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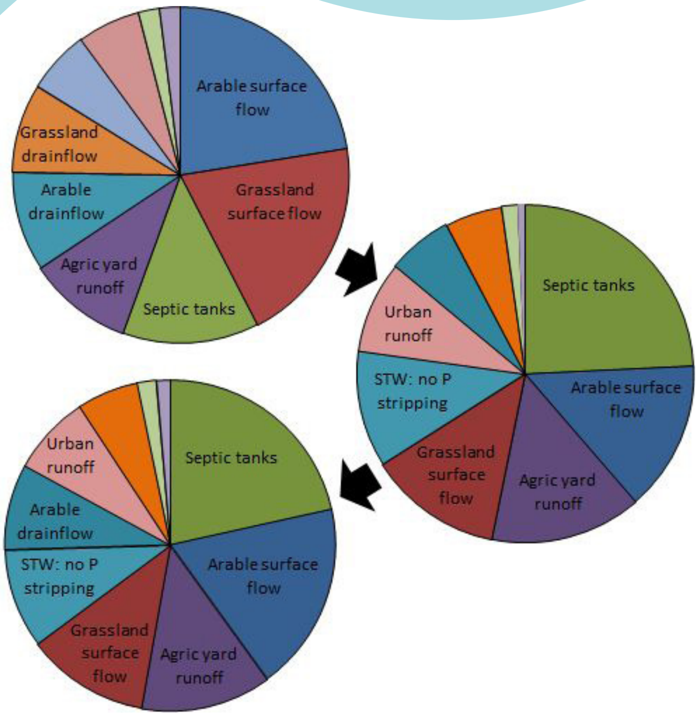
Factoring Ecological Significance of Sources into Phosphorus Source Apportionment



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Contents

Research Summary

Research questions

- How can we develop a scientifically robust methodology for identifying the most ecologically significant sources of phosphorus (P) in a river waterbody catchment?

Key Findings

- Sources vary widely in the proportion of P which is in immediately bioavailable forms (from >80% for septic tanks to <20% for forest and arable crop runoff)
- Septic tanks in particular present a poorly understood source due to the importance and variable design of tanks and soakaway systems for P transfers
- The method developed here to modify results from source apportionment models (e.g. SEPA's Diffuse Pollution Screening Tool) takes each source (e.g. urban runoff, arable drainflow) through a sequential set of tables to account for the source, residence time, and dilution potential to obtain the modified P loads that account for ecological significance of P
- The method can be used alongside catchment source apportionment models to improve understanding of the interactions of source and waterbody characteristics in determining P ecological impacts
- The rules for these new modification procedures should be tested against catchment data in a validation stage
- Further consideration should be given to the integration of these procedures with the core model routines of the existing screening tool to best account for methodological weaknesses
- A worked example of the method (Tarland Catchment): When ecological significance was taken into account, the relative significance of septic tanks rose from 17% to 24% and conversely, the significance of grassland surface flow was reduced. These findings are catchment specific and further work would be required to fully understand the implications of this new methodology on source apportionment.

Background

Phosphorus source apportionment is a key tool in determining priorities for mitigation strategies within the River Basin Management Planning process under the Water Framework Directive (WFD). The methodology for P source apportionment in rivers is potentially subject to errors by assuming that annual total P loads correlate with ecological impact, despite the fact that other factors (e.g. form or timing of the loading) can modify how P affects the system.

Research Undertaken

This project developed a simple methodology for identifying the most ecologically significant sources of P in a catchment. Key sources of P to flowing waters in the UK have been reviewed, together with the bioavailability of the P in these sources and the timing of delivery, to inform the development of a simple rule base for modifying the annual total P estimates produced by SEPA's Diffuse Pollution Screening Tool.

A review section on general approaches to ecological impacts of nutrient loads identified the following factors that need to be accounted for in better linking P loadings with impacts: sources; transport factors; effect factors; bioavailability; timing of delivery; in-stream retention and cycling; loads versus concentrations; and scale.

A further review of septic tank literature suggests most tank effluent contains about 10 mg P l^{-1} when it is discharged and that the load from rural households averages 1.3 kg P $year^{-1}$. The presence (or not) of a soakaway system is an important, but variable, part of the treatment system since soils effectively bind P. However, P plumes from septic tanks tend to travel along downslope gradients by about 1 m per year. As most systems remain in place for 25 years or more, the recommended (UK) safe setback distance of 10 m is probably inadequate. We devised a flow chart methodology for assessing the attenuation of septic tank system impacts for P delivery, as a function of discharge route (direct or leachfield), soil type, Base Flow Index, slope and proximity to water. This can be applied to the Diffuse Pollution Screening Tool estimates of septic tank loads.

