UPDATING THE ESTIMATE OF THE SOURCES OF PHOSPHORUS IN UK WATERS

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Executive summary

The aim of this desk study was to identify the current sources of phosphorus (P) loads to UK waters, and to compare this updated P source apportionment with previous source apportionments of P loads to inland and coastal waters of both the UK and other European countries.

First, methods of estimating and apportioning P loads to watercourses were reviewed to determine an appropriate methodology. This review suggested that P loads and source apportionment could be obtained by combining inventories of industrial point sources with estimates of diffuse P loads based on export coefficients for specific land uses, atmospheric deposition, livestock and human populations.

The diffuse total phosphorus (TP) loads to particular river basin districts (RBDs) were calculated using standard export coefficients that have been validated for many catchments in England and Wales and the areas of particular land cover classes (Land Cover Map 2000, Defra GIS Department), agricultural stocking densities (Agricultural Statistics Census for 2004) and human populations (Office of National Statistics Census 2001). Maximal point source industrial P loads were taken from the Environment Agency's Pollution Inventory. The soluble reactive phosphorus (SRP) loads from all these sources were estimated from the TP loads using specific SRP:TP ratios.

The TP load to the waters of England, Wales and Scotland was estimated to be about 41.6 kt y^{-1}. The agricultural contribution to this TP load was 11.8 kt y^{-1} (28.3%), the household contribution was 25.3 kt y^{-1} (60.7%), the industrial contribution was 1.9 kt y^{-1} (4.6%) and the contribution from background sources was 2.7 kt y^{-1} (6.5%). If the higher value of 0.61 kg TP per capita is used for human point source contributions (STWs), as calculated from the load of all STWs, then these proportions change, with agriculture accounting for 22.5%, households accounting for 68.7% (34.8 kt y^{-1}), industry accounting for 3.6% and background sources accounting for 5.2%. The SRP load to waters of England, Wales and Scotland was estimated to be about 31.3 kt y^{-1}. The agricultural contribution to the SRP load was 5.8 kt y^{-1} (18.6%), the household contribution was 21.1 kt y^{-1} (67.4%), the industrial contribution was 1.7 kt y^{-1} (5.5%) and the contribution from background sources was 2.7 kt y^{-1} (8.5%). The 'average' area normalised TP load to England, Wales and Scotland from agricultural sources is 0.55 kg TP ha^{-1} y^{-1} and that from all sources is about 1.94 kg TP ha^{-1} y^{-1}.

Phosphorus loads to watercourses differed widely between RBDs. The heaviest TP loads occurred in the heavily populated and/or extensive RBDs of Humber, Thames and Severn (Figure S1). The lightest TP loads occurred in the small, sparsely-populated RBDs of Northumbria, Solway Tweed and Dee. In terms of their area-normalised annual TP load, the RBD of Thames (4.41 kg ha^{-1} y^{-1}), North West (3.37 kg ha^{-1} y^{-1}) and Humber (3.33 kg ha^{-1} y^{-1}) were highest, whilst the RBDs of Anglian (1.39 kg ha^{-1} y^{-1}), Scotland (0.84 kg ha^{-1} y^{-1}) and Solway Tweed (0.57 kg ha^{-1} y^{-1}) were lowest. The agricultural contributions to the TP loads of RBDs followed the order Western Wales (62.4%), Solway Tweed (51.9%), Severn (48.2%), Dee (45.9%), South West (43.3%), Humber (34.9%), South East (21.8%), Anglian (20.35%), Scotland (18.6%), Northumbria (16.1%), North West (11.8%) and Thames (9.7%).

There is a notably high contribution from industrial point sources to the P load of Scottish waters (23.4%), and large contributions (>70%) from household sources to the P loads of the South East, North West and Thames RBDs (Figure S1).

It was observed that England contributed the most TP to UK waters. Agriculture contributed 26% of the English TP load, 22% of the Scottish TP load, 57% of the Welsh TP load and about 47% of the TP load to inland and coastal waters of Northern Ireland. These values are within the range estimated for the agricultural contributions to the TP loads to watercourses in many countries across Europe.

These observations are consistent with previous estimates of annual TP loads to UK estuaries and area-normalised TP loads from their catchments. They are also consistent with the observed geographical distribution of TP concentrations in lakes and rivers, and the recent assessment of areas at risk from diffuse P pollution undertaken by the Environment Agency.

The impact of P load and P source on the environment is discussed in the context of seasonal P loads from P sources with contrasting SRP:TP ratios and the sensitivities of different water bodies to eutrophication.

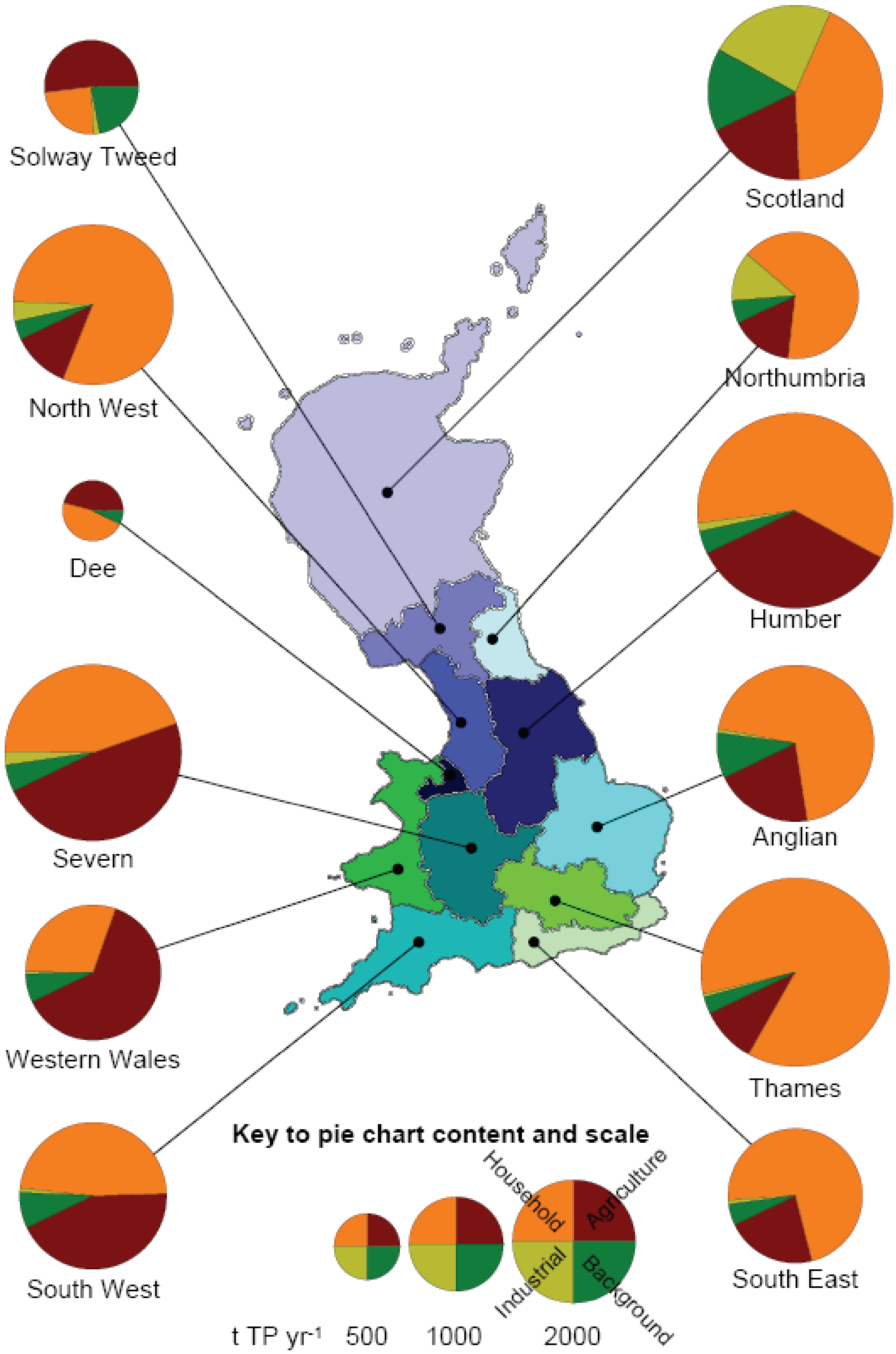


Figure S1. A summary of total TP loads (t TP y^{-1}) and percentage contributions from agricultural, household, industrial and background sources to the River Basin Districts of England, Wales and Scotland. Calculations used the data presented in Table 02.01, Table 02.02, Table 02.05 and Table 02.03 as described in the text.

Objective 01. Methods for estimating P loads and source apportionment

A literature survey was conducted to identify methods of estimating P loads and source apportionment to inland and coastal waters. The appropriateness of these methods for determining catchment-scale P loads and to apportion these loads to different P sources was assessed. The simplest, most versatile and most intuitive method to determine P loads was to sum the contribution from point sources (available from national inventories) and the output from empirical models using region-specific export coefficients for different land cover classes together with data for agricultural stocking densities and human populations. This method allowed the apportioning of P loads to a customised selection of contributing P sources required by Objective 02.

01.01 Methods for estimating P loads

Several methods are available for estimating P loads to watercourses. These separate into: (1) methods based on data from river flow and water quality monitoring programmes and (2) methods based on calculating P loads from inventories of point P sources plus estimates of diffuse P sources based on either empirical equations or mechanistic models.

01.01.01 Estimating P load using data from river flow and water quality monitoring programs.

Several methods are available for estimating the loads of orthophosphate and total phosphorus (TP) in rivers using data from river flow and water quality monitoring programs (Walling and Webb 1985; Watts and Littlewood 1998; Littlewood et al. 1998; Behrendt 1999; Cooper and Watts 2002, Kronvang et al. 2005a). The most common methods are those advocated by the International Oslo and Paris (OSPAR) Commission for estimating the mass loads transported by rivers to their tidal limits (OSPAR Commission 2004a). The advantages and limitations of these methods have been discussed by various authors (e.g. Littlewood et al. 1998; Littlewood and Marsh 2005). Essentially, river loads are calculated as sum of the products of instantaneous flow and concentration measurements sampled over an appropriate period. The relevant databases for river flow and river water quality in the UK are the National River Flow Archive (NRFA), which holds data for daily mean flows for sites throughout the UK, and the Harmonised Monitoring Scheme (HMS), which holds data on concentrations of orthophosphate and TP. Littlewood et al. (1998) and Ferrier et al. (2001) described UK data for orthophosphate as “broadly reliable”, but described data for TP as having “significant, substantial or severe problems”, because of missing data, low sampling frequencies and few sampling sites. This ‘instantaneous monitoring’ approach has been used to determine P loads for estuaries (Robson and Neal 1997; Littlewood et al. 1998; Watts and Littlewood 1998; Nedwell et al. 2002; Littlewood and Marsh 2005) and several smaller catchments (House et al. 1997; Bailey-Watts and Kirika 1999; Cooper et al. 2002; Hilton et al. 2002; Jarvie et al. 2002c; Wood et al. 2005) in the UK (see Section 05.03).

Four methods are commonly used to estimate annual loads from river flows and measured concentrations (Watts and Littlewood 1998; Littlewood et al. 1998). The first method estimates load as the product of the annual mean flow and the annual mean concentration. The second method estimates load as the sum of the products of flow and concentration data sampled regularly throughout the year. The third, and preferred, method is a ‘flow weighted’ version of the second method, in which the sum of the products of the regularly sampled flow and concentration data is multiplied by the mean of the sampled flow rates, divided by the sum of all flows (OSPAR 2004a). However, although these three methods might suffice for estimating orthophosphate loads from STW and/or industrial point sources that discharge at a constant rate throughout the year, they are often inappropriate for estimating the TP loads that are principally determined by low-frequency diffuse runoff events occurring during short periods of high water flow (Catt et al. 1994; Dils and Heathwaite 1996; Muscutt and Withers 1996; House et al. 1997; Jarvie et al. 2002c; Sharpley et al. 2002; Hanrahan et al. 2003; Haygarth et al. 2005b; Johnston and Dawson 2005; Salvia-Castellví et al. 2005) unless routine (monthly or fortnightly) sampling is replaced or complimented by high-frequency sampling during storm events. It is evident that calculating TP loads based on a monthly or fortnightly sampling regime can misestimate the real load considerably (e.g. Kronvang et al. 1996, Nedwell et al. 2002). To address this failing, a fourth method has been advocated. In this method, a parameterized empirical relationship between load and flow rate is used to estimate loads on those occasions when concentrations have not been measured (Hilton et al. 2002; Haygarth et al. 2004b; OSPAR 2004a). This method is advocated when much of the load is transported in predictable but infrequent events. Cooper and Watts (2002) give a detailed assessment of statistical aspects of all these methods. In addition to the equations used to estimate loads, and the regularity and frequency of sampling, the method used to estimate water flow and the chemical methods used to estimate orthophosphate and TP concentrations will also influence the accuracy and precision of the estimated P load (Littlewood et al. 1998; Ferrier et al. 2001; Littlewood and Marsh 2005).

It is often difficult to use monitoring data to estimate the absolute P loads to a large catchment unless the P retention properties of the catchment are known. Retention is a collective term for the diverse biological, geochemical and hydrological processes that remove, transform, or retard the transport of P within a watercourse (Behrendt 1999). It includes processes such as sedimentation or settling of particulate P and the uptake and storage of nutrients by plants, algae, microbes and fish. It is a seasonal phenomenon, with P being retained during the summertime and periods of low water flow and P being remobilised during periods of high water flows in the autumn and winter. Riverine retention of P in the summer may be considerable, and may account for over 50% of the diffuse P load to a watercourse (Behrendt 1999; Grizzetti et al. 2003; Demars et al. 2005), but the annual retention of P in rivers is often low (e.g. Svendsen and Kronvang 1993; Behrendt and Opitz 2000). An estimate of the annual P retention within a watercourse can be calculated from the empirical relationships derived by Behrendt and Opitz (2000), who suggest that P retention is a function solely of specific runoff to a river basin.

01.01.02 Estimating P loads from inventories of point sources and estimates of diffuse sources.

Point source household and industrial P loads can be obtained directly from monitoring data of industrial sources and wastewater treatment plants (STWs). However, some of these data are unavailable for the UK. Nevertheless, an upper value for industrial discharges directly to controlled waters can be obtained from the Environmental Agency's Pollution Inventory (EAPI, www.environment-agency.gov.uk). The greatest industrial discharges arise from chemical factories, abattoirs, creameries and food processing. When data for the actual point source P load from STWs is unavailable, one of two methods for estimating P loads can be adopted. In the first method, household P loads are estimated indirectly from population census data, assuming an average per capita P load after treatment by STWs (Morse et al. 1993; Mainstone et al. 1996; Smith et al. 2005). A calculation of the mean TP load divided by the 'population equivalent' (PE) served by the largest 62 STWs of Severn Trent Water listed on the EAPI suggests a value of 0.42 kg TP PE^{-1} y^{-1} for this (JP Hammond and PJ White, unpublished). It is assumed that this is derived from human excreta (66%) household waste (28%) and detergents (6%; Barr Engineering Company 2004; Smith et al. 2005). It is noteworthy that STWs operating either biological and/or chemical treatments (tertiary treatment) can remove more than 80% of the incoming P, which is the standard required of all STWs serving > 10,000 persons by the EU Urban Waste Water Treatment Directive 1991/271. Alternatively, the P load can be calculated from the product of an assumed constant P concentration STW effluent (10 mg l^{-1} in the UK) and the total permitted daily volume of effluent for the STW in each catchment (Wood et al. 2005). Whilst both these methods provide an estimate of the point source P load from the human population of a catchment, each has a number of potential sources of error. The use of export coefficients may not distinguish between people connected to mains sewerage systems and those served by septic tanks and no account is taken of differences in the degree of P removal (secondary, tertiary) between STWs. The use of permitted volumes leads to an over-estimation of flux from STWs, but omits the contribution from septic tanks. According to Wood et al. (2005) septic tank systems are the main method of disposal of human waste in most areas of rural Britain, although most properties in the UK are sewered (Morse et al. 1993; Marsden and Mackay 2001). The effluent from these tanks usually drains to a soakaway where the P is dissipated into the surrounding soil (Gold and Sims 2006). Carvalho et al. (2003) suggest that P loads to watercourses from septic tank systems may be higher than previously thought due to either lack of maintenance and/or failure during the wet seasons. Several studies indicate that the TP load from septic tanks approximates 0.24 to 0.40 kg TP $person^{-1}$ y^{-1} (Foy and Lennox 2000) and that from dwellings delivering effluent directly to rivers approximates 0.63-0.72 kg P $person^{-1}$ y^{-1} (Pieterse et al. 2003), although some authors suggest higher values (Smith et al. 2005).

The UK glasshouse industry covers a range of commodity sectors, spanning vegetable and ornamental crops. Burns (2004) reported that protected horticulture contributed to the P loads of 604 river catchments in England and Wales and amounted to 594 t TP y^{-1}, most of which came from vegetable and fruit crops (61%), cut flowers (11%) and nursery stock (21%). For crops grown hydroponically, such as tomatoes, cucumbers and peppers, TP losses can amount to 330 kg P ha^{-1} y^{-1} and TP losses from some long-season ornamental soil-grown crops, such as alstroemeria, pinks, carnations and chrysanthemums can be of similar magnitude (Burns 2004). Phosphorus losses from many container crops, for example pot and bedding plants, are generally somewhat lower. It is assumed that P losses from protected horticulture are dominated by orthophosphate and that they contribute to diffuse agricultural P loads.

Diffuse P loads arise from runoff from natural landscapes, agricultural fields and rural and urban surfaces, wastes from non-sewered populations and animals, atmospheric deposition and inputs from aquifers. They can be estimated either (A) by dividing the survey area into different land use categories using appropriate export coefficients, which can be done at varying levels of detail and sophistication with a minimal knowledge of the underlying processes, or (B) from modelling the P input and transport processes occurring in a survey area. These methods for estimating diffuse P loads are often calibrated against the observed P loads in watercourses. However, this calibration might be subject to error if any P bypasses a monitoring station or is temporarily retained in the upstream watercourse.

