowing into the main Bourn Brook which flooded during the visit following heavy overnight rain. The upper catchment includes the new village of Cambourne which is considered to have greatly increased the flashiness of the Brook's response in flow to rainfall. One farmer in the lower catchment area was interviewed. There are two golf courses in the catchment.

The Upper Bourn catchment is dominated by underdrained chalky boulder clay soils growing winter cereals/winter oilseed rape/winter bean rotations on flat to slightly undulating terrain. Smaller areas of lighter land grow a wider range of crops (including potatoes and sugar beet) in the lower catchment area. No information is available on soil P levels but these are likely to be modest (P Index 1/2) given the soil type, although one farmer in the lower catchment has applied biosolids (sewage sludge) in the past and one farmer in the upper catchment area has had pigs for a number of years. Discussions with the farmer suggest most P fertiliser is incorporated before drilling rather than top-dressed. The catchment has a history of mainly arable production.

The majority of fields are minimally cultivated, tramlines occur both across slope and down slope and six-metre riparian strips are fairly ubiquitous. Field margins adjacent the Brook collect standing water during wet weather but there is no evidence that this is flowing into the streams. One notable exception in the lower catchment area is where riparian fields are cultivated up to the river margins and the two-metre cross-compliance buffer strip has largely disintegrated in the field corner. A large area of this field was flooded on day two of the visit. Surface runoff inputs into the Brook are therefore largely confined to overflow from rural tracks and roads, but these do not have a high density due to the large size of the fields.

Sub-surface drainflow is the dominant pathway of P loss from farmed land and land drains run turbid during heavy rain, especially in autumn. One pig farm close to the Brook has been using a low rate irrigation system to recycle pig slurry. This practice is considered to be high-risk due to the dilute nature of pig slurry and the presence of cracking and underdrained clay soils.

The Brook flows directly through the villages of Caxton and Bourn in the Upper catchment. There are numerous 'pipes' in the villages discharging directly to the river. It is not clear where these originate but are likely to be yard or roof runoff or greywater discharges. Runoff from the road network in the villages is a major input to the Brook. There are two fords in the upper catchment area which were both heavily flooded during the visit. In the lower catchment area, a number of STW discharge directly into the Brook.

Overall, agriculture is not considered to be the main source of P to the Brook but there are small areas of high-risk farming activity which probably contribute additional P to the general background sediment-associated P in drainflow during the winter drainage period. Recycling of pig slurry to cracked soils during summer/autumn may also cause localised increases in stream P concentrations.

Lessons learnt

1. Catchment walks provide only a snapshot of potential and actual landscape-runoff interactions but nevertheless allow the capture of considerable on-site information that could not be gathered from desk-based catchment appraisal activity. Information on the historical context gained through interviews with local farmers and residents and visual assessment of stream flow conditions and farming practices was particularly valuable. This information was gathered in a relatively short space of time, with 15 km^2 being covered in one day.

2. Prior to each visit, river P concentration data held by the Environment Agency was evaluated in relation to flow to identify potential point and diffuse source contributions. In each catchment, river P concentrations declined with increasing flow suggesting a

dominance of point source contributions. However, the range in flows sampled by the Environment Agency is probably insufficient to enable diffuse source contributions (which increase with flow) to be adequately assessed. The provision of existing data and the ability to evaluate these data using new modelling approaches are considered essential prior to site visits.

3. Maps of the spatial distribution of modelled P loss (point versus diffuse, surface versus drainflow, hard standing and so on in kg) based on the PSYCHIC model were also provided during the visit, but these were of relatively little use without aerial (kg ha^{-1}) estimates of P loss for the catchment as a whole to assess the relative contribution of sources/pathways and causative factors.