We developed an overall scaling methodology for the range of catchment P sources that can be used alongside current catchment source apportionment models to better understand ecological impacts. The essential components of this comprise consideration of (i) source factors of the bioavailability and delivery of P with (ii) receiving river factors of dilution and residence times that together determine ecological impacts. The benefits of this approach are;

- The model links to outputs from existing methodologies like the DPST but could be used as a scaling factor in sequence with other catchment source-transport model outputs as an improved P source-ecological receptor module.
- The model includes a simplistic basis for source chemistry and discharge behaviour with aspects of river condition. Incorporating interactions between river summary factors for residence and internal processing with source water factors of P concentration, bioavailability and source delivery nature is an important concept in ecological impacts and is tackled simplistically to avoid over-parameterisation.
- The principle of bioavailability of P from source waters is incorporated via algal assays taken from the literature and such batch assays encompassing P bioavailability and co-limitation of other macronutrients (labile C, dissolved inorganic N) in the assays.

1.0 Introduction

Phosphorus (P) source apportionment is a key tool in determining priorities for mitigation strategies within the River Basin Management Planning process under the Water Framework Directive. The Scottish Environment Protection Agency (SEPA) carry out pollutant source apportionment on impacted waterbodies to ensure the most appropriate mitigation measures are being promoted in the catchment. This is being carried out as part of the characterisation process for the 14 priority catchments. The accumulated nutrient loading of P is a key determinant in such source derivations due to the perceived linkages between greater P loading, elevated concentrations of P fractions in the waterbodies and ecological damage associated with eutrophication. The impact of P is assessed by actual or potential ecological impacts on diatoms and macrophytes, as assessed by relationships with concentrations of soluble reactive P (SRP, but in the case of the UK TAG the non-filtered reactive P), which is generally considered to be the bioavailable fraction of P in the environment.

It is, however, appreciated that there could be large errors in assuming that annual total P loads will correlate with impact on diatoms or annual SRP concentrations because the susceptibility of a river or stream receiving P inputs is dependent on a number of biogeochemical and physical factors (generally termed 'multiple stressors'). These stressors act alongside the magnitude of the P loading to determine the exposure of the ecosystem to the elevated P concentration (factors such as catchment size, or the relationship of river flow and timing of delivery with dilution and residence time), the bioavailability of the nutrients (factors of P form and co-availability of C and N), or general ecosystem-resilience factors (like water pH, alkalinity, temperature). In particular:

- Agricultural P losses are mostly during the wetter winter months and are primarily particulate bound P. It is likely that their impact on the ecology is much less than the annual total phosphorous (TP) source apportionment would suggest.

- Sewage treatment works (STW) and septic tank P inputs can in cases have smaller loads, but are a more constant level of input, including throughout the warm growing season and at low summer flows. Because settlement is the basic treatment employed, the percentage of SRP will be very high in the discharges from these sources. In the case of properly functioning septic tanks or sewage works, SRP will comprise a large amount of the total load. This P input is likely to have a larger ecological impact than source apportionment based on annual TP load would suggest.

- The majority of septic tanks discharge to soakaway where much of the P discharged will be locked up in the soil and may never reach watercourses. The current modelling does not take this into account. As there are too many septic tanks to evaluate individually, is there a crude correction factor that could be used to estimate total loads from soakaways that will reach watercourses? Is it fair in terms of identifying appropriate current measures to assume all loads will eventually reach waterbodies?

A detailed catchment to in-stream model of interactions of these multiple stressors on the impacts for ecosystem processes would be too complex to develop as an immediate management tool. In fact, there remains considerable scientific uncertainty of the interactions of many of the processes. However, this report develops a 'bridging procedure' between current source apportionment models that simply sum P sources on an annual basis to give a total P loading, and the major principals of importance in addressing ecological significance of the loading behaviour.

This report therefore concentrates on two areas for improvement:

(i) The first area for improvement acts on the P loadings across a range of point and diffuse catchment P sources. Chapter 1 reviews the literature and then chapter 3 presents a revised procedure for integrating P source behaviour with basic factors for receiving waters that combine to give a better understanding of the exposure of river ecology to P sources. This is developed as a database of P source attributes to enable bioavailable P loading as a scaling factor and continues with several interaction tables, based on source and receiving water behaviours, acting to further scale the P loading weightings between the predetermined catchment sources.

(ii) The second area for improvement specifically targets septic tank sewage systems due to the current uncertainty with P load estimations from this source. Chapter 2 reviews the literature for factors important in P delivery, considering both septic tank and drainage field factors. Chapter 4 subsequently develops a simple rule base to consider these factors with a view to the available data sources for developing screening approaches at catchment to national scales.

Whilst the approaches in this report are based on current literature and expert judgement they are transparent and fully explained here. However validation of these approaches in the next phase of the work is very important. The report concludes on the limitations of these approaches and makes suggestion for how such procedures may be best developed to provide management tools for decision making on source priorities in river basin management planning. It is suggested that (i) the rules (and hence constants/scaling factors) for the sets of procedures should now be tested against catchment data in a validation stage, and that (ii) further consideration be given to the integration of these procedures with the core model routines of the existing screening tool to best account for methodological weaknesses, or possible double accounting of factors, carried forwards from that source apportionment modelling stage to the proposed scaling approaches given in this report.

2.0 A review of the characteristics of catchment phosphorus sources

2.1 Summary

Key sources of P to flowing waters in the UK are reviewed, together with the bioavailability of the P in these sources and the timing of delivery. The aim is to inform the development of a simple rule base for modifying the annual total P estimates produced by SEPA's Diffuse Pollution Screening Tool (DPST), to produce more ecologically-relevant rankings of the various sources.