(A) Estimating P loads for diffuse sources from export coefficients

The approach of estimating P loads to watercourses from diffuse sources using export coefficients for different land use categories was originally developed in North America and subsequently adopted by the Organisation for Economic Cooperation and Development (OECD 1982). It has also been used by the former National Rivers Authority of England and Wales (e.g. Johnes et al. 1994), the Scottish Environmental Protection Agency (e.g. Ferrier et al. 1996) and the Department of the Environment for Northern Ireland (e.g. Irvine et al. 2001; Smith et al. 2005). Geographical Information Systems (GIS) are often used to perform regional assessments of land usage and export coefficients are matched to specific descriptions of land usage. In the UK the most commonly used classification schemes for land use are those of the Land Class Map (LCM2000, http://www.ceh.ac.uk/data/LCM; Fuller et al. 2002; Table 01.01), CORINE (Co-Ordination of Information on the Environment; Table 01.02) and Institute of Terrestrial Ecology (ITE; Table 01.03). This methodology offers a simple method to estimate the average annual diffuse P loads within large, diverse catchments, such as the RBDs, using data from readily available databases (Johnes 1996; Johnes and Hodgkinson 1998; Andersen et al. 2005; Bennion et al. 2005a; Heathwaite et al. 2005; Smith et al. 2005). It can be calibrated against observed water quality data, and provides a relatively inexpensive, robust means of evaluating the impact of land use and management practices on water quality for all standing and running surface freshwaters (Johnes 1996). Although the original export coefficient models were designed to estimate TP loads, an identical approach can be used to estimate PP and orthophosphate loads (McGuckin et al. 1999; Jordan et al. 2000; Andersen et al. 2005). Naturally, there are limitations to this approach. Firstly, estimates of P loads depend critically on the choice of export coefficients for particular land-cover types and climatic conditions. This is not a trivial issue, since the export coefficient for a particular land use may vary widely. Secondly, the method lacks spatial resolution, and relies on average export coefficients for particular land uses, which

Table 01.01 Description of the LCM2000 land cover classes for the UK.

16 LCM Target class	26 LCM Subclasses (Level-2)	Number	72 Variants (Level-3)	Number
Broad leaved / mixed woodland	Broad leaved / mixed woodland	1.1	deciduous	1.1.1
			mixed	1.1.2
			open birch	1.1.3
			scrub	1.1.4
Coniferous woodland	Coniferous woodland	2.1	conifers	2.1.1
			felled	2.1.2
			new plantation	2.1.3
Arable and horticultural	Arable cereals	4.1	barley	4.1.1
			maize	4.1.2
			oats	4.1.3
			wheat	4.1.4
			cereal (spring)	4.1.5
			cereal (winter)	4.1.6
	Arable horticulture	4.2	arable bare ground	4.2.1
			carrots	4.2.2
			field beans	4.2.3
			horticulture	4.2.4
			linseed	4.2.5
			potatoes	4.2.6
			peas	4.2.7
			oilseed rape	4.2.8
			sugar beet	4.2.9
			unknown	4.2.10
			mustard	4.2.11
			non-cereal (spring)	4.2.12
	Non-rotational arable and horticulture	4.3	orchard	4.3.1
			arable grass (ley)	4.3.2
			setaside (bare)	4.3.3
			setaside (undifferentiated)	4.3.4
Improved grassland	Improved grassland	5.1	intensive	5.1.1
			grass (hay/ silage cut)	5.1.2
			grazing marsh	5.1.3
Rough and semi-natural neutral and calcareous grasslands	Setaside grass	5.2	grass setaside	5.2.1
	Neutral grass	6.1	neutral grass (rough)	6.1.1
			neutral grass (grazed)	6.1.2
	Calcareous grass	7.1	calcareous (rough)	7.1.1
			calcareous (grazed)	7.1.2
Acid grass and bracken	Acid grass	8.1	acid	8.1.1
			acid (rough)	8.1.2
			acid with *Juncus*	8.1.3
			acid *Nardus/Festuca/Molinia*	8.1.4
	Bracken	9.1	bracken	9.1.1
Dwarf shrub heath (wet / dry)	Dense dwarf shrub heath	10.1	dense (ericaceous)	10.1.1
			gorse	10.1.2
	Open dwarf shrub heath	10.2	open	10.2.1
Fen, marsh and swamp	Fen, marsh and swamp	11.1	swamp	11.1.1
			fen/marsh	11.1.2
			fen willow	11.1.3
Bog	Bogs (deep peat)	12.1	bog (shrub)	12.1.1
			bog (grass/shrub)	12.1.2
			bog (grass/herb)	12.1.3
			bog (undifferentiated)	12.1.4
Water (inland)	Water (inland)	13.1	water (inland)	13.1.1
Montane habitats	Montane habitats	15.1	montane	15.1.1
Inland Bare Ground	Inland Bare Ground	16.1	semi-natural	16.1.1
			despoiled	16.1.2
Built up areas, gardens	Suburban/rural developed	17.1	suburban/rural developed	17.1.1
	Continuous Urban	17.2	urban residential/commercial	17.2.1
			urban industrial	17.2.2
Supra-littoral rock and sediment	Supra-littoral rock	18.1	rock	18.1.1
	Supra-littoral sediment	19.1	shingle (vegetated)	19.1.1
			shingle	19.1.2
			dune	19.1.3
			dune shrubs	19.1.4
Littoral rock and sediment	Littoral rock	20.1	rock	20.1.1
			rock with algae	20.1.2
	Littoral sediment	21.1	mud	21.1.1
			sand	21.1.2
			sand with algae	21.1.3
	Saltmarsh	21.2	saltmarsh	21.2.1
			saltmarsh (grazed)	21.2.2
Sea / Estuary	Sea / Estuary	22.1	sea	22.1.1

Table 01.02. Export coefficients for CORINE land cover classes in Northern Ireland. Data from McGucken et al. (1999, SRP, PP and SOP), Jordan et al. (2000, SRP) and Smith et al. (2005, TP).

Corine land cover class and name		Export coefficient TP (kg P ha^{-1} y^{-1})	Export coefficient SRP (kg P ha^{-1} y^{-1})
1.1.1.	Continuous urban fabric	1.20 ± 0.90	0.18
1.1.2.	Discontinuous urban fabric	1.20 ± 0.90	0.18
1.2.1.	Industrial or commercial units	2.50 ± 1.60	0.30
1.2.2.	Road and rail networks and associated land	1.20 ± 0.90	0.18
1.2.3.	Sea ports	2.50 ± 1.60	0.30
1.2.4.	Airports	2.50 ± 1.60	0.30
1.3.1.	Mineral extraction site	2.50 ± 1.60	0.30
1.3.2.	Dump	2.50 ± 1.60	0.30
1.3.3.	Construction site		
1.4.1.	Green urban areas	0.83 ± 0.17	0.30
1.4.2.	Sport and leisure facilities	1.20 ± 0.90	0.18
2.1.1.	Non-irrigated arable land	4.88 ± 1.12	2.29
2.1.2.	Permanently irrigated land		
2.1.3.	Rice fields		
2.2.1.	Vineyards		
2.2.2.	Fruit trees and berries		
2.2.3.	Olive groves		
2.3.1.1	Good pasture	0.83 ± 0.17	0.30
2.3.1.2	Poor pasture	0.65 ± 0.25	0.36
2.3.1.3	Mixed pasture	0.78 ± 0.12	0.32
2.4.1.	Annual crops assoc with permanent crops	0.83 ± 0.17	0.30
2.4.2.	Complex cultivation patterns	2.33 ± 0.27	1.05
2.4.3.	Land principally occupied by agriculture with significant areas of natural vegetation.	0.49 ± 0.11	0.25
2.4.4.	Agroforestry area		
3.1.1.	Broadleaved forest	0.26 ± 0.14	0.13
3.1.2.	Coniferous forest	0.36 ± 0.04	0.08
3.1.3.	Mixed forest	0.26 ± 0.15	0.13
3.2.1.	Natural grassland	0.65 ± 0.25	0.36
3.2.2.	Moor and heathland	0.13 ± 0.07	-
3.2.3.	Sclerophyllous vegetation		
3.2.4.	Transitional woodland/scrub	0.26 ± 0.14	0.23
3.3.1.	Beaches, dunes, sand		
3.3.2.	Bare rocks		
3.3.3.	Sparsely vegetated areas		
3.3.4.	Burnt areas		
3.3.5.	Glaciers and permanent snowfields		
4.1.1.	Inland marshes	0.23 ± 0.17	0.23
4.1.2.1.	Unexploited peat bogs	0.23 ± 0.17	0.23
4.1.2.2.	Exploited peat bogs	0.23 ± 0.17	0.23
4.2.1.	Salt marshes		
4.2.2.	Salines		
4.2.3.	Intertidal flats		
5.1.1.	Stream courses		
5.1.2.	Water bodies	-6.57	-13.39
5.2.1.	Coastal lagoons		
5.2.2.	Estuaries		
5.2.3.	Sea and ocean		

Table 01.03. Export coefficients for ITE land cover types (kg ha^{-1} y^{-1}) and livestock (kg $head^{-1}$ y^{-1}) for the UK (Hilton et al. 2002).

Class Land cover types	Export coefficient	Notes
0 Unclassified	0.48	Weighted mean of coefficients below in proportion to the % of each area on the land cover map.
1 Sea / estuary	0.00	Not a contributing area
2 Inland water	0.00	Not a contributing area
3 Beach and coastal bare	0.00	Not a contributing area
4 Salt marsh	0.00	Not a contributing area
5 Grass heath	0.02	Rough grazing Johnes (1996)
6 Mown / grazed turf	0.20	Average of 0.1 for permanent and 0.2 for temporary grass (Johnes 1996)
7 Meadow / verge / semi-natural	0.20	As type 6
8 Rough / marsh grass	0.02	Rough grazing Johnes (1996)
9 Moor land grass	0.02	Rough grazing Johnes (1996)
10 Open shrub moor	0.02	Rough grazing Johnes (1996)
11 Dense shrub moor	0.02	Woodland/rough grazing
12 Bracken	0.02	Rough grazing Johnes (1996)
13 Dense shrub heath	0.02	Woodland/rough grazing
14 Scrub / orchard	0.02	Woodland/rough grazing
15 Deciduous woodland	0.02	Woodland – Johnes (1996)
16 Coniferous woodland	0.02	Woodland – Johnes (1996)
17 Upland bog	0.00	Probable P trap
18 Tilled land	0.66	Cereals (0.65), Rape (0.65), Field vegetables (0.65), root crops (0.8)
19 Ruderal weed	0.02	This land type includes areas of bare ground being colonised by annual and short-lived perennials – value for rough grazing selected.
20 Suburban / rural development	0.83	May et al. (1996)
21 Continuous urban	0.83	May et al. (1996)
22 Inland bare ground	0.70	Weighted mean of urban and tilled land in proportion to the % area in the land cover map.
23 Felled forest	0.02	See 15
24 Lowland bog	0.00	Probable P trap
25 Open shrub heath	0.02	Woodland/rough grazing

might differ from the export coefficients for a particular field or individual location, and therefore must be calibrated appropriately. Thirdly, since the method is designed to produce estimates of annual P loads it is unable to predict P losses over the shorter timescales, although predictions at higher temporal resolution can be achieved by multiplying a flow-normalized annual export coefficient and instantaneous water flows (Hanrahan et al. 2001; May et al. 2001). Fourthly, since export coefficients seek to integrate all the physical processes operating on P fluxes within the catchment, they do not afford much insight to the complex biological or physical dynamics of P fluxes in the environment. Despite these reservations, this approach has been validated in numerous studies in both the UK and elsewhere in Europe (e.g. Johnes et al. 1994, 1996; Johnes 1996; Johnes and Heathwaite 1997; McGuckin et al. 1999; Brigault and Ruban 2000; Jordan et al. 2000; Hanrahan et al. 2001; Irvine et al. 2001, 2003; Haygarth et al. 2003; Bowes et al. 2005; Heathwaite et al. 2005; Smith et al. 2005) and is thought to provide a valuable tool for water district managers in complying with the Water Framework Directive (Andersen et al. 2005).

Traditionally, export coefficients for particular land uses have been derived from either (i) data from experimental plots, where the land use type is narrowly defined, and the measured export rates are acknowledged to be partly dependent on the physical and management variables specific to a particular study site, or (ii) data from monitoring programs of watersheds either directly, by monitoring watersheds with a unique land usage, or, indirectly using data from many contrasting watersheds with diverse types of land use through statistical techniques to apportion loads to each land use. Export coefficients from the latter are assumed to apply over large geographic areas of similar land use. Latterly, theoretical export coefficients for particular types of land use have also been derived from mechanistic models of catchment-scale P fluxes.

Values for export coefficients for different land use categories differ widely. In this report we have categorised land use using the LCM2000 classification scheme (Table 02.01), but export coefficients are available for other land use classification schemes, such as CORINE (Table 01.02) and ITE (Table 01.03). In general, it is observed that forest and woodland have the lowest export coefficients, followed by natural vegetation and grassland. Improved grassland often has a higher export coefficient than unimproved grassland and fallow land. The export coefficient for arable land producing cereals is often greater than that for grassland, but is generally less than the export coefficients for wide row crops such as corn, field vegetables, root crops and oilseed rape. Urban landscapes generally have the highest export coefficients. However, it is evident that many factors influence the value of an export coefficient for a particular land use category. These factors include, for example, vegetation or crop specific management practices, the topology of the landscape, hydrological connectivity and the timing, amount and intensity of rainfall. This is reflected in the contrasting export coefficients obtained for the same land use category (reviewed by Ryden et al. 1973; Dillon and Kirchner 1975; Beaulac and Reckhow 1982; Frinck 1991; Johnes 1996; Johnes et al. 1996; Mattikalli and Richards 1996; Young et al. 1996; Hanrahan et al. 2001; McDowell et al. 2001; McFarland and Hauck 2001; Ierodiaconou et al. 2005; Smith et al. 2005). Thus, export coefficients must be tailored to a particular land use and topography (Johnes et al. 1994, 1996; Johnes 1996; Johnes and Heathwaite 1997; Johnes and Hodgkinson 1998; McFarland and Hauck 2001; Andersen et al. 2005; Bowes et al. 2005). Hence, some 'mixed' models allow export coefficients to differ depending upon fertiliser inputs, livestock densities, soil type, geology, topography, hydrology, response characteristics of the catchment, and management practices (e.g. Johnes et al. 1994; Daly et al. 2002; Heathwaite et al. 2005).

The diffuse urban P load originates from a variety of sources including atmospheric emissions, runoff from roads, driveways, roofs, parking lots, construction sites and gardens, which exhibit seasonality according to rainfall and leaf fall, urban litter, car-washing and industrial spills. It is more difficult to predict than the diffuse P loads from agricultural land, because urban land use is more diverse. This is reflected by differences in the export coefficients ascribed to the CORINE land use categories of urban and industrial or commercial landscapes (Table 01.02). Soluble P can contribute significantly to the diffuse urban P load (Ryden et al. 1973; McGuckin et al. 1999; Jordan et al. 2000).

The diffuse P load from an animal is estimated as the product of the P in wastes voided by the animal, which is assumed to follow a logarithmic relationship with body weight (Johnes et al. 1996; Domburg et al. 1998; Heathwaite et al. 2003a), the amount of these applied to land, which is assumed to be 95% for cattle, 85% for pigs, 100% for sheep and 90% for poultry (Johnes 1996; Smith et al. 1998; Withers and Lord 2002), and an export coefficient to describe the movement of animal wastes to the watercourse (0.03; Johnes 1996). These (uncalibrated) values are often modified for a particular catchment to reflect its topology, geology and hydrology (Table 02.01).

The atmospheric P load to the land is estimated as the product of P deposited to an area and an export coefficient to describe its movement to a watercourse. The atmospheric P load is composed of dry and wet deposition and depends upon the P concentration and amount of rainfall. Values in the literature for P deposition in the UK commonly range between 0.05 and 0.40 kg P ha^{-1} y^{-1}, although some may exceed the latter value (Gibson et al. 1995). Higher values are generally associated with urbanisation, industrialisation and agriculture. A blanket value of 0.4 kg P ha^{-1} y^{-1} and a transport coefficient of 0.3 are assumed to be appropriate for export coefficient models of UK catchments (Gibson et al. 1995, Johnes et al. 1994, 1996; Johnes 1996; Mainstone et al. 1996; Parr et al. 1999; Winter and Duthie 2000; Wood et al. 2005).

(B) Estimating P loads for diffuse sources from mechanistic models

Mechanistic models seek to describe P dynamics in the environment using physical parameters (Grimvall and Stålnacke 1996; Maidstone et al. 1996; Johnes and Hodgkinson 1998; Sharpley et al. 2002; Heathwaite 2003; Heathwaite et al. 2005). Thus, they impose a theory on the data that may, or may not, be correct. They can be divided roughly into “field scale” and “catchment scale” models, but all contain many parameters representing discrete physical processes. These models often predict P dynamics well at the sites for which they were developed, but care must be taken to calibrate them for other sites. This is probably a consequence of the complexity of the physical, chemical and biological processes involved in P dynamics, which are over-simplified in the models’ calculations (Harris and Heathwaite 2005).

Several field-scale models are available. Examples include: ANIMO (Agricultural NutrIent MOdel), EPIC (Erosion-Productivity Impact Calculator; http://www.brc.tamus.edu/epic/), CENTURY (http://www.nrel.colostate.edu/projects/century5/), MACRO, EWQM (Everglades Water Quality Model; Raghunathan et al. 2001) and PSYCHIC (http://www.psychic-project.org.uk/index.php?content=welcome). The ANIMO, EPIC, CENTURY and MACRO models all have their own unique representation of soil P dynamics (reviewed by Pierzynski et al. 2000; Lewis and McGechan 2002; McGechan et al. 2005). The CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems) and ICECREAM models all have copies of the same set of basic equations for P processes as the EPIC model, and the DAYCENT model is the daily time-step output version of the CENTURY ecosystem model. Lewis and McGechan (2002) observed that a comprehensive description of all processes relevant to P in soil would consider transport of both soluble and particulate P, and of both inorganic and organic P, by three routes: overland (surface runoff), through the soil to field drains, and vertically through the soil down to deep groundwater, as well as transformations from one form of P to another following applications of both mineral fertilizer and manure P. None of the models available in 2002 considered all these processes, although most considered a sub-set appropriate to a particular situation. However, since the review of Lewis and McGechan (2002) several of the field-scale models have been improved. For example, ICECREAM has recently incorporated a description of macropore fluxes (Andersson et al. 2003).