List of abbreviations

AMP	Asset Management Plan
BAT	Best Available Technology
BFI	Base Flow Index
BOD	Biological Oxygen Demand
CAP	Common Agricultural Policy
CCW	Countryside Council for Wales
CEA	Cost Effectiveness Analysis
CIS	Common Implementation Strategy
COD	Chemical Oxygen Demand
CSA	Critical Source Area
CSF	Catchment Sensitive Farming
CSO	Combined Sewer Overflow
Defra	Department for Environment, Food and Rural Affairs
DHP	Dissolved Hydrolysable Phosphorus
DPI	Diffuse Pollution Inventory
EA	Environment Agency
EC	European Community/Commission
ECSFDI	England Catchment Sensitive Farming Delivery Initiative
ELS	Entry Level Stewardship
EMC	Event Mean Concentration
EPR	Environmental Permitting Regulations
EQR	Ecological Quality Ratio
EU	European Union
GAEC	Good Agricultural and Environmental Conditions
GP3	“Groundwater Protection: Policy and Practice” documents
GQA	General Quality Assessment
GWD	Groundwater Directive (2006/118/EC).
HD	Habitats Directive
HMWB	Heavily Modified Water Bodies
HLS	Higher Level Stewardship
IPPC	Integrated Pollution and Control Directive
MTR	Mean Trophic Rank
NFU	National Farmers’ Union
NVZ	Nitrate Vulnerable Zone
Ofwat	Water Services Regulation Authority
OSWTS	On-Site Wastewater Treatment System
PAO	Phosphorus Accumulating Organisms
pCEA	Preliminary cost effective analysis
PE	Population Equivalent
PoMs	Programme of Measures
PP	Particulate Phosphorus
PPC	Pollution Prevention and Control

PR09	Periodic Review in 2009
RBC	River Basin Characterisation
RBD	River Basin District
RBMP	River Basin Management Plan
RMNI	River Macrophyte Nutrient Index
SA	Sensitive Area
SAC	Special Area of Conservation
SIMCAT	Simulated Catchment Model
SCV	Species Cover Value
SPA	Special Protection Area
SPZ	Source Protection Zone
SRP	Soluble Reactive Phosphorus
SSAFO	Silage, Slurry and Agricultural Fuel Oil Regulations
SSSI	Site of Special Scientific Interest
STP	Soil Test Phosphorus
STR	Species Trophic Rank
STW	Sewage Treatment Works
SUDS	Sustainable Drainage Systems
TDI	Trophic Diatom Index
TP	Total Phosphorus
TRP	Total Reactive Phosphorus
UKTAG	United Kingdom Technical Advisory Group
UKWIR	United Kingdom Water Industry Research
UWWTD	Urban Wastewater Treatment Directive
WFD	Water Framework Directive
WPZs	Water Protection Zones
WwTW	Wastewater Treatment Works

Glossary

Term	Explanation
Agri-environment scheme	Land management schemes on farmland that are beneficial, for example, to the environment, natural resources, biodiversity, landscape.
Alien species	Non-native species. Many species of plants and animals have been introduced to this country since Roman times. Several of these non-native species are invasive and have been causing serious problems to the aquatic and riverine ecology and environment. Problems include detrimental effects on our native species, deoxygenation of water causing fish mortalities, blocking of rivers and drainage channels, predation and competition with our native species, and in some cases health risks to the public or livestock.
Aquifer	A subsurface layer or layers of rock or other geological strata of sufficient porosity and permeability to allow either a significant flow of groundwater or the abstraction of significant quantities of groundwater.
Artificial water body	A man-made surface water body, rather than a modified natural water body, which supports important aquatic ecosystems. It includes canals, some docks and some man-made reservoirs.
Asset Management Plan	See Periodic Review.
Biological element	A collective term for a particular characteristic group of animals or plants present in an aquatic ecosystem (for example phytoplankton; benthic invertebrates; phytobenthos; macrophytes; macroalgae; angiosperms; fish).
Biological indicators	A parameter that can be monitored to estimate the value of a biological quality element. Indicators may include the presence or absence of a particularly sensitive species.
Biological quality element	A characteristic or property of a biological element that is specifically listed in Annex V of the Water Framework Directive for the definition of the ecological status of a water body (for example composition of invertebrates; abundance of angiosperms; age structure of fish).
Catchment	The area from which precipitation contributes to the flow from a borehole spring, river or lake. For rivers and lakes this includes tributaries and the areas they drain.
Catchment Abstraction Management Strategies	CAMS are developed to manage water resources at a local level. They provide information on water resources and licensing practice to allow the needs of abstractors, other water users and the aquatic environment to be considered in consultation with the local community and interested parties.