Framework for modifying annual loads: The impact of a source on the ecological status of freshwaters is proportional to the effective inputs from that source, i.e. the quantity of the source that has a direct ecological impact. Effective inputs can be calculated by multiplying annual total P (TP) loads by coefficients that reflect the following factors:

Transport factor: Proportion of annual TP transported from source to waterbody
Effect factor: Proportion of transported TP that causes an ecological effect; this might include:
Bioavailability factor: Proportion of TP that is bioavailable
Timing factor: Fraction of the annual nutrient load (TP) delivered during the growing period. For sources whose delivery is linked to rainfall events, working out this factor might involve working

out the proportion of annual rainfall or intense rainfall events that occur during the productive period.
Areal factor: Proportion of the waterbody being impacted by the source
Dilution factor: Potential dilution of the TP load within a given reach/catchment

Sources: A further improvement to the DPST might be to separate out 'intermediate' sources, such as piped farmyard/dairy effluent, which is discharged all year round, from other agricultural contributions that are affected by seasonality. There is also a need to consider any sources (e.g. bank erosion) that are not represented.

Bioavailability: Soluble forms of P (especially inorganic orthophosphate) are generally more bioavailable than particulate forms. However, a full appraisal of bioavailability of all components of P is necessary to compare sources and understand the potential modifying influences associated with some characteristics of river systems. For example, the potential residence times of P-laden particles delivered to river systems will vary according to their different rates of flow and hydraulic retention times. A rigorous assessment of TP bioavailability involves bioassays. This can be applied to samples from a number of P sources to compare their P bioavailability. Techniques for determining bioavailability differ between studies, making results from existing studies difficult to compare. The work of Ekholm and Krogerus (2003), in contrast, provides a useful comparison of total P bioavailability for a variety of waste and other water samples (Table 3); this could form the basis of an estimate of bioavailability factors.

Timing of delivery: Different sources deliver P at different times of the year, largely depending on whether they are runoff-controlled (Table 1). For P sources with a strong link to runoff, annual loads can be modified to produce seasonal 'effective' inputs (Section 2) by looking at the fraction of runoff, or of large storms that takes place during the season of interest.

In-stream retention and cycling: Where P loss occurs predominantly in the winter months, and P is primarily bound to particulate matter, there will be little negative ecological impact unless this P is retained until the subsequent growing season. Subsequent re-suspension and desorption involve complex, dynamic processes. At present, studies suggest that river sediments act as a sink for P, even during the growing season. However, there may be a substantial difference in speed of equilibration response to changes in water column P concentrations between heavily impacted river systems, where the sediments are saturated with P and more in equilibrium with water column SRP, and less impacted and low P saturated river systems where the balance between P sorption/desorption is more dynamic.

Loads *versus* concentrations: Although source apportionment generally considers total P loadings (at, for example, annual time periods), concentration may provide an easier route to determine P exposure, especially when considering vulnerable times of the year. So, annual TP loads may need modifying to take into account the dilution capacity of the system, especially during the summer ecologically sensitive period. A factor describing dilution capacity could be established by looking at seasonal rainfall, baseflow inputs and catchment area. This might be particularly relevant when comparing catchments.

Issues of scale: The scale of interest may alter source ranking. This could be taken into account by multiplying annual TP loads by a factor describing the fraction of the catchment of interest being impacted by the source in question.

2.2 Introduction

Throughout Europe, requirements to protect the ecological status of surface waters have resulted in increasing regulatory controls on P sources entering rivers. The main sources of P comprise effluent (both sewage and industrial) and runoff contaminated by agricultural activities. Different sources result in P inputs entering surface waters in different forms, at different times of year, and at variable locations within the watershed. The dynamics of P transport and delivery range from highly episodic, hydrological event-driven 'diffuse' losses to almost continuous 'point' source contributions (Haygarth et al., 2005; Neal et al., 2005). Accompanying this is a shift in bioavailability, from the relatively unavailable particulate-associated P from diffuse sources, to soluble forms of P from point sources.

Here, we review the key sources of P to flowing waters in the UK, the likely bioavailability of the P in these sources and the timing of delivery. The aim is to inform the development of a simple rule base for modifying the annual TP estimates produced by SEPA's DPST with the goal of producing more ecologically relevant rankings of P from the various sources. However, the reliability of the P loading estimates from this earlier modelling stage must also be considered.

Whilst some freshwaters are P-limited, it is important to realise that P-based nutrient mitigation has, in many cases, not yet yielded the desired improvements in water quality or reductions of nuisance algal growth in rivers that was expected even after two or more decades of reduced P inputs. Legacies of past land management, decoupling of algal growth responses from river P loading in eutrophically impaired rivers, and non-linear recovery trajectories all play a role in delaying ecological response (Jarvie et al., 2013).