Examples of catchment-scale models include the GWLF (Generalized Watershed Loading Function) model, which was developed as an "engineering compromise between the empiricism of export coefficients and the complexity of chemical simulation models" (Schneiderman et al. 2002), the AgNPS (Agricultural Non-point Pollution Source) and ANSWERS-2000 (Areal Nonpoint Source Watershed Environment Response Simulation) models, which describe flows and P loadings following rainfall events, TOPCAT-P, which is a simplification of TOPMODEL and uses components of EPIC, HSPF (Hydrological Simulation Program - Fortran), GAMESP (Guelph model for evaluating the effects of Agricultural Management systems on Erosion, Sedimentation and Phosphorus yields), SWAT (Soil and Water Assessment Tool, *http://www.brc.tamus.edu/swat/index.html*), MONOTERIS (MOdeling Nutrient Emissions in River Systems), INCA-P (INtegrated Catchments model of Phosphorus dynamics; Wade et al. 2002), PIT (Phosphorus Indicators Tool; Heathwaite 2003; Liu et al. 2005) and many others (Mainstone et al. 1996; McGechan et al. 2005).

In the UK, the PIT model has been used to determine diffuse P loads for a number of small catchments (Heathwaite et al. 2003a; Brazier et al. 2005; Heathwaite et al. 2005; Liu et al. 2005), and has been recommended as a screening tool to identify specific areas contributing most to P loads to watercourses within a catchment.

01.02 Methodologies for estimating source apportionment

Several methods are available to apportion the P load to watercourses between different sources. These methods are based on either (1) water quality monitoring data or (2) inventories of point P sources plus estimates of diffuse P sources based on either export coefficients or mechanistic models.

01.02.01 Estimating source apportionment from water quality data

Water quality data can be used in a variety of ways to estimate the contribution of different sources to the P load of a watercourse. The simplest method for estimating the contribution of diffuse sources to a P load is from the difference in the P load estimated from water flow and P concentration data (Section 01.01.01) and the point source inventory (Behrendt 1999; Kronvang et al. 2005a). It has been used to determine P apportionment for catchments in several European countries including Finland and Denmark. However, this simple method neglects the occurrence of retention or loss in the river system and may, therefore, underestimate P losses from diffuse sources. To address this criticism, a coefficient can be included in the calculation to account for systematic losses (Behrendt 1999; Behrendt and Opitz 2000; EEA 2005). This method, including a value for retention, is recommended by the OSPAR Commission to estimate source apportionment in the P loads of rivers (OSPAR Commission 2004b).

Alternatively, source apportionment between point and diffuse sources, and/or between contrasting land uses, may be based on statistical analysis of observed P loads and explanatory variables (i.e., factors explaining variability in loads between sites or in time). This methodology can be divided into two categories: (1) regression analysis between observed P loads and water flow, which assumes that point P sources are constant whilst diffuse P sources fluctuate with water flow, and (2), regression analysis between the observed P load and watershed characteristics (e.g. McGuckin et al. 1999; McFarland and Hauck 2001). Indeed, regression methods were used to determine export coefficients for soluble reactive P (SRP), soluble organic P (SOP) and particulate P (PP) for CORINE land cover classes using data on the observed P loads from catchments containing different proportions of each land cover class (McGuckin et al. 1999).

A complementary, but indirect, apportionment approach based on water quality data uses specific physical, chemical and/or biological properties as tracers for particular P sources (Walling 2005). First, a physical, chemical or biological property is selected that differentiates the P source materials and this property is then assayed in water samples. Useful tracers are present in different P sources with unique relative concentrations, transported in watercourses with the P source material and are resistant to degradation. There are three classes of potential tracers: 1) isotopes of mineral elements, 2) synthetic organic chemicals and 3) biological species. Chloride and boron are often considered as conservative tracers for sewage effluent. Boron is a product of household and industrial detergents and is delivered to watercourses predominantly through STWs. Thus, it is possible to quantify the contribution of STWs to P loads from knowledge of the effluent B:P ratio and the B load of the river (Jarvie et al. 2002ac, 2005; Neal et al. 2005). Silicon exhibits a positive correlation with agricultural P sources, and agrochemicals associated with specific crops and land use practices could provide a tracer for more detailed analyses. Loads from the topsoil can be estimated from the quantity of anthropogenic radioisotopes (^{210}Pb, ^{137}Cs, ^{7}Be) present in the water, and are well suited to use in heterogeneous catchments since their concentrations are independent of soil type and underlying geology (Foster et al. 2003; Gruszowski et al. 2003; Walling 2005). Assaying a number of properties that are distinct for specific P sources allows for a more robust apportionment of P loads between these sources (Walling 2005). Chemical properties have been used successfully to apportion P loads between different sources by various researchers (e.g. Jarvie et al. 2002ac, 2005; Walling et al. 2005).

01.02.02 Estimating source apportionment from inventories of point sources and estimates of diffuse P loads

Source apportionment of P is often accomplished by combining inventories of point sources and estimates of diffuse loads estimated from export coefficients or mechanistic modelling (Mainstone et al. 1996; Behrendt 1999; EEA 2005; Kronvang et al. 2005a). In essence a P budget is drawn up, and the relative contributions of different P sources to the total P load are calculated. In principle, this is a simple, flexible and intuitive way to estimate the contribution of different P sources to the total P load and, if appropriate models are used, is capable of apportioning relative P loads to a customised selection of contributing sources. Nevertheless, the estimates of P apportionment are obviously subject to the choice of the export coefficients used in empirical models, the accuracy of mechanistic models, the ability to estimate point P sources and the information provided in population and agricultural censuses. But, uncertainty in values for export coefficients can be used to estimate uncertainty in source apportionment (e.g. McFarland and Hauck 2001). Nevertheless, it is especially important to calibrate the empirical and mechanistic models used to estimate diffuse P loads against observed water quality data before their application (Heathwaite 2003; Andersen et al. 2005).

Objective 02. Quantifying P loads and source apportionment for UK waters

There is great variability in export coefficients derived for similar land usage (see Section 01.01.02). To estimate diffuse TP loads for RBDs, we have used export coefficients adapted for the 72 Level-3 subclasses of land cover in the Land Cover Map 2000 (LCM2000); condensed to 27 LCM2000 Level 2 land cover classes (Table 02.01). The export coefficients are based on those determined by Johnes and colleagues (Johnes et al. 1994, 1996; Johnes 1996) and May et al. (1996). They comprise six groups of export coefficients validated for specific types of terrain defined in the Land Use of Britain Land Utilisation Survey (Dudley-Stamp 1941). These export coefficients have been used in many studies estimating TP loads to catchments in England and Wales (e.g. Johnes et al. 1994, 1996; Johnes 1996; Mattikalli and Richards 1996; May et al. 1996, 2001; Johnes and Heathwaite 1997; Johnes and Hodgkinson 1998; Bailey-Watts and Kirika 1999; Hanrahan et al. 2001; Hilton et al. 2002; Kampas et al. 2002; Bennion et al. 2005ab; Bowes et al. 2005; Heathwaite et al. 2005). The TP load is calculated from agricultural stocking densities, human populations and atmospheric deposition in addition to land use statistics (Table 02.01). It is assumed that 30% of the rainfall on land reaches a watercourse and that the TP load in rainfall is 0.4 kg TP ha^{-1} y^{-1} (Gibson et al. 1995, Johnes et al. 1994, 1996; Mainstone et al. 1996). The TP loads from human sewerage were calculated from the human populations in RBD multiplied by an estimated TP load per person per year. This estimate was set to the mean TP load per 'population equivalent' (PE) served for the largest 62 STWs of Severn Trent Water listed on the EAPI. This calculation assumes that the average TP load per person per year to watercourses is the same for all STWs and that the average TP loads to watercourses from STWs and septic tanks are identical. The TP load per capita from human sewerage calculated by ADAS from an amended inventory of P loads from STWs was 0.61 kg TP per capita per year. The SRP loads were determined from the TP load from LCM2000 land use classes, livestock, human and rainfall using estimates of their SRP/TP quotients (Table 02.01).

The areas of land use classes, including the area of inland waters receiving atmospheric deposition, in each RBD were derived from the LCM2000 data set, supplied under license by Defra GIS Department (Table 02.02). Livestock numbers were obtained from agricultural census data for 2004 (England, http://statistics.defra.gov.uk/esg; Wales, http://www.wales.gov.uk/keypubstatisticsforwales/content/publication/agriculture/2005/was2004/was2004-e.htm; Scotland, http://www.scotland.gov.uk/Publications/2006/03/13143916/0) and human populations were taken from the 2001 Census data obtained from the Office of National Statistics (http://www.statistics.gov.uk). Glasshouse area and poultry numbers for England were obtained from Agricultural Census Data for 2000 supplied by Defra. The TP load from glasshouses and average national export coefficients from glasshouses were calculated from the data presented by Burns (2004). The number of industrial point sources releasing P into controlled waters was abstracted from the EAPI.

Table 02.01. Export coefficients for land use classes (kg TP ha^{-1} y^{-1}), livestock and humans (kg TP y^{-1}) used to determine TP loads from River Basin Districts. The export coefficients of Johnes and colleagues (Johnes et al. 1994, 1996; Johnes 1996; Bowes et al. 2005) are calibrated for (1) upland and moorland (2) intensive mixed farming, (3) South Devon, (4) limestone and chalk, (5) Eastern England and (6) North West England lowland. The coefficients for the RBD of Northumbria, Solway Tweed and Scotland were for "upland and moorland" those for the Dee, Severn and Western Wales RBD were for "intensive mixed farming", those for the South West RBD were "South Devon", those for the South East and Thames RBD were for "limestone and chalk", those for the Anglian RBD were 'Eastern England' and the RBDs of the Humber and the North West had the average coefficients for "upland and moorland and intensive mixed farming" and "upland and moorland and North West England lowland", respectively. The urban export coefficients in italics were taken from May et al. (1996). Values for human sewerage were calculated as described in the text (Section 02.01). The CORINE export coefficients derived by McGuckin et al. (1999) are given for comparison. The CORINE export coefficients include a contribution for livestock. The final column contains estimates of SRP/TP quotients.

LCM subclass		**(1)**	**(2)**	**(3)**	**(4)**	**(5)**	**(6)**	**CORINE**	**SRP/TP**
1.1	Broad-leaved and mixed woodland	0.02	0.02	0.02	0.02	0.07	0.02	0.26	0.50
2.1	Coniferous woodland	0.02	0.02	0.02	0.02	0.07	0.02	0.36	0.22
4.1	Arable cereals	0.60	0.90	0.60	0.65	0.22	0.60	4.88	0.35
4.2a	Horticulture: Root crops	0.70	0.90	0.70	0.80	0.27	0.70	4.88	0.45
4.2b	Horticulture: Field vegetables	0.60	0.90	0.60	0.65	0.22	0.60	4.88	0.45
4.3.1	Orchard	0.02	0.02	0.02	0.02	0.07	0.02	0.26	0.50
4.3b	Setaside grass (ley)	0.02	0.02	0.02	0.02	0.07	0.02	0.65	0.40
5.1	Improved grassland	0.30	0.80	0.40	0.10	0.03	0.10	0.83	0.40
5.2	Grass setaside	0.02	0.02	0.02	0.02	0.07	0.02	0.65	0.40
6.1	Neutral grass	0.02	0.02	0.02	0.02	0.07	0.02	0.65	0.50
7.1	Calcareous grass	0.02	0.02	0.02	0.02	0.07	0.02	0.65	0.50
8.1	Acid grass	0.02	0.02	0.02	0.02	0.07	0.02	0.65	0.50
9.1	Bracken	0.02	0.02	0.02	0.02	0.07	0.02	0.13	0.80
10.1	Dense dwarf shrub heath	0.02	0.02	0.02	0.02	0.07	0.02	0.13	0.80
10.2	Open dwarf shrub heath	0.02	0.02	0.02	0.02	0.07	0.02	0.13	0.80
11.1	Fen, marsh and swamp	0.00	0.00	0.00	0.00	0.00	0.00	0.23	1.00
12.1	Bogs (deep peat)	0.00	0.00	0.00	0.00	0.00	0.00	0.23	1.00
13.1	Water (inland)	0.00	0.00	0.00	0.00	0.00	0.00	X	X
15.1	Montane								
16.1	Inland bare ground	*0.70*	*0.70*	*0.70*	*0.70*	*0.70*	*0.70*	1.20	0.30
17.1	Suburban/rural development	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	1.20	0.50
17.2.1	Continuous urban	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	1.20	0.50
17.2.2	Urban industrial	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	*0.83*	2.50	0.50
18.1	Supra-littoral rock	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
19.1	Supra-littoral sediment	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
20.1	Littoral rock	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
21.1	Littoral sediment	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
21.2	Saltmarsh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
22.1	Sea / estuary	0.00	0.00	0.00	0.00	0.00	0.00	0.00	X
	Glasshouses	331	331	331	331	331	331	X	1.00
	Cattle	0.096	0.422	0.215	0.215	0.007	0.192	X	0.56
	Pigs	0.072	0.281	0.143	0.143	0.047	0.143	X	0.55
	Sheep	0.023	0.090	0.045	0.045	0.015	0.045	X	0.50
	Horses	0.096	0.422	0.215	0.215	0.007	0.192	X	0.50
	Poultry	0.003	0.011	0.005	0.005	0.002	0.005	X	0.35
	Humans (urban)	0.423	0.423	0.423	0.423	0.423	0.423	0.766	0.85
	Humans (rural)	0.423	0.423	0.423	0.423	0.423	0.423	0.440	0.85

The apportionment was divided into the following sectors: (1) Agriculture, comprising glasshouses, improved grassland, moors and heaths, arable cereals, field horticulture and livestock (cattle, pigs, sheep, poultry); (2) Human and household, comprising point (STWs) and diffuse urban sources; (3) Industrial, comprising point and diffuse sources, and (4) Background, comprising orchards, woodlands, forests, wetlands (bog, fen, marsh, swamp) and atmospheric deposition. Total TP loads for land use classes were calculated by multiplying their area and export coefficient values (Table 02.03). Diffuse TP loads from livestock were calculated by multiplying their number and TP load per capita. Human point source (STW) loads were calculated by multiplying population statistics and TP load per population equivalent. Phosphorus loads from industrial point sources releasing P into controlled waters were obtained from the EAPI and atmospheric deposition was calculated as the sum of the atmospheric P deposition to surface waters and 30% of the atmospheric P deposition to the land. The SRP loads (Table 02.04) were estimated from the TP loads (Table 02.03) using the SRP/TP quotients given in Table 02.01. The TP load to England, Wales and Scotland was estimated to be about 41.6 kt y^{-1} (Table 02.03). The amount of TP apportioned to agriculture is 11.8 kt y^{-1} (28.3%), to households is 25.3 kt y^{-1} (60.7%), to industry is 1.9 kt y^{-1} (4.6%) and to background sources is 2.7 kt y^{-1} (6.5%). If the higher value of 0.61 kg TP per capita is used for human point source contributions (STWs), as calculated from the load of all STWs, then these proportions change, with agriculture accounting for 22.5%, households accounting for 68.7% (34.8 kt y^{-1}), industry accounting for 3.6% and background sources accounting for 5.2%. The SRP load to England, Wales and Scotland was estimated to be about 31.3 kt y^{-1} (Table 02.04). The amount of SRP apportioned to agriculture is 5.8 kt y^{-1} (18.6%), to households is 21.1 kt y^{-1} (67.4%), to industry is 1.7 kt y^{-1} (5.5%) and to background sources is 2.7 kt y^{-1} (8.5%). Improved grassland contributed 18.0%, other grassland, moors and heath contributed 1.2%, arable agriculture contributed 9.6%, field horticulture contributed 16.8%, glasshouses contributed 6.1% and livestock contributed 48.3% to the TP load from agriculture (Table 02.03). Cattle contributed 37.5%, sheep 13.2%, pigs 33.0%, poultry 15.0% and horses 1.3% to the TP load from livestock. Diffuse urban TP contributed 4.4% and STWs contributed 95.6% to the TP load from households. Chemical industries contributed 75% and combustion processes contributed 10% to the industrial TP load. The contribution of agricultural, household, industrial and background sources to the TP loads of specific RBD are itemized in Table 02.05. The ‘average’ area normalised TP load to England, Wales and Scotland from all diffuse sources is about 0.83 kg TP ha^{-1} y^{-1}, from agricultural sources is about 0.55 kg TP ha^{-1} y^{-1} and that from all sources is about 1.94 kg TP ha^{-1} y^{-1}. These values are consistent with many previous studies reporting P loads calculated from monitoring data on water flow and P concentrations in UK rivers (see Section 05.03; Muscutt and Withers 1996; Robson and Neal 1997; Haygarth and Jarvis 1999; Johnston and Dawson 2005).

Table 02.02 Areas of land use classes (ha), livestock numbers and human populations for the River Basin Districts of England and Wales. The LCM2000 land cover classes were used to calculate areas for the following sectors: improved grassland (5.1), grassland, moor and heath (4.3.2, 4.3.3, 4.3.4, 5.2, 6.1, 7.1, 8.1, 9.1, 10.1, 10.2, 15.1), arable cereals (4.1), field horticulture (4.2), orchard, wood and forest (1.1, 2.1, 4.3.1) bog, fen, marsh and swamp (11.1, 12.1), diffuse urban (17.1, 17.2.1) diffuse industrial (16.1, 17.2.2) and inland waters (13.1, 21.2, 22.1).