Catchment Flood Management Plans	These are strategic planning tools through which the Environment Agency seeks to work with other decision-makers within a river catchment to identify and agree policies for sustainable flood risk management.
Catchment modelling techniques	Methods used to describe and/or predict characteristics of a catchment. Traditionally, these have focused on natural processes or movement of pollutants but they can include other factors such as demographic, social and economic characteristics.
Characterisation (of water bodies)	A two-stage assessment of water bodies under the Water Framework Directive. Stage 1 identifies water bodies and describes their natural characteristics. Stage 2 assesses the pressures and impacts from human activities on the water environment. The assessment identifies water bodies at risk of not achieving the environmental objectives set out in the Water Framework Directive. The results are used to prioritise environmental monitoring and further investigations to identify water bodies where action is required.
Chemical status	The classification status for the water body against the environmental standards for chemicals that are priority substances and priority hazardous substances. Chemical status is recorded as good or fail. The chemical status classification for the water body, and the confidence in this (high or low), is determined by the worst test result.
Chemical status (surface waters)	The classification status for the surface water body. This is assessed by compliance with the environmental standards for chemicals that are listed in the Environmental Quality Standards Directive 2008/105/EC, which include priority substances, priority hazardous substances and eight other pollutants carried over from the Dangerous Substance Daughter Directives. Chemical status is recorded as good or fail. The chemical status classification for the water body, and the confidence in this (high or low), is determined by the worst test result.
Chemical status (groundwater)	An expression of the overall quality of the groundwater body. The classification status for a groundwater body against the environmental criteria set out in the Water Framework Directive and the Groundwater Directive (2006/118/EC), as set out in Common Implementation Strategy (CIS) Guidance Document No 18. All five of the component tests for chemical status must be assessed as good or poor and the overall chemical status and the confidence in this (high or low) is determined by the worst test result.
Common Agricultural Policy	A policy that regulates farming activities across the European Union, providing direct subsidies to farmers and land managers. A small part of these funds support rural development actions that mainly relate to agricultural activities, as well as forestry and environmental improvements on farmland.

Common Implementation Strategy (CIS)	This strategy was agreed by the European Commission, Member States and Norway in 2001. The aim of the strategy is to provide support in the implementation of the Water Framework Directive and its daughter directives, by developing a common understanding and guidance on key elements of the Directives.
Cost effective	In the context of the Water Framework Directive, it describes the least cost option for meeting an objective. For example, where a number of actions could be implemented to achieve good ecological status for a water body, cost effectiveness analysis is used to compare each of the options and identify which option meets the objective for the least overall cost.
Countryside Council for Wales	The Countryside Council for Wales is the Welsh Assembly Government's statutory adviser on sustaining natural beauty, wildlife and the opportunity for outdoor enjoyment in Wales and its inshore waters. The Countryside Council for Wales is the national wildlife conservation authority for Wales.
Diffuse pollution	Pollution resulting from scattering or dispersed sources that are collectively significant but to which effects are difficult to attribute individually.
Drinking Water Protected Areas	Bodies of water that are used or could be used in the future for the abstraction of water intended for human consumption.
Ecological potential	The status of a heavily modified or artificial water body measured against the maximum ecological quality it could achieve given the constraints imposed upon it by those heavily modified or artificial characteristics necessary for its use. There are five ecological potential classes for Heavily Modified Water Bodies/Artificial Water Bodies (maximum, good, moderate, poor and bad).
Ecological status	Ecological status applies to surface water bodies and is based on the following quality elements: biological quality, general chemical and physico-chemical quality, water quality with respect to specific pollutants (synthetic and non-synthetic), and hydromorphological quality. There are five classes of ecological status (high, good, moderate, poor or bad).
Environment Agency	Environment Agency of England and Wales.
Estuarine	For our purposes, by estuarine we mean transitional (see definition).
Eutrophication	The enrichment of waters by inorganic plant nutrients that results in increased production of algae and/or other aquatic plants, which can affect the quality of the water and disturb the balance of organisms present within it.
Exemptions	The environmental objectives of the Water Framework Directive are set out in Article 4. These include the general objective of aiming to achieve good status in all water bodies by 2015 and the principle of preventing any further deterioration in status. There are a number of exemptions to the general objectives that allow for less stringent objectives, extension of deadline beyond 2015 or the implementation of new projects. Common to all these exemptions are strict conditions that must be met and a justification must be included in the river basin management plan.