2.3 A framework for modifying annual loads

The impact of a source on the ecological status of freshwaters is strongly influenced by the effective inputs from that source, which combine with other waterbody 'sensitivity' factors. This effective input concept has been expressed by authors using ideas based on life cycle analysis, and its adaptation to deal with eutrophication (Gallego et al., 2010; Huijbregts and Seppälä, 2001; Seppälä et al., 2004; Struijs et al., 2011). Other authors have suggested that the effective input, or 'rating', represents the quantity of a source that directly impacts aquatic ecology. Spatial, temporal and other aspects of the source can be taken into account by modifying the rating:

$$I_{effective} = T \times E \times (annual\ TP\ load)$$

[Equation 1]

Where $I_{effective}$ is the effective input, T is a transport factor (range 0-1) and E is an effect factor (range 0-1); adapted from Seppälä et al. (2004). These factors can be determined from models, data or expert judgement, and their exact definition depends on the problem. For the current task, sensible definitions might be:

Transport factor: Proportion of annual TP load transported from source to waterbody
Effect factor: Proportion of transported TP load that causes ecological damage

The effect factor can itself be made up of several factors, for example taking bioavailability and timing of delivery into account:

$$E = B \times \frac{TP_{Productive\ period}}{TP_{Total}}$$

[Equation 2]

The bioavailability factor (B) is the proportion of TP that is bioavailable (range 0-1). The timing of delivery in the above example follows Seppälä et al. (2004), as the fraction of the annual nutrient load (TP) delivered during the growing period (productive period/total). When using a coefficients-based approach, Seppälä et al. (2004) highlight the need for uncertainty/sensitivity analyses and scenario analysis to test model robustness, given that the outputs cannot be tested empirically.

2.3.1 Overview of sources

The DPST used for pollutant source apportionment includes the following sources:

Urban runoff	Agricultural runoff & drainflow
Septic tanks	Agricultural seepage to groundwater
Point sewage treatment works (STWs)	Forest runoff & drainflow
Road runoff	Forest seepage to groundwater

Different sources respond very differently to hydrological events, as summarised in Table 1 (adapted from Edwards and Withers, 2007; Edwards and Withers, 2008). Whilst some overlap is inevitable, where mobilisation and transport rely upon storm events, actual delivery of P is generally highly episodic (Defew et al., 2013), with a seasonal pattern that matches the frequency of precipitation events. In contrast, sewage treatment and industrial outflows are much more continuous.

The single 'agricultural runoff and drainflow' source in the DPST is a composite of a number of sources, in particular diffuse sources (surface runoff) and 'intermediate' sources (field drains; road/track runoff; piped farmyard/dairy effluent). Table 1 summarises how these sources might be expected to differ in their response to rainfall, and likely differences in their P concentration and speciation. CSO – combined sewer overflows; SS – suspended sediments.

The 'intermediate' sources (Edwards and Withers, 2008) have been shown to be important components of the total agricultural load. During a survey of 14 hard-standings in England (Defra, 2002), over 80% of dairy and pig farm hard-standings were found to contribute runoff. Much smaller contributing areas were found for beef and sheep farms (24% and 63%, respectively). Dairy farms, in particular, may discharge fresh effluent daily throughout the year, leading to potentially continuous inputs to water bodies, with concentrations of nutrients ranging from those characteristic of rainfall to those more typical of concentrated slurry. Dunne et al. (2005) for example, found no clear seasonally-related change in composition resulting from either dilution or changes in management practices. For a 2000m^2 area of hardstanding and average daily run-off flow data, they found a SRP load in farmyard dirty water of 1 kg SRP/cow/yr, i.e. equivalent to the diffuse loss from ~50 ha of land.

Table 1: Characteristics of phosphorus sources. Adapted from Edwards & Withers (2007, 2008).

Source	Hydrological		Chemical composition	
Type/source	**Discharge**	**Rainfall dependency**	**Concentration**	**Speciation**
Point				
STW/industry	Continuous	Low	Concentrated	Soluble
CSOs	Episodic	High	Concentrated	Soluble
Intermediate				
Septic tanks	Semi-continuous	Low	Variable	Soluble
Field drains	Semi-continuous	Low-high	Variable	Variable
Road/track runoff	Episodic	High	Variable (high SS)	Variable
Piped farmyard/dairy	Episodic to semi-continuous	Low-high	Variable	Variable
Diffuse				
Surface runoff	Episodic	High	Variable (high SS)	Particulate
Subsurface runoff	Episodic	High	Dilute	Soluble
Groundwater	Continuous	Low	Dilute	Soluble

Hively et al. (2005) attempted to compare the concentrations and physical forms of P from different features around a farm, as well as the likely contribution each would make during summer storm events. Sites were sampled after simulated rainfall conditions during the summer. Of the nine sites evaluated, the four non-field locations were the quickest to produce overland flow under dry summer conditions. Of these, barnyard hard standings and cow paths also exhibited the highest concentrations of total and dissolved P (Table 2), indicating that they are critical source areas in terms of P loading to water.

A further improvement to the DPST might therefore be to separate out 'intermediate' sources, in particular piped farmyard/dairy effluent, from other agricultural contributions.