Sector	Anglian	Dee	Humber	North West	Northumbria	Severn	Solway Tweed	South East	South West	Thames	Western Wales	Scotland	Total
Agriculture													
Glasshouses	363	4	335	254	21	198	10	358	162	315	43	94	2,156
Improved grassland	184,874	63,434	467,528	315,351	171,128	645,024	423,037	180,591	653,624	274,398	479,537	688,628	4,547,154
Grassland, moor, heath	274,751	80,007	431,001	311,303	225,725	452,016	516,326	97,772	210,531	186,374	333,423	3,236,917	6,356,146
Arable cereals	746,067	9,535	188,295	55,718	103,554	203,556	78,273	124,919	255,492	236,157	3,818	115,517	2,120,899
Field horticulture	823,004	11,964	723,100	76,216	96,793	283,701	97,348	140,746	227,519	279,094	52,144	382,743	3,194,372
Cattle	369,515	135,596	1,117,030	695,370	303,994	1,211,408	874,911	219,729	1,417,745	323,479	839,812	1,396,858	8,905,447
Pigs	1,379,448	17,173	1,612,687	141,924	89,874	282,818	62,370	106,715	347,083	213,809	4,529	430,514	4,688,944
Sheep	891,094	320,081	2,842,974	2,301,456	1,949,846	6,773,039	3,510,918	688,127	2,770,846	805,427	5,241,899	5,549,336	33,645,043
Poultry	40,657,121	266,575	22,951,750	9,107,788	2,409,368	24,620,444	4,933,234	5,995,114	14,043,396	8,649,962	515,296	11,563,961	146,714,009
Horses	27,875	3,513	37,081	19,985	9,176	42,416	7,319	20,266	36,100	34,760	24,779	24,765	288,035
Household													
Humans	5,280,889	404,929	10,729,079	6,569,087	2,513,519	5,227,623	435,259	3,297,821	2,877,362	13,544,265	1,409,270	4,807,482	57,096,586
Diffuse urban	152,358	10,531	253,915	127,528	53,304	131,534	18,444	78,026	91,442	265,817	41,789	115,034	1,339,722
Septic tanks	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Industrial													
Point sources	1		2	4	2	3	1					5	18
Diffuse industrial	25,725	4,154	71,388	34,556	17,351	50,594	6,911	25,219	32,981	44,839	15,289	83,935	412,940
Background													
Bog, fen, marsh, swamp	11,225	749	32,780	20,653	32,563	2,363	38,701	1,330	8,382	2,009	3,721	321,763	476,238
Orchard, wood, forest	184,348	25,323	194,086	85,598	90,077	262,521	244,912	140,742	204,372	239,948	163,110	803,107	2,638,144
Inland water	26,406	2,784	8,849	11,072	3,290	8,056	11,125	7,156	6,952	14,836	7,396	124,323	232,246
Land area	2,417,258	206,289	2,367,834	1,054,232	794,890	2,036,933	1,446,775	798,165	1,693,305	1,536,042	1,106,525	5,798,485	21,256,734

Table 02.03 Total TP loads (kg y^{-1}) from agricultural, household, industrial and background sources to the River Basin Districts of England, Wales and Scotland. Calculations used the data presented in Table 02.01 and Table 02.02 as described in the text.

Sector	Anglian	Dee	Humber	North West	Northumbria	Severn	Solway Tweed	South East	South West	Thames	Western Wales	Scotland	Total
Agriculture													
Glasshouses	120,532	1,169	111,043	84,146	6,882	65,518	3,239	118,899	53,778	104,370	14,261	31,082	714,919
Improved grassland	5,546	50,747	374,022	94,605	51,338	516,019	126,911	18,059	261,450	27,440	383,629	206,588	2,116,356
Grassland, moor, heath	25,957	1,600	8,620	6,226	4,514	9,040	10,327	1,993	4,211	4,888	6,668	58,433	142,478
Arable cereals	164,135	8,581	169,465	33,431	62,132	183,200	46,964	81,197	153,295	153,502	3,436	69,310	1,128,649
Field horticulture	211,447	9,552	649,012	46,804	58,131	249,455	58,756	93,480	138,630	184,114	46,486	232,780	1,978,648
Cattle	2,586	57,222	471,387	66,756	29,183	511,214	83,991	47,242	304,815	69,548	354,401	134,098	2,132,443
Pigs	64,834	4,826	453,165	10,219	6,471	79,472	4,491	15,260	49,633	30,575	1,273	30,997	751,214
Sheep	13,366	28,807	255,868	52,933	44,846	609,574	80,751	30,966	124,688	36,244	471,771	127,635	1,877,450
Poultry	81,314	13,932	252,469	27,323	7,228	270,825	14,800	29,976	70,217	43,250	5,668	34,692	851,694
Horses	195	1,482	15,648	1,919	881	17,900	703	4,357	7,762	7,473	10,457	2,377	71,154
Household													
Humans	2,233,166	171,235	4,537,080	2,777,916	1,062,909	2,210,641	184,061	1,394,573	1,216,770	5,727,558	595,948	2,032,973	24,144,830
Diffuse urban	126,458	8,740	210,749	105,848	44,242	109,173	15,308	64,761	75,897	220,628	34,685	95,478	1,111,969
Septic tanks	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Industrial													
Point sources	7,140	-	52,800	116,600	200,000	85,864	11,100	-	-	-	-	1,104,800	1,578,304
Diffuse industrial	20,611	3,259	55,855	27,511	13,526	38,776	5,273	19,217	24,842	35,542	11,137	60,253	315,801
Background													
Bog, fen, marsh, swamp	-	-	-	-	-	-	-	-	-	-	-	-	-
Orchard, wood, forest	12,904	506	3,882	1,712	1,802	5,250	4,898	2,815	4,087	4,799	3,262	16,062	61,980
Atmospheric	300,633	25,868	287,680	130,937	96,703	247,655	178,063	98,642	205,977	190,260	135,741	745,548	2,643,706
Total	3,390,826	387,527	7,908,746	3,584,886	1,690,789	5,209,576	829,635	2,021,437	2,696,052	6,840,191	2,078,823	4,983,108	41,621,595

Table 02.04 Soluble/Molybdate Reactive P loads (kg y^{-1}) from agricultural, household, industrial and background sources to the River Basin Districts of England, Wales and Scotland. Calculations used the data presented in Table 02.01, Table 02.02 and Table 02.03 as described in the text.

Sector	Anglian	Dee	Humber	North West	Northumbria	Severn	Solway Tweed	South East	South West	Thames	Western Wales	Scotland	Total
Agriculture													
Glasshouses	120,532	1,169	111,043	84,146	6,882	65,518	3,239	118,899	53,778	104,370	14,261	31,082	714,919
Improved grassland	2,218	20,299	149,609	37,842	20,535	206,408	50,764	7,224	104,580	10,976	153,452	82,635	846,542
Grassland, moor, heath	11,762	900	4,858	3,495	2,663	4,899	6,119	998	2,160	2,247	3,623	40,129	83,854
Arable cereals	57,447	3,003	59,313	11,701	21,746	64,120	16,437	28,419	53,653	53,726	1,203	24,259	395,027
Field horticulture	95,151	4,298	292,055	21,062	26,159	112,255	26,440	42,066	62,384	82,851	20,918	104,751	890,391
Cattle	1,448	32,044	263,977	37,383	16,343	286,280	47,035	26,455	170,696	38,947	198,464	75,095	1,194,168
Pigs	35,659	2,654	249,241	5,620	3,559	43,710	2,470	8,393	27,298	16,816	700	17,048	413,168
Sheep	6,683	14,404	127,934	26,467	22,423	304,787	40,376	15,483	62,344	18,122	235,885	63,817	938,725
Poultry	28,460	4,876	88,364	9,563	2,530	94,789	5,180	10,491	24,576	15,137	1,984	12,142	298,093
Horses	98	741	7,824	959	440	8,950	351	2,179	3,881	3,737	5,228	1,189	35,577
Household													
Humans	1,898,191	145,550	3,856,518	2,361,228	903,473	1,879,045	156,452	1,185,387	1,034,255	4,868,424	506,555	1,728,027	20,523,106
Diffuse urban	63,229	4,370	105,375	52,924	22,121	54,587	7,654	32,381	37,949	110,314	17,343	47,739	555,985
Septic tanks													
Industrial													
Point sources	7,140	-	52,800	116,600	200,000	85,864	11,100	-	-	-	-	1,104,800	1,578,304
Diffuse industrial	9,508	1,426	24,269	12,495	5,820	15,923	2,138	7,762	9,695	15,968	3,896	19,990	128,888
Background													
Bog, fen, marsh, swamp	-	-	-	-	-	-	-	-	-	-	-	-	-
Orchard, wood, forest	5,744	190	1,697	774	646	2,213	1,417	1,309	1,824	2,222	1,190	4,731	23,956
Atmospheric	300,633	25,868	287,680	130,937	96,703	247,655	178,063	98,642	205,977	190,260	135,741	745,548	2,643,706
Total	2,643,904	261,793	5,682,557	2,913,197	1,352,043	3,477,001	555,236	1,586,087	1,855,048	5,534,116	1,300,444	4,102,983	31,264,409

The TP loads to watercourses differ widely between RBD. Perhaps unsurprisingly, the heaviest P loads occur in the heavily populated and/or extensive RBD of Humber, Thames and Severn (Table 02.03). The lightest P loads occur in the small, sparsely-populated RBD of Northumbria, Solway Tweed and Dee. In terms of their area-normalised annual TP load, the RBD of Thames (4.41 kg ha^{-1} y^{-1}), North West (3.37 kg ha^{-1} y^{-1}) and Humber (3.33 kg ha^{-1} y^{-1}) were highest, whilst the RBDs of Anglian (1.39 kg ha^{-1} y^{-1}), Scotland (0.84 kg ha^{-1} y^{-1}) and Solway Tweed (0.57 kg ha^{-1} y^{-1}) were lowest. These values are consistent with previous studies (Robson and Neal 1997; Nedwell et al. 2002). The agricultural contributions to the TP loads of RBDs followed the order Western Wales (62.4%), Solway Tweed (51.9%), Severn (48.2%), Dee (45.9%), South West (43.3%), Humber (34.9%), South East (21.8%), Anglian (20.35%), Scotland (18.6%), Northumbria (16.1%), North West (11.8%) and Thames (9.7%; Table 02.05). There is a notably high contribution from industrial point sources to the P load of Scottish waters (23.4%), and large contributions (>70%) from household sources to the P loads of the South East, North West and Thames RBDs (Table 02.05).

Table 02.05 A summary of total TP loads (t y^{-1}) and percentage contributions from agricultural, household, industrial and background sources to the River Basin Districts of England, Wales and Scotland. Calculations used the data presented in Table 02.01, Table 02.02 and Table 02.03 as described in the text. Orchards, woods and forests are included under background loads.

Sector	Anglian	Dee	Humber	North West	Northumbria	Severn	Solway Tweed	South East	South West	Thames	Western Wales	Scotland	Total
Agriculture	690	178	2,761	424	272	2,512	431	441	1,168	661	1,298	928	11,765
Household	2,360	180	4,748	2,884	1,107	2,320	199	1,459	1,293	5,948	631	2,128	25,257
Industrial	28	3	109	144	214	125	16	19	25	36	11	1,165	1,894
Background	314	26	292	133	99	253	183	101	210	195	139	762	2,706
Total	3,391	388	7,909	3,585	1,691	5,210	830	2,021	2,696	6,840	2,079	4,983	41,622
Percentage contributions from													
Agriculture	20.3	45.9	34.9	11.8	16.1	48.2	51.9	21.8	43.3	9.7	62.4	18.6	28.3
Household	69.6	46.4	60.0	80.4	65.5	44.5	24.0	72.2	47.9	87.0	30.3	42.7	60.7
Industrial	0.8	0.8	1.4	4.0	12.6	2.4	2.0	1.0	0.9	0.5	0.5	23.4	4.6
Background	9.2	6.8	3.7	3.7	5.8	4.9	22.1	5.0	7.8	2.9	6.7	15.3	6.5

Objective 03: Comparison of P loads and source apportionment for England, Wales, Scotland and Northern Ireland.

The TP loads for England, Wales and Scotland were estimated based on the land use statistics, livestock numbers and human populations derived from values presented in Objective 02 (Table 03.01). There are three cross-boarder RBDs in the UK; the Severn and the Dee RBDs span the English-Welsh boarder and the Solway Tweed RBD spans the English-Scottish boarder. Land use statistics, livestock numbers and human populations for these RBDs, obtained in Objective 02, were partitioned appropriately into their respective national areas. Data for the apportionment of phosphorus sources for England were derived from the values obtained in Objective 02 for the Anglian, Humber, North West, Northumbria, South East, South West and Thames RBDs and the areas of the Solway Tweed, Dee and Severn covering England. Data for the apportionment of phosphorus sources for Wales were derived from the values obtained in Objective 02 for the Western Wales RBD and the areas of the Dee and Severn covering Wales. Data for the apportionment of phosphorus sources for Scotland were derived from the values obtained in Objective 02 for the Scotland RBD and the area of the Solway Tweed covering Scotland. This approach allows appropriate export coefficients to be used for the different geo-climatic regions across the UK. It can be observed that England contributes most to the TP load of UK waters (Table 03.02). Agriculture in England contributes 26.0% of this TP load. Household and Industrial diffuse and point sources together contribute 69.2% to the TP load of English waters. Agriculture contributes less (22%) of the TP load of Scottish waters, and more (57%) of the total TP load to Welsh waters. Livestock and improved grasslands contribute 66% of the agricultural P loads of England, Wales and Scotland (Table 03.03). Recently, Smith et al. (2005) estimated that agriculture contributed about 47% of all P exports to inland and coastal waters in Northern Ireland (Table 03.03, 03.05). They observed that agricultural land contributed 1130 tonnes of P, and other diffuse sources from urban areas, moorland, forests and peat bogs, contributed a further 165 tonnes of P to these waters. The human contribution to the overall P budget was divided into mains-sewered households (945 tonnes), households connected to septic tanks (118 tonnes) and industrial discharges (40 tonnes; Table 03.03, 03.05).

Table 03.01 Areas of land use classes (ha), livestock numbers and human populations for England, Wales and Scotland.

Sector	England	Wales	Scotland
Agriculture			
Glasshouses	2,016	43	96
Improved grassland	2,852,488	722,142	972,524
Grassland, moor, heath	2,003,810	697,092	3,655,244
Arable cereals	1,933,215	18,218	169,466
Field horticulture	2,659,274	80,077	455,021
Cattle	5,658,740	1,266,693	1,980,014
Pigs	4,211,534	6,739	470,672
Sheep	15,817,720	9,829,626	7,997,698
Poultry	130,217,600	1,773,187	14,723,223
Horses	218,096	40,393	29,546
Household			
Humans	49,031,565	3,003,009	5,062,011
Diffuse urban	1,135,705	76,488	127,529
Septic tanks	NA	NA	NA
Industrial			
Point sources	11	1	6
Diffuse industrial	292,761	32,901	87,278
Background			
Bog, fen, marsh, swamp	115,177	6,436	354,625
Orchard, wood, forest	1,323,183	287,465	1,027,496
Inland water	89,165	10,841	132,240
Land area	12,415,540	1,937,109	6,904,085

Table 03.02 A comparison of TP inputs (tonnes y^{-1}) and percentage contributions from agricultural, household, industrial and background sources to the waters of England, Scotland, Wales and Northern Ireland. Data for Northern Ireland are adapted from Smith et al. (2005). Orchards, woods and forests are included under background loads.

Sector	England	Wales	Scotland	Northern Ireland	Total
Agriculture	8,391	2,151	1,223	1,023[a]	12,788
Household	21,677	1,333	2,246	988[b]	26,245
Industrial	645	70	1,179	40	1,934
Background	1,561	243	902	123	2,829
Total	32,274	3,797	5,551	2,174	43,796
Percentage contributions from					
Agriculture	26.0	56.6	22.0	47.1	29.2
Household	67.2	35.1	40.5	45.4	59.9
Industrial	2.0	1.8	21.2	1.8	4.4
Background	4.8	6.4	16.2	5.7	6.5

[a] Adjusted for rural population (-107 t TP y^{-1}) as described by Smith et al., 2005. [b] Adjusted for P removal programme (-145 t TP y^{-1}) as described by Smith et al., 2005, see Table 03.05.

Table 03.03 An itemised comparison of TP inputs (tonnes y^{-1}) to waters of England, Scotland, Wales and Northern Ireland. Data for Northern Ireland are adapted from Smith et al. (2005).

Sector	England	Wales	Scotland	Northern Ireland	Total
Agriculture					
Glasshouses	669	14	32		715
Improved grassland	1,247	578	292	328[a]	2,444
Grassland, moor, heath	62	14	67	223[a]	365
Arable cereals	1,011	16	102	130[a]	1,259
Field horticulture	1,632	71	276	342[a]	2,321
Cattle	1,408	535	190	-	2,132
Pigs	715	2	34	-	751
Sheep	809	885	184	-	1,877
Poultry	788	20	44	-	852
Horses	51	17	3	-	71
Household					-
Humans	20,734	1,270	2,141	800[b]	24,945
Diffuse urban	943	63	106	70	1,182
Septic tanks	NA	NA	NA	118	118
Industrial					
Point sources	416	46	1,116		1,578
Diffuse industrial	229	24	63	40	356
Background					
Bog, fen, marsh, swamp	-	-	-	73	73
Orchard, wood, forest	36	6	21	22	84
Atmospheric	1,526	237	881	28	2,672
Total	32,274	3,797	5,551	2174	43,796

[a] Adjusted for rural population (-107 t TP y^{-1}) as described by Smith et al., 2005. [b] Adjusted for P removal programme (-145 t TP y^{-1}) as described by Smith et al., 2005, see Table 03.05.

Table 03.04 An itemised comparison of SRP inputs (tonnes y^{-1}) to waters of England, Wales and Scotland.

Sector	England	Wales	Scotland	Total
Agriculture				
Glasshouses	669	14	32	715
Improved grassland	499	231	117	847
Grassland, moor, heath	31	8	45	84
Arable cereals	354	6	36	395
Field horticulture	734	32	124	890
Cattle	788	299	106	1,194
Pigs	393	1	19	413
Sheep	404	442	92	939
Poultry	276	7	15	298
Horses	26	9	1	36
Household				
Humans	17,624	1,079	1,820	20,523
Diffuse urban	471	32	53	556
Septic tanks	NA	NA	NA	NA
Industrial				
Point sources	416	46	1,116	1,578
Diffuse industrial	99	9	21	129
Background				
Bog, fen, marsh, swamp	-	-	-	-
Orchard, wood, forest	16	2	6	24
Atmospheric	1,526	237	881	2,644
Total	24,327	2,453	4,484	31,264

Table 03.05 Phosphorus inputs to inland and coastal waters of Northern Ireland (Smith et al. 2005).