	The conditions and process in which the exemptions can be applied are set out in Article 4.4, 4.5, 4.6 and 4.7.
Favourable conservation status	Favourable conservation status means to protect and, where necessary, improve the water or water-dependent environment to the extent necessary to maintain or restore water-dependent habitats and species for which the protected area is designated. Where this term is used in the River Basin Management Plans, the above definition applies.
Good chemical status (surface waters)	Concentrations of chemicals in the water body do not exceed the environmental standards specified in the Environmental Quality Standards Directive 2008/105/EC. These chemicals include priority substances, priority hazardous substances and eight other pollutants carried over from the Dangerous Substance Daughter Directives.
Good chemical status (groundwater)	See chemical status (groundwater). Concentrations of pollutants in the groundwater body do not exceed the criteria set out in Article 3 of the Groundwater Daughter Directive (2006/118/EC).
Good ecological potential	Surface waters identified as Heavily Modified Water Bodies and Artificial Water Bodies must achieve 'good ecological potential' (good potential is a recognition that changes to morphology may make good ecological status difficult to meet). In the first cycle of river basin planning, good potential may be defined in relation to the mitigation measures required to achieve it.
Good ecological status	The objective for a surface water body to have biological, structural and chemical characteristics similar to those expected under nearly undisturbed conditions.
Good quantitative status (groundwater)	See quantitative status (groundwater). Means the level of groundwater in the groundwater body meets the criteria set out in Annex V (2.1.2) of the Water Framework Directive.
Good status	The status achieved by a surface water body when both the ecological status and its chemical status are at least good or, for groundwater, when both its quantitative status and chemical status are at good status.
Groundwater	All water which is below the surface of the ground in the saturation zone and in direct contact with the ground or subsoil.
Hazardous substances	Substances or groups of substances that are toxic, persistent and liable to bioaccumulate, and other substances or groups of substances which give rise to an equivalent level of concern.
Heavily Modified Water Body	A surface water body that does not achieve good ecological status because of substantial changes to its physical character resulting from physical alterations caused by human use, and which has been designated, in accordance with criteria specified in the Water Framework Directive, as 'heavily modified'.
High ecological status	A state, in a surface water body, where the values of the hydromorphological, physico-chemical, and biological quality elements correspond to conditions undisturbed by anthropogenic activities.
Hydromorphology	Describes the hydrological and geomorphological processes and attributes of surface water bodies. For example for rivers, hydromorphology describes the form and function of the channel

	as well as its connectivity (up and downstream and with groundwater) and flow regime, which defines its ability to allow migration of aquatic organisms and maintain natural continuity of sediment transport through the fluvial system. The Water Framework Directive requires surface waters to be managed in such a way as to safeguard their hydrology and geomorphology so that ecology is protected.
Impact assessment	A tool to enable the Environment Agency to weigh and present the evidence on the positive and negative effects of a plan. For example, information on the estimated cost and benefit of proposed measures.
Macroalgae	Multicellular algae such as seaweed.
Macrophyte	Larger plants, typically including flowering plants, mosses and larger algae but not including single-celled phytoplankton or diatoms.
Measure	This term is used in the Water Framework Directive and domestic legislation. It means an action which will be taken on the ground to help achieve Water Framework Directive objectives.
Mechanisms	The policy, legal and financial tools which are used to bring about actions (measures). Mechanisms include for example: legislation, economic instruments; codes of good practice; negotiated agreements; promotion of water efficiency; educational projects; research; development and demonstration projects.
Misconnections	Misconnections of foul sewage into surface water drains are a significant source of urban diffuse pollution in those areas where a separate drainage system is used. Misconnections happen when domestic plumbing has been connected to surface water drains instead of the foul sewer. This means untreated dirty water goes directly into rivers/waterways without receiving treatment.
Morphology	Describes the physical form and condition of a surface water body, for example the width, depth and perimeter of a river channel, the structure and condition of the riverbed and bank.
National	This term refers, in this document, to England and Wales. The Environment Agency covers the whole of England and Wales and is the Competent Authority for the Water Framework Directive.
Natura 2000 sites	Protected areas established for the protection of habitats or species under the Birds Directive (79/409/European Economic Community) (Special Protection Areas) and the Habitats Directive (92/43/ European Economic Community) (Special Areas of Conservation).
Natural England	The government-funded body whose purpose is to promote the conservation of England's wildlife and natural features. The previously existing organisations English Nature, the Countryside Agency and Rural Development Service were merged to form Natural England.
Nitrate Vulnerable Zone	The land draining to waters that contain, or are likely to contain, 50 mg l^{-1} of nitrate, or waters that are eutrophic or likely to become so. Within these zones, an action programme under the Nitrates Directive is put in place which farmers have to observe to