2.3.2 Bioavailability of total phosphorus

Using TP concentrations to estimate eutrophication risk is problematic for management purposes, as only some forms of P are biologically available for uptake by autotrophic and heterotrophic systems. A more realistic assessment of eutrophication risk is gained by looking at bioavailable P loads or concentrations. The term 'bioavailability' is most commonly defined as some fraction of the TP present within an aquatic system that is considered rapidly available for biological uptake. The most common and general indications of bioavailability relate to the physical form of P, e.g. the proportion of particulate and soluble forms. However, in reality, the 'non-available' fractions may be partially bioavailable. For example Stutter et al. (2007) measured bioavailability of soluble particulate P in the River Dee (northeast Scotland) using the widely used FeO strip paper test and found that it ranged from 2% (tributary sites) to 31% (main stem sites) of the dissolved SRP. Ellison and Brett (2006), using algal assays with Pseudokirchneriella subcapita, found 17-26% of particulate P to be bioavailable in water samples from streams draining a variety of land use types, rising to 73% in urban streams during baseflow conditions.

Variability in the bioavailability of particulate P may be a key aspect of sources that differ in nature. For example, erosive areas such as managed forestry on steep ground may have a high proportion of particulate P (PP), but low PP bioavailability, whereas urban sources may have proportionally lower PP but with much greater bioavailability. The relationship between bioavailable PP and ecological impacts must not be underestimated. It should also be noted that the impact may just affect a certain component of the biota. For example, Stutter et al. (2007) showed that the presence of certain river invertebrate species with specific roles of filter feeding was significantly negatively correlated with bioavailable particulate P in the River Dee. Although it is difficult to explain all the multiple factors in such observed relationships the principles are that functional aspects of certain target species are a better indicator of impacts than more general indices.

More rigorous assessments of bioavailable P employ some form of a bioassay, using a variable incubation period under defined conditions. If this is undertaken on samples from a number of different P sources, then the bioavailability of their P can be compared directly. The flux of TP may then be multiplied by this source-dependent availability coefficient. This approach assumes that the availability of TP within a source remains relatively constant, and has been shown to provide an unbiased estimate of bioavailable P in a preliminary application (e.g. Ekholm and Krogerus, 2003). The following steps are needed:

a. Monitor/model the TP flux from each relevant P source
b. Determine bioavailable P for a subset of samples from these sources
c. Normalize the bioavailability (range 0 – 1) to derive coefficients ('B' in Equation 2 above)

Table 2: Concentrations and loads delivered in overland flow from rainfall simulation sites during a 25 minute simulated rainfall event, under dry summer conditions. Data from Hively et al. (2005).

Site	Total P		Total dissolved P		% particulate P	
	Concentration (mg/l)	Load (mg/m2)	Concentration (mg/l)	Load (mg/m2)	Concentration (mg/l)	Load (mg/m2)
Hard standing (heifer yard)	13.2	56.4	11.6	57.4	12	0
Compacted cow path	0.99	14.9	0.18	3.59	82	22
Grass – not yet grazed	0.58	0	0.37	0	36	-
Grass – recently grazed	0.95	0	0.64	0	33	-
Hay (recently cut)	0.68	0	0.43	0	38	-
Pasture	0.25	0	0.12	0	50	-
Spring in maize field	0.62	1.54	0.11	0	82	2
Spring in heifer pasture	0.30	5.13	0.02	0.51	93	9
Forest	0.19	0	0.007	0	94	-

The most robust method for determining these coefficients is to measure P bioavailability of the sources of interest within the region of interest. If this is not possible, then literature values provide the next best estimate. As techniques for determining bioavailability differ between studies, it is difficult to compare published results directly, so the work of Ekholm and Krogerus (2003) provides a useful comparison of TP bioavailability, measured for a variety of waste waters and other water samples (Table 3). Unfortunately, this does not include several DPST sources, but finding comparable bioavailability measurements for all DPST sources is unlikely.

It is important to point out that estimates like those in Table 3 are inevitably simplifications of reality. There may be seasonal patterns in the composition and bioavailability of P, particularly in agricultural runoff. For example, immediately following slurry application, field drains have higher concentrations of soluble P, whilst particulate P may represent up to 80% of total P during the autumn/winter period, especially after ploughing (Schelde et al., 2006). The dynamics of sediment/water exchange during transport may further affect P speciation and bioavailability. Finally, P is only one of a number of factors that influence the structure and function of ecological communities. Other nutrients, shade, flow regimes, substrate and other factors may provide the dominant control on ecological community and function, rather than bioavailable P loading, in many areas. These other factors become increasingly important when P becomes more moderately to weakly limiting, as is likely to be the case in many nutrient impacted rivers.

Table 3: Aerobic total P bioavailability measured for a variety of waste waters and other water samples using the same technique. Adapted from Ekholm and Krogerus (2003).