	CORINE land class	Total phosphorus (tonnes y^{-1})
Agricultural contribution		
Complex cultivation	2.41, 2.42, 2.43	369
Good pasture	2.3.1.1	355
Poor pasture	2.3.1.3	47
Mixed pasture	2.3.1.2	203
Arable	2.1.1 1	156
Rural population		-107
Total		1023
Other diffuse contribution		
Urban	1.1.1–1.4.2	70
Moorland	3.2.1, 3.2.2	39
Forest	3.1.1, 3.1.2	22
Peat bogs	3.2.4	31
Others		3
Total		165
Human contributions		
Urban sewered		945
P removal programme		-145
Industrial discharges (point source)		40
Septic tanks		118
Total		958
Rainfall		28
Overall total		2174

Objective 04. An appraisal of the impact of P pollution on the aquatic ecosystem in the UK, with specific consideration of the impacts of various P-sources in different seasons and sensitivities of water bodies to eutrophication

04.01 Definition and threshold P concentrations for eutrophication

Eutrophication was defined by the Environment Agency of England and Wales (1998) as "the enrichment of waters by inorganic plant nutrients which results in the stimulation of an array of symptomatic changes. These include the increased population of algae and/or other aquatic plants affecting the quality of the water and disturbing the balance of organisms present within it". The principal effects of eutrophication are: increased aquatic plant growth and changes in species composition, oxygen depletion and pH variability in the water, and consequent impacts on food-webs and amenity use (Morse et al. 1993; Sharpley et al. 1994; Pierzynski et al. 2000; Reynolds and Davies 2001; Mainstone and Parr 2002; Defra 2004; Haygarth et al. 2004a, 2005). The sensitivity of a water body to eutrophication depends upon climate (rainfall, light and temperature), water chemistry, residence time and depth (Vollenweider and Kerekes 1980; Sharpley et al. 2002; Bennion et al. 2005a; Heathwaite et al. 2005). Thus, shallow, standing waters are more susceptible to eutrophication than fast-flowing rivers (Haygarth et al. 2003; Bennion et al. 2005a). Similarly, hard water lakes are generally more susceptible to eutrophication than soft water lakes, reflecting the greater fertility of base rich soils (Haygarth et al. 2003).

The trophic state of most lakes and rivers in the UK appears to be determined by P status (Mainstone and Parr 2002; Defra 2004). Eutrophic symptoms in lakes are generally related to SRP rather than to TP (Reynolds and Davies 2001), and eutrophic symptoms in rivers are often linked to transient increases in SRP concentrations (Jarvie et al. 2002b, 2004). This is because SRP is immediately available for the growth of aquatic algae and plants. Other forms of P are less available. Dissolved organic P occurs mostly as esters of inositol, which are refractile compounds, and PP quickly sediments to the bottom of lakes and may not contribute greatly to algal P nutrition. Point sources contribute most immediately bioavailable SRP throughout the year, but diffuse P sources, particularly from agriculture, contribute most PP that can be accessed by benthic algae and rooted plants (Mainstone and Parr 2002). Both biota and sediments will accumulate P throughout the summer and P will be released during the autumn and winter. Strong microbial activity in the spring can produce a flush of bioavailable P prior to the season of new growth (Mainstone and Parr 2002).

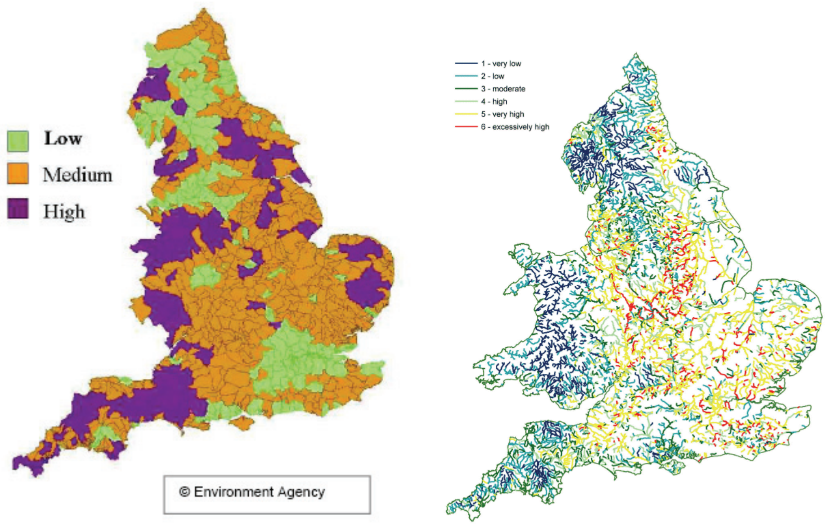
It is believed that critical, threshold values for bioavailable N and P concentrations in water must be exceeded for eutrophication to occur (Vollenweider and Kerekes 1980; Reynolds and Davies 2001), and that the atomic N:P ratio distinguishes between N and P limitation during subsequent symptomatic changes (Mainstone and Parr 2002). From their pioneering studies, Vollenweider and colleagues suggested that P concentrations above 10 $\mu g\ P\ L^{-1}$ could stimulate algal growth in lakes (Vollenweider and Kerekes 1980), and the OECD (1982) suggested that the threshold concentration of bioavailable P at which the standing stock of phytoplankton reached nuisance proportions in lakes was 35 $\mu g\ P\ L^{-1}$. Work by Redfield et al. (1963) suggested that P limited growth of marine plankton when the atomic N/P quotients were >16 (corresponding to a N:P mass ratio of 7.3:1). Subsequent studies of nutrient limitation in freshwater lakes have indicated that P limitation occurs at an atomic N/P quotient >10, although concentrations of bioavailable P must typically fall below 5-10 μg P L^{-1} for P limitation to occur (Reynolds and Davies 2001). Mainstone and Parr (2002) suggested pragmatic management targets of between 20 and 100 μg SRP L^{-1} for UK rivers, depending on river type, and the Environmental Agency Rivers Task Team suggested threshold values for 'good ecological status' of 40-50 μg SRP L^{-1} for non-calcareous rivers and 120 μg SRP L^{-1} for calcareous rivers (Duncan et al. 2006). These standards are being used in the first RBD plan for England and Wales (www.environment-agency.gov.ul/wfd).

04.02 Sensitivity of UK surface waters to eutrophication

The risk of eutrophication is now widespread in the UK (EA 2000; Defra 2004; Haygarth et al. 2004a) and many inland waters contain excessive P concentrations derived from anthropogenic sources (Muscutt and Withers 1996; Skinner et al. 1997; Foy and Bailey-Watts 1998; Withers and Lord 2002; Bennion et al. 2005a; Heathwaite et al. 2005; Smith et al. 2005).

In general, the majority of rivers with high altitude catchments have median TP concentrations < 30 μg TP L^{-1}, but lowland rivers often exceed this value (Mainstone and Parr 2002). A NRA survey in the early 1990s found that, out of 915 UK surface waters (lakes and reservoirs) surveyed, 169 (18.5%) had sufficiently high densities of blue-green algae to warrant alerts to owners and Environmental Health Officers (Anon 1992). In a survey of 98 major rivers of England and Wales, Muscutt and Withers (1996) found that median SRP concentrations exceeded 100 μg SRP L^{-1} in 78 rivers and they reported values in excess of 1000 μg SRP L^{-1} in 16 rivers. Similar observations were made by Mainstone et al. (1995, quoted by Parr et al. 1999) for rivers in all of the former NRA regions and the EA (1998) designated 61 rivers and canals and 14 lakes and reservoirs in England and Wales as sensitive to eutrophication. Scottish rivers generally had lower SRP concentrations with most rivers having <100 μg SRP L^{-1} (Marsden et al. 1997; Ferrier et al. 2001). Marsden and Mackay (2001) attributed the P pollution incidence in Scottish rivers to sewage (45%), diffuse agriculture (30%), point-source agriculture (31%) and urban drainage (15%). In the UK as a whole, the highest mean concentrations were associated with rivers draining the most populated catchments and receiving major point-source inputs from sewage treatment works. Currently, 4% of the river length in Scotland, 8% of the river length in Wales, 23% of the river length in Northern Ireland, and 58% of the river length in England has average SRP concentrations above 100 μg L^{-1} (http://www.defra.gov.uk/environment/statistics/inlwater/iwnutrient.htm; Figure 04.01). Although the ecological quality of many larger rivers in England and Wales has improved recently due to a reduction in point source discharges, the ecological quality of smaller tributaries and streams has not always improved (Jarvie et al. 2002b, 2004).

Figure 04.01 Left: Catchments at risk from diffuse (agricultural) P loads (EA 2004). Right: The SRP concentrations in rivers of England and Wales (EA 2000).



In 1998, Foy and Bailey-Watts observed that many (73%) of the lakes in the Scottish Highlands was oligotrophic, and a survey of 173 lochs by Marsden and Mackay (2001) indicated that only 20 had elevated P concentrations. However, of the 128 lakes and ponds surveyed by Foy and Bailey-Watts (1998) in England and Wales, 69% had TP concentrations greater than 100 μg L^{-1} and only 8% had TP concentrations less than 35 μg P L^{-1} and a contemporaneous survey of 102 Sites of Special Scientific Interest that were suspected to be eutrophic, 84% showed symptoms (Carvalho and Moss 1995). Recently, Heathwaite et al. (2005) applied an export coefficient model based on land use and animal stocking data to determine P concentrations in all lakes in Great Britain of more than 1 ha area (>14,000 lakes). Their analysis suggested that 51% of lakes in Great Britain (and 88% in England) were at risk of not meeting the "good ecological status" objective of the EU Water Framework Directive because of diffuse agricultural P loads. In a similar study, Bennion et al. (2005a) assessed the risk of eutrophication in these lakes based on four parameters: lake importance, calculated P load, retention time (used as a measure of sensitivity to P enrichment) and Wederburn depth (used as a measure of potential response to remediation). They suggested that, of the 2,362 important lakes in Great Britain, 1,736 lakes had a low risk of eutrophication but were potentially threatened because of their high sensitivity to enrichment, 332 lakes were damaged by eutrophication but were likely to respond to rehabilitation, and a further 212 were likely to be damaged with relatively poor chance of recovery.

As part of their River Basin Characterisation reports, the Environment Agency (EA) and Scottish Environment Protection Agency (SEPA) have published the results of risk assessment of pressures on freshwater sites from diffuse P sources (EA 2004; SEPA 2004). The EA (2004) report indicates that 47.4% of total river length, 36.8% of lakes and 17.4% of groundwaters are at risk, or probably at risk, from diffuse agricultural P sources (Figure 04.01). Diffuse agricultural P loads pose the greatest risks in the arable and vegetable growing areas of Norfolk, Suffolk, Lincolnshire, Yorkshire and Nottinghamshire, in the west in the dairy farming regions of Cornwall, Devon, Somerset, Dorset, Wiltshire, Herefordshire, Shropshire, Staffordshire and the cattle and sheep areas of Lancashire and Cumbria. In Scotland, diffuse pollution has been considered in general, not in terms of individual pollutants. The SEPA (2004) report indicated that, of the 2,380 river water bodies examined, approximately 531 (22%) water bodies are at risk, or probably at risk, from diffuse pollution, whilst 1,849 (78%) are not at risk, or probably not at risk.

04.03 Forms of aquatic phosphorus

Phosphate is present in watercourses in many forms (Ryden et al. 1973; Haygarth 1997; Haygarth and Jarvis 1999; Pierzynski et al. 2000; Schoumans and Chardon 2003; James and Barko 2005; Johnston and Dawson 2005). These forms are arbitrarily divided into particulate P (PP) and dissolved P (DP). Dissolved P is defined as the fraction passing through a 0.45 µm filter, and comprises molybdate-reactable P (MRP or SRP = orthophosphate plus some acid-labile organic P compounds) and molybdate-unreactable, dissolved organic P (SOP). Common forms of organic P include inositol phosphates, nucleic acids, phospholipids and phosphoglycerides.

Forests loose P in DOP and MRP forms, but rarely as PP (Ryden et al. 1973; Lemunyon and Gilbert 1993; McGuckin et al. 1999; Reynolds and Davies 2001). Similarly, grasslands loose most P in soluble forms due to their dense vegetative cover, which prevents particulate losses (Lemunyon and Gilbert 1993; Haygarth 1997). Grassland improvement may increase P loads to watercourses, but the extent and duration of these accelerated P losses depend on the method of improvement and fertiliser applied (Roberts et al. 1989). It is estimated that about 60% of TP losses from improved grassland occur as DP and the remainder as PP (Haygarth and Jarvis 1997; McGuckin et al. 1999). Improved grasslands often have less DOP losses and more MRP losses than non-improved grasslands (Haygarth et al. 1998; McGuckin et al. 1999) and, occasionally, greater PP losses due to improved drainage. Artificially-drained fields loose large quantities of both DP, due to a reduced opportunity for adsorption, and PP, due to preferential flow through macropores and field drains (Foster et al. 2003). Losses from arable land are occasionally dominated by DP forms also. In well drained heavy clay-loam soils, SRP may constitute 62-71%, DOP 4-25% and PP 4-17% of the total P losses (Culley et al. 1983; Heckrath et al. 1995). However, although the ratio of SRP/DOP remains similar, much higher values for PP (64-97% of TP losses) are often recorded (Sharpley et al. 1994; Heckrath et al. 1995; Dils and Heathwaite 1996; Haygarth 1997; Catt et al. 1998; McGuckin et al. 1999; McDowell et al. 2001; Reynolds and Davies 2001; Heathwaite et al. 2003b Johnston and Dawson 2005). Interestingly, in the arable catchments at ADAS Rosemaund the contribution of SRP to TP losses was 15-28% when flowrates were high, but increased to 44-69% when flowrates were low (Johnes and Hodgkinson 1998). Similarly, Dils and Heathwaite (1996) showed that the forms of P lost from agricultural land during storms changed from autumn to spring. During storms in the early autumn most TP was lost as MRP, whereas later in the year TP losses were dominated by PP. The relative contributions of different forms of P to TP losses is also influenced by the form of fertilizer applied to the land (Heathwaite et al. 1998; Johnes and Hodgkinson 1998; Sharpley et al. 2002). There are high P losses from land growing horticultural and other wide row crops since they require high P inputs, there is more soil disturbance and the soil is often left barren for longer periods. In addition, planting in rows promotes rapid runoff and soil erosion through channellisation. It is noteworthy that the particles transported in runoff normally have higher P concentrations than the soil from which they originated (Pierzynski et al. 2000; Haygarth et al. 2005b). This enrichment occurs because of the greater sorption capacity of finer clay particles and the relatively high concentration of P in organic matter relative to silt and sand particles. The size of the particles in runoff depends on the intensity of rainfall, larger particles occur in runoff at higher hydraulic energies. In urban areas, 15 to 80% of the diffuse TP load is MRP (Ryden et al. 1973; McGuckin et al. 1999) and the P effluent from STWs is also predominantly MRP (Table 02.01). It has been estimated that 26% to 88% of the TP load of British rivers is associated with PP (Walling 2005). This implies a large contribution of agriculture to the P loads of rivers. Furthermore, about 60 to 70% of the PP in rivers is defined as 'non-apetite inorganic phosphate', which contributes to the total bioavailable P (Walling et al. 2001).

04.04 Seasonality of P loading to surface waters

Eutrophication is linked primarily to P concentrations during periods of ecological sensitivity (Mainstone and Parr 2002; Jarvie et al. 2005). Thus, SRP concentrations in early spring are often found to be the best predictors of algal growth. The P load from STWs is generally constant throughout the year, but the contribution of diffuse agricultural sources to instantaneous P loads is greater with increased hydrological activity (Muscutt and Withers 1996; Sharpley et al. 2002; Bowes et al. 2005; Haygarth et al. 2005b). Thus, the spring and summer P loads from STWs make an immediate contribution towards eutrophication, whereas the winter agricultural P loads make a delayed contribution to eutrophication during the subsequent spring and summer. Over the past fifty years, there has been a trend towards greater extremes of weather in the UK, resulting in lower summer baseflow and higher winter flows (Marsh and Sanderson 1997). The trend towards lower summer baseflow in rivers reduces the capacity for sewage dilution resulting in elevated P concentrations. The trend towards higher winter flows increases agricultural PP loads. The impact of these changes is most severe in the south and east of England, which experiences low rainfall and high evapotranspiration. From a study of water quality at 54 river sites in 7 UK catchments with diverse land use characteristics, Jarvie et al. (2005) observed that P loads from STWs rather than diffuse agricultural P loads posed the most significant risk for eutrophication, even in rural areas. They observed that, although annual P budgets are dominated by diffuse P inputs in storm runoff from agricultural land, the risk of eutrophication was largely linked to SRP concentrations during times of ecological sensitivity (spring/summer low flow periods) when biological activity was at its highest. They also observed that PP loads from agriculture could remove SRP at times of ecological risk, provided that SRP concentrations were greater than about 50 $\mu g\ L^{-1}$.

Objective 05. Comparing updated P loads and apportionment figures with previously published data for the UK and fellow EU countries with similar climatic conditions and land use.

05.01 Loads and apportionment studies for the UK

Several studies in the last 20 years have sought to determine the relative contribution of agriculture to the P loads to UK waters (Table 05.01). In 1989, the Soap and Detergent Industry Association (SDIA) estimated that, of the 105 kt P y^{-1} delivered to UK rivers, about 43% was derived from agriculture (17% from soil run-off, 2% from silage losses and 25% from livestock), 53% came from sewage effluent (with human faecal matter and detergents contributing roughly equal amounts) and 3% was derived from direct industrial charges. Similar values were reported by Morse et al. (1993), who estimated that of the TP load to UK waters (82 kt P y^{-1}) approximately 43% was of agricultural origin (14% from fertiliser use, 29% from livestock), approximately 43% was of domestic origin (24% from human sources, 19% from detergents), and the remainder was from industrial (8%) and background (6%) sources (Morse et al. 1993). Contemporaneously, Withers (1993, quoted by Mainstone et al. 1996) suggested that agriculture contributed about 36% to the P load of UK waters and Haygarth et al. (2003), using the AERC Export Coefficient Model, estimated that agriculture was responsible for 43% of the P load to UK waters in 1991. By contrast, other authors suggested that agriculture contributed only about 20% of the P loads to surface waters in this period, with sewage being the main culprit (Anon 1992).

Table 05.01 The percentage contributions of different sectors to the TP loads of UK waters.

Source	SIDA (1989)	Morse et al. (1993)	Defra (2004)	This study*
Agriculture	**43**	**43**	**53**	**28.3**
Soil run-off	17			
Silage	2			
Fertiliser		14		
Livestock	25	29		
Domestic	**53**	**43**	**30**	**60.7**
Sewage	20-25	24	25	
Detergent	20-25	19	5	
Industrial	**3**	**8**		**4.6**
Background		**6**		**6.5**
Total (kt y^{-1})	**105**	**82**		**42**

*Values are based on figures from P source apportionment at the level of the RBD (Objective 02) and do not include NI

The contribution of STWs to the P load of UK waters has declined in recent years, as has the use of P containing detergents. Thus, it is likely that the relative contribution of agriculture to P loads of UK waters has increased from earlier estimates. Indeed, Defra (2004) suggested that agriculture was responsible for over 50% of the TP loads to UK surface waters in 2002. However, the source apportionment undertaken in here (Table 02.05) attributes only 28.3% of the total TP load to UK waters to agriculture (51.7% to land use and 48.3% to livestock), with about 60.7% attributed to human and household waste (65% to sewage, 6% to detergents, 24% to waste, 5% to diffuse sources), 4.6% to industrial sources (point and diffuse) and 6.5% to background sources (atmospheric loads). If a higher export coefficient for human point sources is used in the calculation, a value of 22.5% is obtained for the contribution of diffuse agricultural sources to the total TP load of UK waters.