	reduce nitrate pollution.
No deterioration (in water body status)	None of the quality elements used in the classification of water body status deteriorates to the extent that the overall status is reduced.
Non-native species	See alien species.
Objective (surface waters)	Three different status objectives for each water body. These are: overall status objective; ecological status or potential objective; and chemical status objective. These are always accompanied by a date by when the objective will be achieved. Ecological status (or potential) objectives are derived from the predicted outcomes for the biological elements and physico-chemical elements, plus any reasons for not achieving good ecological status (or potential) by 2015. Chemical status objectives are derived from the predicted outcomes for the chemical elements plus any reasons for not achieving good chemical status by 2015. Overall status objectives are derived from the ecological status and chemical status objectives.
Objective (groundwater)	There are three status objectives for each groundwater body: overall status objective; quantitative status objective; and chemical status objective. These are always accompanied by a date by when the objective will be achieved. Overall status objectives are derived from the quantitative status and chemical status objectives. In addition to status objectives, there are additional environmental objectives: to prevent deterioration of status, to prevent or limit the inputs of pollutants to groundwater and to reverse any significant and sustained upward trends in pollutant concentrations.
Office of Water Services	The economic regulator for the water and sewerage industry in England and Wales. Office of Water Services has been renamed the Water Services Regulation Authority.
Periodic Review	Process carried out every five years by the Office of Water Services to assess the strategic plans for water company spending and investment. The plans include environmental improvements. The investment will often affect water customer charges and incorporates company business plans (called Asset Management Plans).
Phytobenthos	Bottom-dwelling multi-cellular and unicellular aquatic plants such as some species of diatom.
Phytoplankton	Unicellular algae and cyanobacteria, both solitary and colonial that live, at least for part of their lifecycle, in the water column.
Point source pollution	Pollution arising from an identifiable and localised area, structure or facility, such as a discharge pipe or landfill.
Pollutant	Any substance liable to cause pollution.
Pollution	The direct or indirect introduction, as a result of human activity, of substances or heat into the air, water or land which: (i) may be harmful to human health or the quality of aquatic ecosystems or terrestrial ecosystems directly depending on aquatic ecosystems; (ii) result in damage to material property; or (iii) impair or interfere with amenities and other legitimate uses of the environment.