Rank	P source	Algal-available P (% of Tot-P)		Number of samples	Details of source
		Mean ± 95% CI[1]	Min-max		
1	Septic tank outflows	89 ± 6	74-98	10 (4 or 8 hour composites)	4 untreated, 1 sand filtered, 5 sand filtered plus P removal through Al and Fe oxides
2	STW, biol treated	83 ± 11[2]	61-103	10 (1-day composite or instantaneous)	Outflow of plant in St. Petersburg
3	Dairy house	69 ± 32	27-93	5	1 raw, 4 purified samples. Purification: sequenced batch reactor technique involving an activated-sludge system with simultaneous precipitation of P with $FeSO_4$
4	STW, biol & chem treated	36 ± 10	0-67	20 (1-day composites)	Outlet of 5 Finnish treatment plants with activated-sludge system and simultaneous P precipitation with $FeSO_4$. 2 plants also used post-precipitation with Al or Al-Fe
5	Field runoff	31 ± 8	15-50	11	Samples taken during spring/autumn from clayey agricultural fields.
6	Industrial effluent	30 ± 14	4-89	18 (mostly instantaneous)	5 pulp and/or paper mills, 1 viscose-producing plant. 5 of the plants had activated-sludge system, one employing simultaneous P precipitation. One plant used only chemical P precipitation with lime
7	Fish farms	29 ± 14	9-72	10	6 faeces and 4 fodder samples; large rainbow trout
8	Large rivers	20 ± 8[2]	3-45	12	From locations along the Neva River (281,000 km^2; Russia) and one from the Narva River (56,200 km2; Estonia).
9	Agricultural rivers	20 ± 3	12-30	14	From 7 rivers draining catchments (6–1088 km^2) with high proportions of agricultural land (22–43%) on clay/silt soils (small point source inputs).
10	Field soils	19 ± 4	6.8-24	10	9 surface soils, 1 subsoil. From 3 experimental fields. All under crop production; clay the predominant soil fraction
11	Forest runoff	16 ± 8	0-55	19	Samples associated with snow melt from outlet of 4 forested 0.05–0.40 km2 catchments (13-18% peat)
12	Lake settling matter	7.9 ± 3.2	1.6-21	16	Sedimentation traps from Lake Karhijärvi
13	Lake bottom sediments	3.3 ± 1.4	0.1-11	12	Sediment samples (0–3 cm) from shallow lakes with internal P loading problems (Lakes Karhijärvi, Pyhäjärvi and Võrtsjärv).

[1]Confidence Interval. [2]True value higher – suspect growth of indigenous algae in sample chamber.

2.3.3 Timing of delivery

Nutrient requirements of aquatic communities have a strong seasonal pattern, with maximum demand occurring in late spring and summer and minimum demand in winter. As such, the summer concentration of bioavailable P is critical for ecological impact. Different sources deliver P at different times, largely depending on whether they are runoff-controlled or not (Table 1).

Aside from runoff from dairy hard standing (discussed in section 2), agricultural P is largely lost to receiving waters during the winter months. In an intensively-monitored Pennsylvanian catchment, Pionke et al. (2000) found 90% of algal-available P to be derived from surface runoff, with 90% of export occurring during storm flow. As storms occurred more frequently during late winter/spring, they recorded 70% of export during this period, most during 5 of the 7 largest storms per year.

For P sources with a strong link to runoff (Table 1), annual loads can be modified to produce seasonal or monthly 'effective' inputs (see Section 1), by taking seasonal or monthly runoff into account. May et al. (2001), for example, produced estimates of monthly P exports by multiplying annual loads by the monthly relative hydraulic runoff. Firstly, the fractional monthly runoff (r; range 0-1) is calculated:

$$r_j = \frac{Q_j - B_j}{AP_j} \qquad \text{[Equation 3]}$$

Where Q_j is discharge from the catchment (m³/month), B_j is discharge from baseflow (m³/month), A is catchment area (m2) and P_j is total rainfall (m). Subscript *j* is month. For diffuse losses, we can assume that total P loss is proportional to hydraulic runoff. The monthly load (kg) can then be estimated as a function of the annual load ($load_{tot}$; kg):

$$load_j = \frac{r_j R_j \times load_{tot}}{r_{tot} R_{tot}} \qquad \text{[Equation 4]}$$

Where r_j is monthly fractional runoff, R_j is monthly total runoff (m), r_{tot} is annual fractional runoff and R_{tot} is annual rainfall (m); subscript *j* denotes month (adapted from May et al., 2001). This approach could be simplified to produce seasonal, rather than monthly, loadings, for example for the late spring/summer (productive) period versus the autumn/winter (unproductive) period.

For Scotland, average climate data for the 30 year period 1981 – 2010 can be broadly summarised (Table 4). These numbers represent very general class averages and could be more rigorously defined from the raw data. However, they serve to illustrate that there is, in fact, remarkably little difference between rainfall during the spring/summer and autumn/winter periods, whilst on average moderate intensity rainfall events (here classed as days where >10mm of rain fell) occur slightly less frequently during the spring/summer (40% of the total). However, due to the lower temperatures and biological activity in winter, evapotranspiration rates are lower, and winter rainfall often falls on saturated soils, so it contributes more readily to overland and subsurface flow, and ultimately, to stream discharge. If hydrological effects are to be taken into consideration in the DPST in a robust way, a more sophisticated hydrological approach than comparing the timing of rainfall events is probably needed.