Table 05.02 Summary of recent estimates of the contribution from diffuse sources to the TP (or SRP) loads to various UK rivers and catchments.

River / catchment	Date	Diffuse sources (%)	Reference
Avon catchment (various locations)	1994-1996	7 - 91	Bowes et al. (2005)
Avon near Evesham	1994-1996 2000-2001	< 24	Bowes et al. (2005) Hilton et al. (2002)
Hampshire Avon		43	Mainstone & Parr (2002)
Warwickshire Avon		30	Mainstone & Parr (2002)
Upper River Cherwell	1998-1999	56	May et al. (2001)
Frome catchment	1998	60	Hanrahan et al. (2001)
River Kennet (Mildenhall)	1998 1999	29 (SRP) 45 (SRP)	Jarvie et al. (2002c)
Taw catchment (various locations)	1998-1999	60 – 100	Wood et al. (2005)
River Thame	1999	36-53	Cooper et al. (2002)
River Windrush	Mid 1990s	89	Heathwaite et al. (2003a)
Pevensey Levels		14	Mainstone & Parr (2002)
Glaslyn/Dwyryd estuary		26	Parr et al. (1999)
Mawddach estuary		52	Parr et al. (1999)
Dyfi estuary		56	Parr et al. (1999)
Loch Leven, Scotland	1995	57	Bailey-Watts & Kirika (1999)

The TP load estimated in the present study is far lower than that estimated by Morse et al. (1993). This suggests a large reduction in TP loads since the 1990s. A decline in TP loads to UK waters since the 1990s might be expected due to improvements in sewerage treatment, a reduction of actual industrial and detergent P loads, reduced P fertiliser applications and a decrease in livestock numbers. Furthermore, it is clear that the actual P loads to waters in England and Wales are declining, with a 20% reduction in the percentage of total river length with average phosphate concentrations greater than 0.1 mg P l^{-1} between 1990 and 2005 (http://www.environment-agency.gov.uk). However, several caveats should be added to any comparison of absolute TP loads and/or TP source apportionment between the present study and that of Morse et al. (1993).

(1) Morse et al. (1993) used a value of 0.62 kg TP cap^{-1} y^{-1} for the *per capita* human contribution to the TP load to surface waters, including P from detergents. This can be compared with the lower estimate of the human contribution to TP loads of 0.43 kg TP cap^{-1} y^{-1} used in the present study, which is consistent with a reduction in the use of P in detergents as well as increased use of secondary and tertiary sewage treatment since the 1990s. However, the higher TP load *per capita* from human sewerage (0.61 kg TP cap^{-1} y^{-1}) calculated recently by ADAS from an amended inventory of P loads from STWs suggests that Morse et al. (1993) may have underestimated the *per capita* human contribution to the TP load to UK surface waters. Neither Morse et al. (1993) nor the present study took account of regional variation when calculating the human contribution to TP loads to surface waters.

(2) The industrial contribution to the TP load to UK waters appears to have declined greatly since the 1990s. However, it is important to note that Morse et al. (1993) simply assumed that industrial P sources were equivalent to 20% of the domestic P contribution, whilst data from the present study suggest that industrial P sources are equivalent to only 7% of domestic sources (Table 03.02).

(3) The methods used to estimate TP loads from agricultural sources differ substantially between the present study and that of Morse et al. (1993). In contrast to the present study, Morse et al. (1993) did not account for any regional differences in export coefficients for P from agricultural land or livestock, and used a single (high) export coefficient for all agricultural land rather than modifying export coefficients to reflect contrasting land uses.

These caveats, which suggest that Morse et al. (1993) may have underestimated the human contribution and/or overestimated the agricultural contribution to the TP load to UK surface waters, could provide the reason why the agricultural contribution to the TP load of UK surface waters was previously estimated to be 43% for the UK (Morse et al. 1993), whereas the present study indicates a value of 28% (Table 05.01).

05.02 Loads and apportionment studies for UK coastal waters

The UK coastal waters are divided into these areas: North Sea, English Channel, Celtic Sea, Irish Sea and Atlantic Ocean (Littlewood and Marsh 2005). Several studies have estimated the annual orthophosphate and TP loads to these waters (e.g. Robson and Neal 1997; Littlewood et al. 1998; Watts and Littlewood 1998; Parr et al. 1999; Nedwell et al. 2002; OSPAR Commission 2003; Littlewood and Marsh 2005). Under the terms of the Convention for the Protection of the Marine Environment of the North-East Atlantic, the UK has supplied the OSPAR Commission annual mass loads of orthophosphate and total phosphorus to its coastal waters since 1990 (Littlewood et al. 1998, Littlewood and Marsh 2005). These loads are calculated as the sum of the mass loads transported by rivers to their tidal limits and the direct discharges to estuaries and costal waters.

Between 1975 and 1994, the North Sea received 54%, the English Channel received 6%, the Celtic Sea received 18%, the Irish Sea received 18% and the Atlantic Ocean received 4% of the total riverine orthophosphate load to UK coastal waters (Littlewood et al. 1998; Watts and Littlewood 1998). In 1994 this approximated 15 to 20 kt orthophosphate y^{-1} (Littlewood et al. 1998; Watts and Littlewood 1998) and slightly exceeded this in 2001 (Littlewood and Marsh 2005). Inclusion of point sources discharging directly to these waters almost doubles these values for orthophosphate loads. Littlewood et al. (1998) estimated that riverine TP loads to the North Sea in 1993 and 1994 were 50 to 70 kt TP y^{-1}, although values submitted to the OSPAR Commission were lower (45 kt TP y^{-1} in 1994; Littlewood et al. 1998). The flow-adjusted load of orthophosphate to UK waters (from both riverine sources and direct discharges) was estimated from HMS data to be 33 kt y^{-1} in 1999 (OSPAR Commission 2003) and was thought to represent approximately 90% of the total UK orthophosphate load. The average export coefficient from catchments delivering to North Sea waters was 1.17 kg orthophosphate ha^{-1} y^{-1} over this period (Howarth et al. 1996). The average annual orthophosphate load to the Celtic Sea was estimated to be about 3 kt y^{-1} in 1994 (Littlewood et al. 1998). The average annual orthophosphate load to coastal waters from UK rivers between the Tweed in the Scottish Borders and the Yare in Norfolk between 1974 and 1994 was estimated to be 5.8 kt y^{-1}, with an average delivery of 1.7 kg ha^{-1} y^{-1} from these catchments (Robson and Neal 1997). The average export coefficient for orthophosphate from the Humber catchment was estimated to be 2.6 kg ha^{-1} y^{-1}, that from the Northerly catchments 0.4 kg ha^{-1} y^{-1} and that from the East Anglian catchments 1.1 kg ha^{-1} y^{-1} (Robson and Neal 1997). Littlewood and Marsh (2005) estimated that export coefficients for orthophosphate from the catchments delivering to the Atlantic averaged 1.40 kg ha^{-1} y^{-1}, to the Irish Sea averaged 1.81 kg ha^{-1} y^{-1}, to the Celtic Sea averaged 1.33 kg ha^{-1} y^{-1}, to the English Channel averaged 1.22 kg ha^{-1} y^{-1} and to the North Sea averaged 1.22 kg ha^{-1} y^{-1}.

Nedwell et al. (2002) estimated annual riverine orthophosphate loads to the 93 major estuaries on the UK mainland for 1995 and 1996 from HMS data. They observed that the estuaries of the Severn, Mersey, Clyde, Humber, Thames and Solent had the greatest annual orthophosphate loads, whilst estuaries along the west coast of Wales and northern Scotland had much smaller loads than those along the east coast of England. The average UK export coefficient for orthophosphate in all UK catchments was 1.55 kg ha^{-1} y^{-1}. The greatest export coefficients were for catchments around the Mersey and for all catchments along the south east coast of England from the Tyne to the Solent. Again, including the contributions from STWs discharging directly to coastal waters in these estimates increases the calculated orthophosphate loads and export coefficients substantially (Nedwell et al. 2002). These data are consistent with those presented by Parr et al. (1999), who estimated the P loads from 49 UK catchments and, in a detailed study of P loads to three Welsh estuaries (Glaslyn/Dwyryd, Mawddach and Dyfi), suggested that agriculture contributed between 26 to 56% of the total P load (Table 05.02).

05.03 Apportionment studies of specific UK watersheds

An historical correlation between increased soil P and TP in watercourses provides circumstantial evidence that agriculture contributes significantly to riverine TP loads in the UK (Haygarth 1997; Sharpley et al. 1994, Smith et al. 1995; Haygarth and Jarvis 1999). Similarly, estimates that 26 - 88% of the TP load to UK rivers is associated with PP (Walling 2005; Defra 2004), also suggest a significant contribution by agriculture to riverine loads. These observations are consistent with estimates of the agricultural contribution to TP loads obtained in source apportionment studies of specific rivers and catchments, which range from 7% to 100% (Table 05.02; Mainstone and Parr 2002; Macleod and Haygarth 2003). It is not surprising that diffuse agricultural sources contribute most to agricultural catchments and least to urbanised catchments. However, several recent studies reporting the dilution of TP with increasing water flows and the correlation of TP loads with boron, suggest significant contributions from smaller STWs and/or septic tanks even in rural areas (Muscutt and Withers 1996; Jarvie et al. 2002a, 2005; Neal et al. 2006ab).

Several studies have attempted to identify the sources of P loads in specific catchments (Table 05.02). Estimates of the diffuse (mostly agricultural) contributions vary widely, reflecting contrasting land uses and human population density. For example, May et al. (2001) estimated that about 46% of the TP load to the upper River Cherwell catchment originated from agricultural sources, whilst urban, suburban and rural developments contributed 9% and STW point sources contributed 44%. Mainstone and Parr (2002) reported that fertilizers contributed 22.4%, livestock contributed 21%, STWs contributed 39.9%, septic tanks contributed 2.3%, industry contributed 2.8% and atmospheric P deposition contributed 14% of the P load to the

Hampshire Avon, and that fertilizers contributed 22.0%, livestock contributed 9.5%, STWs contributed 62.6% and atmospheric P deposition contributed 5.5% of the P load to the Warwickshire Avon. Similarly, in their study of the urbanised river Avon catchment in 1994-1996, Bowes et al. (2005) observed that STWs were the dominant TP source at 14 out of 44 monitoring sites and that the contribution of STWs to the TP load ranged from 9 to 93% depending upon the location. Bowes et al. (2005) estimated that STWs contributed 76% of the TP load near the outflow of the study catchment (Evesham), which is consistent with the previous estimates of Hilton et al. (2002) that point sources contributed 76%, diffuse arable sources contributed 15% and livestock contributed 8% to the TP loads at this location.

Hanrahan et al. (2001) estimated that, in 1998, diffuse sources (land use, livestock and septic tanks) contributed 65% to the TP load in the rural Frome catchment (Dorset), with STWs contributing the remainder. Similarly, Wood et al. (2005) estimated that diffuse agricultural sources contributed at least 60% of the TP loads to rivers in the Taw catchment (Devon) at all the locations they sampled, and argued that their conclusions are relevant to the 14–24% of land in England and Wales sharing similar soils and/or hydrology. However, Haygarth et al. (2005b) observed that the relative contribution of diffuse sources to the TP loads of the Taw basin was seasonal. Diffuse P sources contributed most during periods of high flows, whilst point sources contributed significantly during periods of low flow. Similarly, in a study of the Thame catchment in 1999, Cooper et al. (2002) observed that during periods of low flows most P could be attributed to point sources (largely in the form of SRP), while at high flows their contribution to TP loads was less than 10%. The annual contribution from diffuse sources to the TP load of the Thame catchment was 36-53%, and the annual contribution of diffuse sources to the SRP load approximated 20% (Cooper et al. 2002). Jarvie et al. (2002c) apportioned the SRP loads of the River Kennet at Mildenhall between point and diffuse sources for the period 1998-1999. Again the relative contribution of point and diffuse P sources depended upon the season. They suggested that that diffuse sources contributed between 20% (during periods of low flows) and 90% (during periods of high flows) of the instantaneous SRP load, with an annual contribution of between 29 and 45% of the SRP load.

05.04 Apportionment studies for EU countries

In 1993, it was estimated that about 50% of the P load to surface waters in the EU came from agriculture, with 41% from waste waters (24% from human sources, 10% from detergents, 7% from industry) and 9% from background sources (Morse et al. 1993). Values for the agricultural contribution to the P load of surface waters ranged from 43% for the UK to 73% for Ireland (Table 05.03). These estimates were consistent with other estimates of the period (e.g. Isermann 1990; Kronvang et al. 1996; Brunner and Lampert 1997; de Wit et al. 2002). Revised estimates were recently reported by Herbke et al. (2005) for the European Environment Agency (Table 05.04), which suggest agricultural contributions to TP loads to surface waters of between 10% for Norway (although others estimate a value of 45% for Norway, Bechmann et al. 2005) to 80% for Lithuania. In general, these estimates are similar to those of other recent studies (e.g. Kauppila and Bäck 2001; Andersen et al. 2005; Andersson et al. 2003; Schoumans and Chardon 2003; Kronvang et al. 2005ab) and are consistent with studies of major estuaries, rivers and lakes, which also suggest that agriculture contributes significantly to the P loads of European waters (Behrendt 1999; Macleod and Haygarth 2003; Stälnacke et al. 2001; Pieterse et al. 2003; EEA 2005; Kronvang et al. 2005a).

Table 05.03 Phosphorus sources (%) and loads (kt TP y^{-1}) to surface waters in EC/EFTA countries estimated by Morse et al. (1993).

Country	Human	Detergent	Livestock	Fertiliser	Industry	Background	Load
Belgium	26	11	43	7	8	5	13
Denmark	12	11	55	11	5	6	15
France	18	15	31	19	6	11	106
Germany	28	3	44	12	6	7	97
Greece	21	7	18	34	5	15	17
Ireland	9	7	49	24	2	9	15
Italy	35	2	26	18	8	11	56
Netherlands	23	3	57	9	5	3	24
Portugal	24	14	27	16	7	12	15
Spain	19	16	18	26	7	14	72
UK	24	19	29	14	8	6	82
Austria	20	10	36	16	6	12	13
Finland	18	9	17	15	3	38	9
Norway	23	2	9	12	5	49	7
Sweden	21	10	15	14	7	33	14
Switzerland	28	3	38	15	6	10	8

Table 05.04 Recent estimates of phosphorus loads (kt TP ha^{-1} y^{-1}) to surface waters in Europe (Herbeke et al. 2005). Values for the percentage contribution from agriculture are those given by (A) the European Environment Agency based on the data presented in this table or (B) the Water Research Centre (WRc), based on an independent analysis.

Country	**Area-normalised phosphorus loading (kg TP ha^{-1} y^{-1})**					**Agriculture (%)**		**Source**
	Total Diffuse	**Background losses**	**Agriculture**	**Point sources**	**Sum**			
						A	B	
Austria		0.025	0.161	0.172	0.358	45.0	30	Umweltbundesamt (AT) 2001
Belgium	0.760			1.750	2.510	30.3	na	OSPAR 2003
Denmark		0.077	0.252	0.194	0.523	48.2	27	Bøgestrand 2004
Eire						na	36	WRc 2005
Estonia		0.057	0.215	0.031	0.303	71.0	na	HELCOM 2004
Finland		0.080	0.098	0.018	0.196	50.0	na	Finlands miljöcentral 2005
Germany		0.101	0.480	0.348	0.929	51.7	70	Umweltbundesamt (DE) 2004
Latvia		0.052	0.131	0.043	0.226	58.0	72	HELCOM 2004
Lithuania		0.026	0.152	0.013	0.191	79.6	na	HELCOM 2004
Netherlands	1.130			1.250	2.380	47.5	na	OSPAR 2003
Northern Ireland		0.062	0.831	0.647	1.540	54.0	na	Smith et al. 2004
Norway		0.039	0.026	0.203	0.268	9.7	na	Selvik et al. 2004
Poland		0.010	0.380	0.175	0.565	67.3	na	HELCOM 2004
Sweden		0.080	0.036	0.034	0.150	24.0	25	SLU and SMHI
UK*		0.127	0.551	1.272	1.951	28.3	na	**This report**
England		0.126	0.677	1.802	2.606	26.0	na	**This report**
Wales		0.126	1.117	0.729	1.971	56.6	na	**This report**
Scotland		0.128	0.174	0.488	0.790	22.0	na	**This report**

*Value does not include Northern Ireland

Objective 06. A discussion of the updated P source apportionment for UK waters in the context of the Water Framework Directive (WFD) Article 5 Risk Assessment maps and data

Under the WFD, risk assessments for each RBD were conducted to establish the likely impact of human activity on surface and groundwaters within each RBD. Surface waters within each RBD were classified into rivers, lakes, transitional waters (estuaries) and coastal waters. These waters were also sub-divided based on natural factors, such as altitude, latitude, longitude, geology and size. Significant pressures and impacts on these water bodies were then assessed and the likelihood of them achieving the relevant objectives set out in the WFD determined. Water bodies were classified into the following reporting categories (1a) Water body at significant risk of failing objectives, (1b) Water body probably at significant risk of failing objectives, (2a) Water body probably not at risk of failing objectives, (2b) Water body not at risk of failing objectives. The reports summarising the results from this risk assessment exercise were obtained from Defra (http://www.defra.gov.uk/environment/water/wfd/article5/index.htm), SEPA (www.sepa.org.uk/wfd/) and EHSNI (www.ehsni.gov.uk/environment/waterManage/wfd/wfd.shtml). The proportions of rivers and lakes at risk (Reporting Categories 1a and 1b) from point and diffuse sources of phosphorus were determined from the data presented in these reports (Table 06.01). No transitional or costal waters were deemed to be at risk from either diffuse or point P sources.