Pressures	Human activities such as abstraction, effluent discharges or engineering works that have the potential to have adverse effects on the water environment.
Priority substances	A pollutant, or group of pollutants, presenting a significant risk to or via the aquatic (surface water) environment that has been identified at European Community level under Article 16 of the Water Framework Directive. They include ‘priority hazardous substances’.
Programme of Measures	A Programme of Measures, as used in the Water Framework Directive, is a group of actions designed to improve the environment in a river basin district and meet the objectives of the Directive.
Protected Areas	Areas that have been designated as requiring special protection under European Community legislation for the protection of their surface water and groundwater or for the protection of habitats and species directly depending on water.
Quality element	A feature of an aquatic (surface water) ecosystem that can be described as a number for the purposes of calculating an ecological quality ratio, such as the concentration of a pollutant; the number of species of a type of plant.
Quantitative status (groundwater)	An expression of the degree to which a body of groundwater is affected by direct and indirect abstractions. The classification status for a groundwater body against the environmental criteria set out in the Water Framework Directive and as set out in Common Implementation Strategy Guidance Document No 18. All four of the component tests for quantitative status must be assessed as good or poor and the overall quantitative status and the confidence in this (high or low) is determined by the worst test result.
Ramsar site	A wetland area designated for its conservation value under The 1971 Convention on Wetlands of International Importance, especially as Waterfowl Habitat. The Ramsar Convention seeks to promote the conservation of listed wetlands and their wise use.
Reference conditions	The benchmark against which the effects on surface water ecosystems of human activities can be measured and reported in the relevant classification scheme. For waters not designated as heavily modified or artificial, the reference conditions are synonymous with the high ecological status class. For waters designated as heavily modified or artificial, they are synonymous with the maximum ecological potential class.
Risk	The likelihood of an outcome (usually negative) to a water body or the environment, or the potential impact of a pressure on a water body.
Risk assessment	The analysis that predicts the likelihood that a water body is at significant risk of failing to achieve one or more of the Water Framework Directive objectives.
Risk category	The numerical or descriptive category assigned to water bodies that have been risk assessed, in order to make the risk-based prioritisation of water bodies for action under the Water

	Framework Directive more manageable.
River basin	A river basin is the area of land from which all surface run-off and spring water flows through a sequence of streams, lakes and rivers into the sea at a single river mouth, estuary or delta. It comprises one or more individual catchments.
River Basin District	A river basin or several river basins, together with associated coastal waters.
River basin management	The management and associated planning that underpins implementation and operation of the Water Framework Directive.
River Basin Management Plan	For each River Basin District, the Water Framework Directive requires a River Basin Management Plan to be published. These plans set out the environmental objectives for all the water bodies within the River Basin District and how they will be achieved. The plans are based upon a detailed analysis of pressures on the water bodies and an assessment of their impacts. The plans must be reviewed and updated every six years.
River quality objective	A river quality objective is an agreed strategic target, expressed in terms of river ecosystem standards, which is used as the planning base for all activities affecting the water quality of a stretch of water. A river quality objective is the level of water quality that a river should achieve in order to be suitable for its agreed uses.
Rural Development Programme	The England Rural Development Programme and the Rural Development Plan for Wales are schemes in the Government's Public Incentive Programme. These programmes are of major significance for rural land management as they provide substantial funding to land managers conditional on the implementation of environmental (and other) actions.
Safeguard zone	A catchment or other defined zone around a point where the water is abstracted for potable use and where actions may be taken to protect raw water quality and prevent deterioration, so minimising the need for purification treatment. For groundwater they are likely to be based on source protection zones under the Environment Agency's Groundwater Protection Policy.
Saturation zone	Subsurface rock or other geological strata within which the pore spaces between the particles of rock or other strata, and the cracks in those strata, are filled with water and for which a water table may be determined.
Significant and sustained upward trend	A statistically significant trend in pollutant concentrations in groundwater that could lead to a future failure to meet one or more of the environmental objectives for groundwater unless it is reversed.
Site of Special Scientific Interest	An area of land notified under the Wildlife and Countryside Act 1981 by the appropriate nature conservation body (Scottish Natural Heritage in Scotland) as being of special interest by virtue of its flora and fauna, geological or physiogeographical features.
Source Protection Zone	A zone around a well, borehole or spring where groundwater is abstracted for human consumption (for example drinking water or food production), as defined under the Environment Agency's Groundwater Protection Policy (GP3). Zone 1 (SPZ1) is the area