Inputs of P from sewage sources occur more evenly throughout the year and may, therefore, be the dominant source of P in the summer months (Defra, 2002). Dorioz et al. (1998) formalised the link between P export and runoff intensity and the shift in source dominance (Table 5).

Table 4: Generalised summary of rainfall amount and days with intense rainfall in Scotland, split broadly by region. Source: http://www.metoffice.gov.uk/climate/uk/averages/ukmapavge.html.

Time period	Rainfall (mm)			Days of rain ≥ 10 mm		
	Northwest	Southwest	East	Northwest	Southwest	East
a. Annual	2000	1250	900	>70	40	25
b. Spring	450	300	200	14	8	4
c. Summer	500	300	250	14	10	6
d. Combined spring + summer	950	600	450	28	18	10
e. (spring + summer)/Total	48%	48%	50%	40%	45%	40%

Table 5: Summary of the main characteristics of the four P export regimes that comprise the typology developed for the study watershed. Adapted from Dorioz et al. (1998). Bold indicates the predominant source.

P export regime	Primary sources of stream flow	P source	P form	TP storage in watershed
Dry period	Baseflow Point sources	Baseflow Point sources	Soluble & bioavailable	TP storage in river and on urban surfaces
Dry period (increased flow)	Baseflow Point sources Urban surface runoff	Baseflow Point sources Urban surface runoff TP stored in river	Largely particulate but highly bioavailable	TP storage on urban surfaces removed.
Transition from dry to wet	Baseflow Point sources Urban surface runoff	Baseflow Point sources Urban surface runoff TP stored in river	Highly bioavailable.	TP storage on urban surfaces removed. River TP removed
Wet period	Baseflow Point sources Urban surface runoff Agricultural runoff	Baseflow Point sources Urban surface runoff TP stored in river Agricultural runoff TP in river banks	Particulate, low bioavailability	None available for removal except for river banks in very high flows

2.3.4 Retention and in-stream processing

As agricultural P is largely bound to particulate matter and lost to receiving waters during the winter months, its ability to affect autotrophs and heterotroph in rivers depends on its retention within the aquatic system until the subsequent growing season. In rivers, retention in sediments is controlled by the same processes that control siltation, with higher retention in slower flowing reaches or on floodplains. Retention times in a particular river reach or water body can, therefore, vary from short (seconds) to much longer timescales (months or years).

For P that is retained within a river reach, the importance of agriculture as a source of eutrophication then depends on the extent to which the bed sediments subsequently supply P to the water column (Jarvie et al., 2006). If sediments do not release P during the biologically active period, then catchment measures aimed at controlling agricultural P loading will not have the desired effect of reducing eutrophication risk, even though such measures may control sediment transfer effectively (Edwards and Withers, 2007).

There are two main mechanisms for P remobilisation from bed sediments in rivers:

1. Sediment re-suspension during high flow events

To determine the amount of sediment that is re-suspended during summer, the proportion of large storms occurring during this period could be taken into account. However, it is very difficult to generalize the amounts that are re-suspended without looking in detail at local river geomorphology and taking hysteresis effects into account (Defew et al., 2008).

Evidence for rapid desorption of P from remobilised sediments comes from high temporal resolution (four-hourly) storm hydrochemistry. Stutter et al. (2008) observed sharp increases in SRP concentrations during storm events, which were so rapid (and close to sediment concentration peaks) that they were likely to be driven by resuspension of bed sediments (and/or delivery from very fast acting riparian sediment stores) and rapid P desorption. However, such resuspension occurs during high flows and provides a mechanism for the longer-term flushing of sediments from that reach (albeit while increasing delivery to downstream systems).

2. Phosphorus desorption from suspended and bed sediment

Desorption/adsorption equilibria control P release from sediment. Timings and amounts of P released are difficult to predict: within-river P cycling is a highly dynamic process involving complex interactions between sediments, aquatic plants and the water column (Ekholm and Lehtoranta, 2012; House, 2003). In batch experiments on moderately impacted Scottish river sediments, Stutter and Lumsdon (2008) found that bed sediments changed from sinks during autumn/winter to sources during summer low flows. Conversely, these authors observed that sediments from P-saturated rivers in England acted more closely in equilibrium with river water SRP concentrations. However, other studies covering broader pollution gradients suggest that, in general, sediments act as sinks. Jarvie et al. (2005) measured the P sorption properties of sediments collected from catchments impacted by agriculture to varying degrees, and found that they remained P sinks throughout the year. Only in areas of low population and extensive agriculture was the overlying water column sufficiently low in P to cause P release from the sediment. Similar P retention through the spring to early autumn has been reported by other authors (Jarvie et al., 2011; May et al., 2001).

These results fit with earlier reach-based studies (e.g. McDaniel et al., 2009; Svendsen et al., 1995) that show that in-stream processes can play an important role in reducing ambient P concentrations, particularly during summer low flows. Stutter et al. (2010) studied a two week enrichment of a headwater river with P caused by the diversion of a sewage treatment plant effluent direct to the river. The P uptake was studied by a number of complimentary abiotic and biological studies and was found to approximate to 50% sediment abiotic uptake and 50% biotic uptake, which was split equally between microbial and autotrophic uptake. A proportion of the uptake was re-released (after the effluent flow had stopped) within a further four week period and it was expected this was strongly sediment P desorption.