The estimated annual TP loads and area-normalised annual TP loads for individual RBDs (Objective 02) were compared to the proportions of rivers and lakes within each RBD classified as being at risk from point and/or diffuse P sources. No correlations were observed between the estimated annual TP loads or the area-normalised annual TP loads for individual RBDs and the proportions of rivers or lakes at risk from diffuse P sources. This might be a consequence of the methodology used in the risk assessment. In attributing risk categories to water bodies, the geology, size, and depth of the water body are considered, which have the potential to augment or mitigate the effects of P loads from diffuse and point P sources. Nevertheless, area-normalised data for the length of rivers ‘at risk’ from point P sources in each RBD did show a good correlation ($r^2 = 0.9$) with area-normalised annual TP loads (Figure 06.02).

The average annual orthophosphate concentration of all rivers in the eight EA regions of England and Wales were also compared to the area-normalised annual TP loads for individual RBDs estimated in this study (Table 06.01). Comparisons were made between the main RBDs and EA regions covering a common area. A positive correlation ($r^2 = 0.32$) was observed between river orthophosphate concentrations and area-normalised TP loads (Figure 06.03). One outlying point for this relationship represents the Midlands EA region, which is dominated by the Severn RBD but has significant influences from other RBDs. An average of all RBDs influencing the Midlands EA region increases the area-normalised TP load from 2.5 kg P ha^{-1} y^{-1} to 3.0 kg P ha^{-1} y^{-1}, further improving the correlation ($r^2 = 0.43$) between river orthophosphate concentrations and area-normalised TP loads.

In conclusion, the updated estimates of area-normalised TP loads for RBDs (Objective 02) are consistent with the distribution of orthophosphate concentrations in UK rivers and correlate with the length of rivers ‘at risk’ from point P sources (but not diffuse P sources) in each RBD. These observations suggest a large contribution from point sources to bioavailable P and risk of eutrophication.

Table 06.01. Proportions of rivers and lakes in each RBD classified as 'at risk' from point and diffuse P sources in the WFD Article 5 risk assessments, compared with the annual total TP loads and area-normalised annual TP loads calculated in Objective 02 (JP Hammond and PJ White, unpublished).

RBD	% River length at risk from		% Lake area at risk from		Total P load	Area-normalised TP load
	Point sources of P	Diffuse sources of P	Point sources of P	Diffuse sources of P	t TP y^{-1}	kg TP ha^{-1} y^{-1}
Anglian	36.4	48.1	79.5	90.9	3,391	1.39
Dee	23.5	40.0	0.0	19.8	388	1.85
Humber	37.5	59.2	29.7	36.9	7,909	3.33
North West	27.5	43.0	44.1	20.2	3,585	3.37
Northern Ireland[1]	nd	nd	nd	nd	2,174	1.54
Northumbria	23.5	54.8	14.8	18.5	1,691	2.12
Scotland[1]	nd	nd	nd	nd	4,983	0.84
Severn	27.0	57.7	36.4	24.2	5,210	2.55
Solway Tweed	0.6	11.4	18.2	3.1	830	0.57
South East	21.3	14.7	33.3	33.3	2,021	2.51
South West	22.2	59.0	23.1	52.9	2,696	1.59
Thames	57.1	23.5	55.2	37.9	6,840	4.41
Western Wales	13.7	51.9	24.0	30.9	2,079	1.87

[1] not characterised in terms of risk from point or diffuse sources of phosphorus

Figure 06.02. Area-normalised data for the length of rivers 'at risk' from point P sources in each River Basin District (RBD) versus the area-normalised annual TP loads for each RBD.

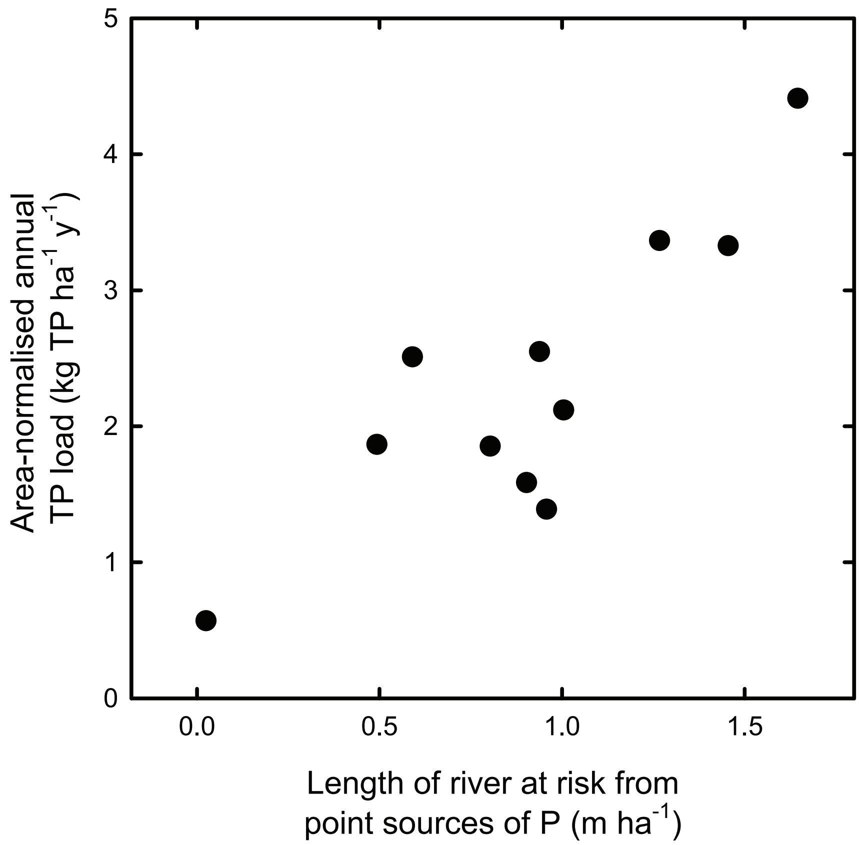
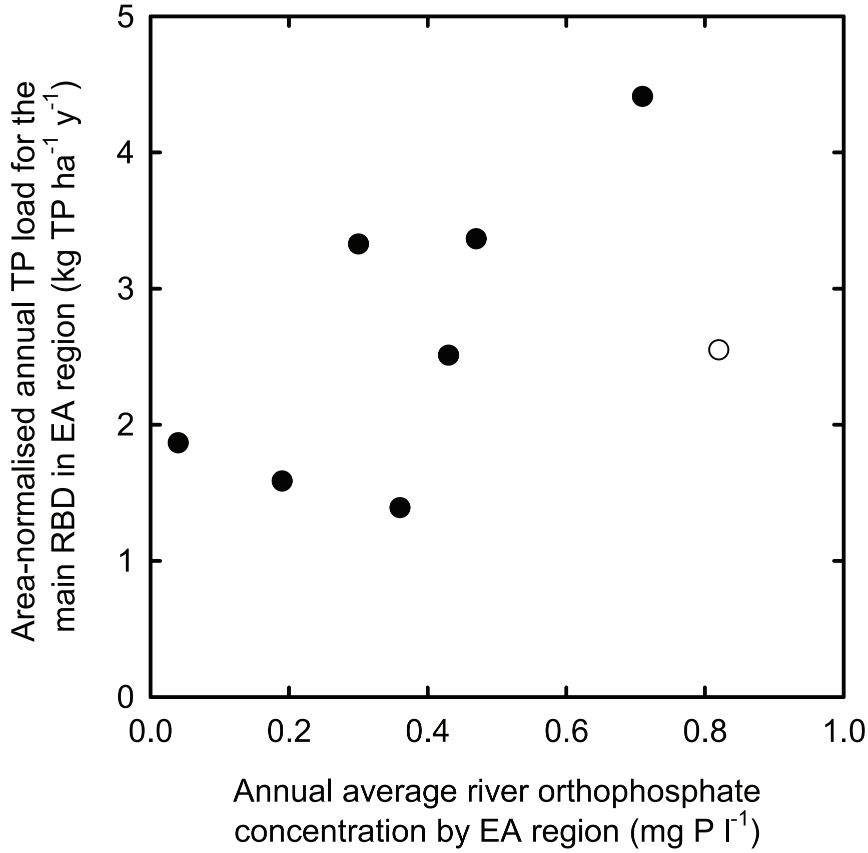


Figure 06.03. Annual average river orthophosphate concentrations for the eight Environment Agency regions versus the area-normalised annual TP load for the main River Basin District covering an EA region calculated in this study. The outlying data point representing the Midlands EA region is denoted by the open symbol.



References

Andersen HE, Kronvang B, Larsen SE (2005) Development, validation and application of Danish empirical phosphorus models. Journal of Hydrology 304, 355-365.

Andersson, Arheimer B, Larsson M, Olsson J, Pers C, Rosberg J, Tonderski K, Ulén B (2003) Integrated modelling of phosphorus fluxes at the catchment scale. Diffuse Pollution Conference Dublin 2003, 1085-1090.

Anon (1992) The UK Environment. London: HMSO.

Bailey-Watts AE, Kirika A (1999) Poor water quality in Loch Leven (Scotland) in 1995 in spite of reduced phosphorus loadings since 1985: the influences of catchment management and inter-annual weather variation. Hydrobiologia 403, 135-151.

Barr Engineering Company (2004) Detailed Assessment of Phosphorus Sources to Minnesota Watersheds. Minnesota Pollution Control Agency (http://www.pca.state.mn.us/hot/legislature/reports/phosphorus-report.html)

Beaulac MN, Reckhow KH (1982) An examination of land use - nutrient export relationships. Water Resources Bulletin 18, 1013-1024.

Bechmann M, Krogstad T, Sharpley A (2005) A phosphorus index for Norway. Acta Agriculturae Scandinavica Section B – Soil and Plant 55: 205-213.

Behrendt H (1999) A comparison of different methods of source apportionment of nutrients to river basins. Water Science and Technology 39, 179-187.

Behrendt H, Opitz D (2000) Retention of nutrients in river systems: dependence on specific runoff and hydraulic load. Hydrobiologia 410, 111-122.

Bennion H, Hilton J, Hughes M, Clark J, Hornby D, Fozzard I, Phillips G, Reynolds C (2005) The use of a GIS-based inventory to provide a national assessment of standing waters at risk from eutrophication in Great Britain. Science of the Total Environment 344, 259-273.

Bennion H, Johnes P, Ferrier R, Phillips G, Haworth E (2005b) A comparison of diatom phosphorus transfer functions and export coefficient models as tools for reconstructing lake nutrient histories. Freshwater Biology 50, 1651–1670.

Bowes MJ, Hilton J, Irons GP, Hornby DD (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.

Brazier RE, Heathwaite AL, Lui S (2005) Scaling issues relating to phosphorus transfer from land to water in agricultural catchments. Journal of Hydrology 304, 330-342.

Brigault S, Ruban V (2000) External phosphorus load estimates and P-budget for the hydroelectric reservoir of Bort-Les-Orgues, France. Water Air and Soil Pollution 119, 91-103.

Brunner PH, Lampert C (1997) The flow of nutrients in the Danube River basin. Sources and final sinks. EAWAG News 43, 15-17.

Burns IG (2004) Defra Project NT2505: Desk study to assess the potential for N and P pollution from glasshouse production in England and Wales. Defra, London.

Carvalho L, 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 5, 191-204.

Carvalho L, Maberly S, May L, Reynolds C, Hughes M, Heathwaite AL, Brazier R, Liu S, Hilton J, Hornby D, Bennion H, Elliott A, Willby N, Dils RM, Pope L, Phillips G, (2003) Technical Report P2-260/9: Risk assessment methodology for determining nutrient impacts in surface freshwater bodies. Environment Agency and Scottish Environment Protection Agency.

Catt JA, Howse KR, Farina R, Brockie D, Todd A, Chambers BJ, Hodgkinson R, Harris GL, Quinton JN (1998) Phosphorus losses from arable land in England. Soil Use and Management 14, 168-174.

Catt JA, Quinton JN, Rickson RJ, Styles P (1994) Nutrient losses and crop yields in the Woburn erosion reference experiment. In: RJ Rickson, editor, Conserving Soil Resources: European Perspectives, CAB International, Wallingford, UK

Cooper DM, House WA, May L, Gannon B (2002) The phosphorus budget of the Thame catchment, Oxfordshire, UK: 1. Mass balance. Science of the Total Environment 282/283, 233-251.

Cooper DM, Watts CD (2002) A comparison of river load estimation techniques: application to dissolved organic carbon. Environmetrics 13, 733–750.

Culley JLB, Bolton EF, Bernyk V (1983) Suspended solids and phosphorus loads from a clay soil: I. Plot studies. Journal of Environmental Quality 12, 493-498.

Daly K, Mills P, Coulter B, McGarrigle M (2002) Modeling phosphorus concentrations in Irish rivers using land use, soil type, and soil phosphorus data. Journal of Environmental Quality 31, 590–599.

Demars BOL, Harper DM, Pitt J-A, Slaughter R (2005) Impact of phosphorus control measures on in-river phosphorus retention associated with point source pollution. Hydrology and Earth System Sciences Discussions 2, 37-72.

Department for Environment, Food and Rural Affairs [Defra] (2004) Mapping the problem. Risks of diffuse water pollution from agriculture. Defra, London.

Dillon PJ, Kirchner WB (1975) The effects of geology and land use on the export of phosphorus from watersheds. Water Research 9, 135-148.

Dils RM, Heathwaite AL (1996) Phosphorus fractionation in hillslope hydrological pathways contributing to agricultural runoff. In: MG Anderson, SM Brooks, editors, Advances in Hillslope Processes, Vol. 1, John Wiley and Sons Ltd, Chichester, UK, pp 229–252.

Domburg P, Edwards AC, Sinclair AH, Wright GG, Ferrier RC (1998) Changes in fertilizer and manurial practices during 1960-1990: implications for N and P inputs to the Ythan catchment, N.E. Scotland. Nutrient Cycling and Agroecosystems 52, 19-29.

Dudley-Stamp L (1941) The Land of Great Britain, Vols I-X. HMSO, London.

Duncan W, Guthrie R, Owen R (2006) The development of soluble reactive phosphorus regulatory values in UK rivers. http://wfduk.org/LibraryPublicDocs/.

Environment Agency (1998) Aquatic eutrophication in England and Wales: a proposed management strategy. Environment Agency, Bristol.

Environment Agency (2000) Aquatic eutrophication in England and Wales: a management strategy. National Centre for Ecotoxicology and Hazardous Substances, Wallingford. (http://www.environment-agency.gov.uk)

Environment Agency (2004) Pressures from diffuse sources of phosphorus. Environment Agency, London. (http://www.environment-agency.gov.uk)

European Environment Agency [EEA] (2005) Source apportionment of nitrogen and phosphorus inputs into the aquatic environment. Office for Official Publications of the European Communities, Luxembourg. ISBN 92-9167-777-9

Ferrier RC, Edwards AC, Hirst D, Littlewood IG, Watts CD, Morris R (2001) Water quality of Scottish rivers: spatial and temporal trends. Science of the Total Environment 265, 327–342.

Ferrier R, Malcolm A, McAllister E, Morrice J (1996) Report No. SR95 (06)F: Hindcasting of in-loch phosphorus concentrations based on land cover classification. Report for the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER), MLURI, Aberdeen.

Foster IDL, Chapman AS, Hodgkinson RM, Jones AR, Lees JA, Turner SE, Scott M (2003) Changing suspended sediment and particulate phosphorus loads and pathways in underdrained lowland agricultural catchments; Herefordshire and Worcestershire, U.K. Hydrobiologia 494, 119-126.

Foy RH, Bailey-Watts AE (1998) Observations on the spatial and temporal variation in the phosphorus status of lakes in the British Isles. Soil Use and Management 14, 131-138.

Foy RH, Lennox SD (2000) Contributions of diffuse and point sources to the phosphorus loads in the River Main over a 22-year period. Boreal Environment Research 5, 27–37.

Frink CR (1991) Estimating nutrient exports to estuaries. Journal of Environmental Quality 20, 717-724.

Fuller RM, Smith GM, Sanderson JM, Hill RA, Thomas AG (2002) The UK Land Cover Map 2000: construction of a parcel-based vector map from satellites images. The Cartographic Journal 39, 15-25.

Gibson CE, Wu Y, Pinkerton D (1995) Substance budgets of an upland catchment: the significance of atmospheric phosphorus inputs. Freshwater Biology 33, 385-392.

Gold A, Sims J (2006) Fate of Phosphorus in Septic Tanks (Autonomous Waste Water Treatment Systems). SCOPE Newsletter 63, January 2006.

Grimvall A, Stålnacke P (1996) Statistical methods for source apportionment of riverine loads of pollutants. Environmetrics 7, 201-213.

Grizzetti, B, Bouraoui, F, Granlund, K, Rekolainen, S, Bidoglio, G (2003) Modelling diffuse emission and retention of nutrients in the Vantaanjoki watershed (Finland) using the SWAT model. Ecological Modelling 169, 25-38.

Gruszowski KE, Foster IDL, Lees JA, Charlesworth SM (2003) Sediment sources and transport pathways in a rural catchment, Herefordshire, UK. Hydrological Processes 17, 2665-2681.

Hanrahan G, Gledhill M, House WA, Worsfold PJ (2001) Phosphorus loading in the Frome catchment, UK: Seasonal refinement of the coefficient modeling approach. Journal of Environmental Quality 30, 1738-1746.

Hanrahan G, Gledhill M, House WA, Worsfold PJ (2003) Evaluation of phosphorus concentrations in relation to annual and seasonal physico-chemical water quality parameters in a UK chalk stream. Water Research 37, 3579–3589.

Harris G, Heathwaite AL (2005) Inadmissible evidence: knowledge and prediction in land and riverscapes. Journal of Hydrology 304, 3-19.

Haygarth P (1997) Agriculture as a source of phosphorus transfer to water: sources and pathways. SCOPE Newsletter 21, 1-15.

Haygarth PM (2005) Linking landscape sources of phosphorus and sediment to ecological impacts in surface waters. Science of the Total Environment, 344, 1-3.

Haygarth PM, Chapman PJ, Jarvis SC, Smith RV (1998) Phosphorus budgets for two contrasting farming systems in the UK. Soil Use and Management 14, 160-167.

Haygarth PM, Condron LM, Heathwaite AL, Turner BL, Harris GP (2005a) The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. Science of the Total Environment 344, 5-14.

Haygarth PM, Jarvis SC (1997) Soil derived phosphorus in surface runoff from grazed grassland lysimeters. Water Research 31, 140-148.