	closest to the abstraction, representing the highest risk to the source. Zones 2 and 3 are progressively larger. Risk-based policies to prevent pollution are applied within these zones.
Spatial planning	Spatial planning is wider ranging than land-use planning based on regulation and control of land, and aims to ensure the best use of land by assessing competing demands. Social, economic and environmental factors are taken into account in making a decision that is more conducive to sustainable development.
Special Area of Conservation	Natura 2000 sites that are designated under the Habitats Directive.
Special Protection Area	Natura 2000 sites that are designated under the Birds Directive.
Specific pollutant	A substance considered as being discharged to the aquatic environment in significant quantities at the national level and for which environmental quality standards have been established. As part of the ecological classification criteria, and in places where these pollutants are monitored, these standards must be met for a surface water body to be classified as good ecological status.
Stakeholder	Individuals or groups that are or could become interested in, involved in or affected by our policies and activities. Our stakeholders include regulators, statutory bodies, professional organisations, local organisations and members of the public.
Strategic Environmental Assessment Directive (2001/42/EC)	European environmental legislation which requires an 'environmental assessment' to be carried out for certain plans and programmes whose formal preparation began after 21 July 2004 (or are prepared but not adopted or submitted by a legislative procedure by 21 July 2006), and which are considered likely to have significant effects on the environment. The term "Strategic Environmental Assessment" is used in UK guidance to mean an environmental assessment under this Directive.
Status	The physical, chemical, biological, or ecological quality of a water body.
Summary of Significant Water Management Issues	Report on each River Basin District that highlights significant water management issues in that River Basin District which will need to be addressed to achieve environmental objectives under the Water Framework Directive.
Supplementary Plans	Plans additional to the River Basin Management Plan which contain additional detail to that within the River Basin Management Plan but which fits wholly within its strategic principles and policies. Supplementary Plans do not cover issues outside the remit of the Water Framework Directive.
Sustainable Drainage Systems	A system of management practices and control structures designed to drain surface water in a more sustainable fashion than some conventional techniques.
Technical feasibility	Technical feasibility is determined through an assessment of whether the implementation of a measure or programme designed to meet Water Framework Directive objectives is technically possible at the national and local level and includes the consideration of uncertainty as well as environmental and socio economic feasibility.

	Technical feasibility depends upon the availability of a technical solution and information on the cause of the problem and hence the identification of the solution.
Toolkit of Measures	A variety of measures which consist of actions that when implemented can help meet Water Framework Directive objectives. These may include basic measures (the minimum set of measures that must be available) and supplementary ones.
Transitional water	A Water Framework Directive term for waters that are intermediate between fresh and marine water. Transitional waters include estuaries and saline lagoons.
Typology	The means by which the Water Framework Directive requires surface water bodies to be differentiated according to their physical and physico-chemical characteristics.
Water body	A manageable unit of surface water, being the whole (or part) of a stream, river or canal, lake or reservoir, transitional water (estuary) or stretch of coastal water. A 'body of groundwater' is a distinct volume of groundwater within an aquifer or aquifers.
Water Framework Directive	European Union legislation – Water Framework Directive (2000/60/EC) – establishing a framework for European Community action in the field of water policy.
Water Framework Directive management catchment	An amalgamation of a number of Water Framework Directive river water body catchments that provide a management unit at which level actions are applied.
Water Framework Directive objectives	The objectives set out in Article 4 of the Water Framework Directive together with objectives set out in paragraphs 2 and 3 of Article 7 of the Directive and which are required to be met.
Water Level Management Plans	Water Level Management Plans provide a means by which water level requirements for a range of activities including agriculture, flood defence and conservation can be balanced and integrated.
Water Protection Zones	Areas designated by the Secretary of State, within which activities polluting the water environment can be restricted or forbidden. Water Protection Zones can be designated at any scale (sub-catchment, catchment or a larger area) and restrictions are enforced to combat point and/or diffuse sources of water pollution, over and above other existing statutory powers.
Water Services	All services which provide, for households, public institutions or any economic activity: (a) abstraction, impoundment, storage, treatment and distribution of surface water or groundwater; (b) waste water collection and treatment facilities which subsequently discharge into surface water.
Water table	The upper limit of the saturation zone.
Water use	Water Services together with any other human activity identified as having a significant impact upon the status of water.
Weight of evidence	A weight of evidence approach integrates results or evidence from several data sources, weighted appropriately, to make risk-based decisions.

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