Haygarth PM, Jarvis SC (1999) Transfer of phosphorus from agricultural soils. Advances in Agronomy 66, 195-249.

Haygarth P, Johnes P, Butterfield D, Foy B, Withers P (2003) Amendment to Project PE0203: Land use for achieving 'good ecological status' of water bodies in England and Wales: a theoretical exploration for nitrogen and phosphorus. Defra, London.

Haygarth P, Scholefield D, Chadwick D, Cardenas L, Butler P, Shepherd M, Goodlass G, Withers P, Chambers B, Lord E, Cottrill B, Smith K, Harris D, Ferrier R, Stutter M, Carvalho L, Anderson J, White P, Cuttle S, Dewhurst R, Heathwaite L, Johnes P (2004a) Project ES0201: Developing measures to promote catchment-sensitive farming. A joint Defra-HM Treasury Consultation. http://www.defra.gov.uk/environment/water/dwpa/index.htm

Haygarth P, Turner BL, Fraser A, Jarvis S, Harrod T, Nash D, Haliwell D, Page T, Beven K (2004b) Temporal variability in phosphorus transfers: classifying concentration - discharge event dynamics. Hydrology and Earth System Sciences 8, 88-97.

Haygarth PM, Woods TFL, Heathwaite AL, Butler PJ (2005b) Phosphorus dynamics observed through increasing scales in a nested headwater-to-river channel study. Science of the Total Environment 344, 83– 106.

Heathwaite AL (2003) Making process-based knowledge useable at the operational level: a framework for modelling diffuse pollution from agricultural land. Environmental Modelling and Software 18, 753-760.

Heathwaite AL, Dils RM, Liu S, Carvalho L, Brazier RE, Pope L, Hughes M, Phillips G, May L (2005) A tiered risk-based approach for predicting diffuse and point source phosphorus losses in agricultural areas. Science of the Total Environment 344, 225-239.

Heathwaite AL, Fraser AI, Johnes PJ, Hutchins M, Lord E, Butterfield D (2003a) The Phosphorus Indicators Tool: a simple model of diffuse P loss from agricultural land to water. Soil Use Management 19, 1-11.

Heathwaite L, Sharpley A, Bechmann M (2003b) The conceptual basis for a decision support framework to assess the risk of phosphorus loss at the field scale across Europe. Journal of Plant Nutrition and Soil Science 166, 447-458.

Heckrath G, Brookes PC, Poulton PR, Goulding KWT (1995) Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. Journal of Environmental Quality 24, 904-910.

Herbke N, Dworak T, Karaczun ZM (2005) WFD and Agriculture – Analysis of the pressures and impacts. European Environmental Agency.

Hilton J, Buckland P, Irons GP (2002) An assessment of a simple method for estimating the relative contributions of point and diffuse source phosphorus to in-river phosphorus loads. Hydrobiologia 472, 77-83.

House WA, Leach D, Warwick MS, Whitton BA, Pattinson SN, Ryland G, Pinder A, Ingram J, Lishman JP, Smith SM, Rigg E, Denison FH (1997) Nutrient transport in the Humber rivers. Science of the Total Environment 194/195, 303-320.

Howarth RW, Billen G, Swaney D, Townsend A, Jaworski N, Lajtha K, Downing JA, Elmgren R, Caraco N, Jordan T, Berendse F, Freney J, Kudeyarov V, Murdoch P, Zhu ZL (1996) Regional nitrogen budgets and riverine N and P fluxes for the drainage to the North Atlantic Ocean: natural and human influences. Biogeochemistry 35, 75–139.

Ierodiaconou D, Laurenson L, Leblanc M, Stagnitti, F, Duff G, Salzman S, Versace V (2005) The consequences of land use change on nutrient exports: a regional scale assessment in south-west Victoria, Australia. Journal of Environmental Management 74, 305-316.

Irvine K., Allott N., de Eyto E., Free G., White J., Caroni R., Kennelly C., Keaney J., Lennon C., Kemp A., Barry E., Day S., Mills P., O'Ríain G., Quirke B., Twomey H, Sweeney P (2001) Environmental Research R&D Report Series No. 12. Ecological assessment of Irish lakes. Environmental Protection Agency, Ireland.

Irvine K, Donohue I, Wemaere A, Styles D, Hobbs W (2003) Connecting land-use with water quality: Sources, sinks and timebombs. Diffuse Pollution Conference, Dublin 2003

Isermann K (1990) Share of agriculture in nitrogen and phosphorus emissions into the surface waters of Western Europe against their background of eutrophication. Fertiliser Research 26, 253-269.

James WF, Barko JW (2005) Biologically labile and refractory phosphorus loads from the agriculturally-managed Upper Eau Galle River Watershed, Wisconsin. Lake and Reservoir Management 21, 165-173.

Jarvie HP, Lycett E, Neal C, Love A (2002a) Patterns in nutrient concentrations and biological quality indices across the upper Thames river basin, UK. Science of the Total Environment 282/283, 263-294.

Jarvie HP, Neal C, Warwick A, White J, Neal M, Wickham HD, Hill LK, Andrews MC (2002b) Phosphorus uptake into algal biofilms in a lowland chalk river. Science of the Total Environment 282/283, 353-373.

Jarvie HP, Neal C, Williams RJ, Neal M, Wickham HD, Hill LK, Wade AJ, Warwick A, White J (2002c) Phosphorus sources, speciation and dynamics in the lowland eutrophic River Kennet, UK. Science of the Total Environment 282/283, 175-203.

Jarvie HP, Neal C, Williams RJ (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– 55.

Jarvie HP, Neal C, Withers PJA (2005) Sewage-effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus? Science of the Total Environment doi:10.1016/j.scitotenv.2005.08.038

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

Johnes P, Moss B, Phillips G (1994) NRA Project 286: Lakes - Classification and monitoring. A strategy for the classification of lakes. National Rivers Authority, Bristol.

Johnes PJ, Heathwaite AL (1997) Modelling the impact of land use change on water quality in agricultural catchments. Hydrological Processes 11, 269-286.

Johnes PJ, Hodgkinson RA (1998) Phosphorus loss from agricultural catchments: pathways and implications for management. Soil Use and Management 14, 175-185.

Johnes P, Moss B, Phillips G (1996) The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: Testing of a model for use in conservation and water quality management. Freshwater Biology 36, 451-473.

Johnston AE, Dawson CJ (2005) Phosphorus in agriculture and in relation to water quality. Agricultural Industries Federation, Peterborough.

Jordan C, McGuckin SO, Smith, RV (2000) Increased predicted losses of phosphorus to surface waters from soils with high Olsen-P concentrations. Soil Use and Management 16, 27-35.

Kampas A, Edwards AC, Ferrier RC (2002) Joint pollution control at a catchment scale: compliance costs and policy implications. Journal of Environmental Management 66, 281-291.

Kauppila P, Bäck S (2001) The state of Finnish coastal waters in the 1990s. The Finnish environment. Helsinki: The Finnish Environment Institute, 2001.

Kronvang B, Bechmann M, Lundekvam H, Behrendt H, Rubæk GH, Schoumans OF, Syversen N, Andersen HE, Hoffmann CC (2005a) Phosphorus losses from agricultural areas in river basins: effects and uncertainties of targeted mitigation measures. Journal of Environmental Quality 34, 2129-2144.

Kronvang B, Bechmann M, Pedersen ML, Flynn N (2003) Phosphorus dynamics and export in streams draining micro-catchments: Development of empirical models. Journal of Plant Nutrition and Soil Science 166, 469–474.

Kronvang BB, Jeppesen E, Conley DJ, Søndergaard M, Larsen SE, Ovesen NB, Carstensen J (2005b) Nutrient pressures and ecological responses to nutrient loading reductions in Danish streams, lakes and coastal waters. Journal of Hydrology 304, 274-288.

Kronvang B, Græsbøll P, Larsen SE, Svendsen LM, Andersen HE (1996) Diffuse nutrient losses in Denmark. Water Science and Technology 33, 81-88.

Lemunyon LE, Gilbert RG (1993) The concept and need for a phosphorus assessment tool. Journal of Production Agriculture 6, 483-496.

Lewis DR, McGechan MB (2002) Review of field scale phosphorus dynamics models. Biosystems Engineering 82, 359-380.

Littlewood IG, Marsh TJ (2005) Annual freshwater river mass loads from Great Britain, 1975–1994: estimation algorithm, database and monitoring network issues. Journal of Hydrology 304, 221–237.

Littlewood IG, Watts CD, Custance JM (1998) Systematic application of United Kingdom river flow and quality databases for estimating annual river mass loads (1975-1994). Science of the Total Environment 210/211, 21–40.

Liu S, Brazier R, Heathwaite L (2005) An investigation into the inputs controlling predictions from a diffuse phosphorus loss model for the UK; the Phosphorus Indicators Tool (PIT). Science of the Total Environment 344, 211-223.

Macleod C, Haygarth P (2003) A review of the significance of non-point source agricultural phosphorus to surface water. SCOPE Newsletter 51, 1-10.

Mainstone CP, Davis RD, House A, Parr W (1996) A review of methods for assessing and controlling non-point sources of phosphorus. NRA R&D Project Record 562/5/W. NRA, Bristol.

Mainstone CP, Parr W (2002) Phosphorus in rivers - ecology and management. Science of the Total Environment 282/283, 25-47.

Marsden MW, Mackay DW (2001) Water quality in Scotland: the view of the regulator. Science of the Total Environment 265, 369-386.

Marsden MW, Smith MR, Sargent RJ (1997) Trophic status of rivers in the Forth catchment, Scotland. Aquatic Conservation: Marine and Freshwater Ecosystems 7, 211-221.

Marsh TJ, Sanderson FJ (1997) A review of hydrological conditions throughout the period of the LOIS monitoring programme - Considered within the context of the recent UK climatic volatility. Science of the Total Environment 194, 59-69.

Mattikalli NM, Richards KS (1996) Estimation of surface water quality changes in response to land use change: Application of the export coefficient model using remote sensing and geographical information system. Journal of Environmental Management 48, 263-282.

May L, House WA, Bowes M, McEvoy J (2001) Seasonal export of phosphorus from a lowland catchment: Upper River Cherwell in Oxfordshire, England. Science of the Total Environment 269,117-130.

May L, Place CJ, George DG (1996) Report ED/T11059s/1: The effects of soil type on nutrient losses and run-off in the catchment of Bassenthwaite Lake. NRA North West Region.

McDowell RW, Sharpley AN, Condron LM, Haygarth PM, Brookes PC (2001) Processes controlling soil phosphorus release to runoff and implications for agricultural management. Nutrient Cycling in Agroecosystems 59, 269-284.

McFarland AMS, Hauck LM (2001) Determining nutrient export coefficients and source loading uncertainty using in-stream monitoring data. Journal of the American Water Resources Association 37, 223-236.

McGechan MB, Lewis TDR, Hooda PS (2005) Modelling through-soil transport of phosphorus to surface waters from livestock agriculture at the field and catchment scale. Science of the Total Environment 344, 185-199.

McGuckin SO, Jordan C, Smith RV (1999) Deriving phosphorus export coefficients for CORINE land cover types. Water Science and Technology 39, 47-53.

Morse GK, Lester JN, Perry R (1993) The Economic and Environmental Impact of Phosphorus Removal from Wastewater in the European Community. Selper Publications, London. ISBN 0-948411-08-2

Muscutt AD, Withers PJ (1996) The phosphorus content of rivers in England and Wales. Water Research 30, 1258-1268

Neal C, Jarvie HP, Neal M, Love AJ, Hill L, 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. Journal of Hydrology 304, 103-117.

Neal C, Neal M, Hill L, Wickham H (2006a) The water quality of the River Thame in the Thames Basin of south/south-eastern England. Science of the Total Environment, in press.

Neal C, Neal M, Hill L, Wickham H (2006b) River water quality of the River Cherwell: An agricultural clay-dominated catchment in the upper Thames Basin, southeastern England. Science of the Total Environment, in press.

Nedwell DB, Dong LF, Sage A, Underwood GJC (2002) Variations of the nutrients loads to the mainland U.K. estuaries: correlation with catchment areas, urbanization and coastal eutrophication. Estuarine, Coastal and Shelf Science 54, 951–970.

Organisation for Economic Cooperation and Development [OECD] (1982) Eutrophication of waters, monitoring, assessment and control. OECD, Paris.

OSPAR Commission (2003) Nutrients in the convention area. Implementation of PARCOM recommendations 88/2 and 89/4, 13-16.

OSPAR Commission (2004a) HARP-NUT Guideline 7: Quantification and reporting of the monitored riverine load of nitrogen and phosphorus, including flow normalisation procedures. Reference Number: 2004-2-E.

OSPAR Commission (2004b) HARP-NUT Guideline 8: Quantification of nitrogen and phosphorus losses from diffuse sources by riverine load apportionment. Reference Number: 2004-2-E.

Parr W, Wheeler M, Codling I (1999) Report No. CO4704: Nutrient status of the Glaslyn/Dwyryd, Mawddach and Dyfi estuaries - its context and ecological importance. WRc Final Report to the Countryside Council for Wales.

Pierzynski GM, Sims JT, Vance GF (2000) Soils and Environmental Quality. Boca Raton CRC Press.

Pieterse NM, Bleuten W, Jorgensen SE (2003) Contribution of point sources and diffuse sources to nitrogen and phosphorus loads in lowland river tributaries. Journal of Hydrology 271, 213-225.

Raghunathan R, Slawecki T, Fontaine TD, Chen Z, Dilks DW, Bierman Jr VJ, Wade S (2001) Exploring the dynamics and fate of total phosphorus in the Florida Everglades using a calibrated mass balance model. Ecological Modelling 142, 247-259.

Redfield AC, Ketchum BH, Richards FA (1963) The influence of organisms on the composition of sea water. In: N Hill, editor, The Sea, Wiley, New York, USA, pp. 26–77.

Reynolds CS, Davies PS (2001). Sources and bioavailability of phosphorus fractions in fresh-waters: a British perspective. Biological Review 76, 27-64.

Roberts AM, Hudson JA, Roberts G (1989) A comparison of nutrient losses following grassland improvement using two different techniques in an upland area of mid-Wales. Soil Use and Management 5, 174-179.

Robson AJ, Neal C (1997) A summary of regional water quality for eastern UK rivers. Science of the Total Environment 194/195, 15-37.

Ryden JC, Syers JK, Harris RF (1973) Phosphorus in runoff and streams. Advances in Agronomy 25, 1-45.

Salvia-Castellví M, Iffly J F, Vander Borght P, Hoffmann L (2005) Dissolved and particulate nutrient export from rural catchments: A case study from Luxembourg. Science of the Total Environment 344, 51-65.

Schneiderman EM, Pierson DC, Lounsbury DG, Zion MS (2002) Modeling the hydrochemistry of the Cannonsville watershed with Generalized Watershed Loading Functions (GWLF). Journal of the American Water Resources Association 38, 1323-1347.

Schoumans OF, Chardon WJ (2003) Risk assessment methodologies for predicting phosphorus losses. Journal of Plant Nutrition and Soil Science 168, 43-408.

Sharpley AN, Chapra SC, Wedepohl R, Sims JT, Daniel TC, Reddy KR (1994) Managing agricultural phosphorus for the protection of surface waters: Issues and options. Journal of Environmental Quality 23, 437-451.

Sharpley AN, Kleinman PJA, Bryant RB (2002) Modeling phosphorus transport in agricultural watersheds: Processes and possibilities. Journal of Soil and Water Conservation 57, 425-439.

Skinner JA, Lewis KA, Bardon KS, Tucker P, Catt JA, Chambers BJ (1997) An overview of the environmental impact of agriculture in the UK. Journal of Environmental Management 50, 111-128.

Smith KA, Chalmers AG, Chambers BJ, Christie P (1998) Organic manure phosphorus accumulation, mobility and management. Soil Use and Management 14, 154-159.

Smith RV, Jordan C, Annett, JA (2005) A phosphorus budget for Northern Ireland: inputs to inland and coastal water. Journal of Hydrology 304, 193-202.

Soap and Detergent Industry Association [SDIA] (1989) Detergent phosphate and water quality in the UK. Booklet produced by the Soap and Detergent Industry Association, Hayes, Middlesex.

Stälnacke P, Pachel K, Basova S (2001) Identification of uncertainties in assessments of the nutrient loads and sources to Lake Peipsi and recommendations to the Estonian-Russian Transboundary Water Commission. Workshop Report, Pskov, Russia.

Svendsen LM, Kronvang B (1993) Retention of nitrogen and phosphorus in a Danish lowland river system: implications for the export from the watershed. Hydrobiologia 251, 123-135.

Vollenweider RA, Kerekes J (1980) The loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. Progress in Water Technology 12, 5-38.

Wade AJ, Whitehead PG, Butterfield D (2002) The Integrated Catchments model of Phosphorus dynamics (INCA-P), a new approach for multiple source assessment in heterogeneous river systems: model structure and equations. Hydrology and Earth System Sciences 6, 583–606.

Walling DE (2005) Tracing suspended sediment sources in catchments and river systems. Science of the Total Environment 344, 159-184.

Walling DE, Russell MA, Webb BW (2001) Controls on the nutrient content of suspended sediment transported by British rivers. Science of the Total Environment 266, 113-123.

Walling DE, Webb BW (1985) Estimating the discharge of contaminants to coastal waters by rivers: some cautionary comments. Marine Pollution Bulletin 16, 488–492.

Watts CD, Littlewood IG (1998) CORAL: a system for calculating river mass loads from UK databases, In: HS Wheater, C Kirby, editors, Hydrology in a Changing Environment British Hydrological Society International Symposium, Volume 1. University of Exeter, pp. 559–581.

Winter JG, Duthie HC (2000) Export coefficient modeling to assess phosphorus loading in an urban watershed. Journal of the American Water Resources Association 36, 1053-1061.

de Wit M, Behrendt H, Bendoricchio G, Bleuten W, van Gaans P (2002) The contribution of agriculture to nutrient pollution in three European rivers, with reference to the European nitrates directive. European Water Management Online. European Water Association.

Withers PJA, Lord EI (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.

Wood FL, Heathwaite AL, Haygarth PM (2005) Evaluating the contribution of diffuse agricultural phosphorus contributions to river fluxes at different scales in the Taw catchment, Devon, UK. Journal of Hydrology 304, 118-138.

Young WJ, Marston FM, Davis JR (1996) Nutrient exports and land use in Australian catchments. Journal of Environmental Management 47, 165-183.