In summary, P adsorption/desorption will vary by catchment and throughout the year at any given point. A general assumption that sediments act as a sink for P appears to be broadly valid under conditions of stable flows. However, scouring flow events act to reset sediment systems and can alter this situation depending on the source of the fresh surfaces (e.g. topsoil having recently been fertilised compared with stream bank subsoil).

Traditionally, P loads that are mobilised under high flow conditions have been attributed to non-point agricultural sources. However, it has been long considered (and recently highlighted in the work by Jarvie et al., 2012) that within-river retention of effluent P is also significant, and that it makes an important contribution to river P loads when remobilised under high flows. By not accounting for this, and assuming all remobilised P is from diffuse agricultural sources, agricultural sources may be significantly overestimated whilst wastewater sources are underestimated.

2.3.5 Loads versus concentrations

Short residence times of flowing waters can mean that uptake kinetics and growth rates are influenced more by P concentrations rather than fluxes (Edwards et al., 2000). However, this may be more important for unimpacted rivers where P is very limiting and can readily be affected by biological mechanisms such as the buffering to P concentration variations in the water column provided by P adsorption in biofilms. Setting P threshold values for environmental impact as concentrations rather than loads introduces a further difficulty in translating annual loads into ecologically-relevant concentrations. The potential for dilution of bioavailable P concentrations is crucial in determining their ultimate ecological impact. This could be introduced by employing a factor describing catchment or reach hydrology, for example by looking at seasonal rainfall, baseflow inputs and area of catchment upstream of the point of interest. This might be particularly relevant when comparing catchments. For example, a catchment receiving higher P loads may be less impacted than one receiving lower loads if it receives sufficiently higher rainfall or has large, unpolluted headwaters.

Systems for determination of concentration exposure duration are gaining favour as a means of linking pollutant concentrations to impacts and these may be readily calculated to evaluate exposure during critical ecological periods. Bilotta et al. (2010) argue that, for any given sediment concentration, setting tolerable levels of suspended sediment should also take into account the duration of exposure to provide a realistic assessment of the threat posed to aquatic ecology. However, such systems require a good quality hydrochemical dataset at a high (ideally daily to sub-daily) temporal resolution. Hence, such assessments are better suited to research catchments or those where surrogate electronic in-situ sensor data (i.e. turbidity calibrated to suspended sediment concentrations) are available.

2.3.6 Issue of scale

The scale of interest may alter the ranking of sources. Table 6 summarises the typical attributes of headwaters versus major tributaries and main stems, showing a shift in the hydrological response and general significance of individual source groups. Therefore, Neal et al. (2005) and Pieterse et al. (2003) argue that, while the reduction of point sources may help to minimise the size of nutrient fluxes at the catchment scale, a reduction of diffuse sources may help restore water quality to more of the individual tributaries within a given catchment, i.e. it may be more spatially important. To take this into account, it might be possible to multiply annual TP loads by a factor describing the fraction of the catchment of interest that is being impacted by the source in question. However, the impact of a given P loading will differ between the tributary and main stems sites according to factors of dilution, residence and ecological function.

Table 6: Summary of the differences between headwaters and larger tributaries in terms of their hydrological response, key P sources and scale of remediation. Adapted from Edwards and Withers (2008).

Typical attributes	Headwaters	Major tributaries/mainstem
Receiving water hydrological response	Episodic	Damped
Groundwater contribution to receiving water	Small	High
Retention capacity of receiving water	Low	High
Key point or intermediate sources	Septic, farmyard, track, field drains, roads	Large urban STW and industrial
Key diffuse sources	Surface runoff, Subsurface runoff	Groundwater
Likely impacts	Local and acute	Chronic
Best Management Practices	Individual farm-specific	Integrated catchment level

2.4 Potential issues on derivation of a P source - impacts rule-base

This review has outlined important concepts in the derivation of a rule-base to improve linkages between P sources and their potential impacts, such as (i) P forms/bioavailability, (ii) source connectivity to waterbody, (iii) nature of discharge (i.e. continuous to episodic), and (iv) timing of delivery. By including these considerations, it would be possible improve the current system of source prioritisation for tackling P pollution. Improvements could also be achieved by including indicators of relevance to ecological impacts to modify the simple additive annual loads from different sources that are currently used in models such as the Diffuse Pollution Screening Tool (DPST).

However, it should be recognised that many interactions exist between factors (i) to (iv), above, and the nature of the receiving water body. Example waterbody properties in this respect are dilution capacity for point sources and residence times for sediments. However, other aspects of the nutrient chemistry (C and N) of the source and river habitat conditions (riparian condition, light and temperature, physical modification) are key links between P source behaviour and realised ecological impact. Thus, we should not separate source behaviour and receiving water typology, but combine simple aspects of the two to improve the current system. Combining source and waterbody characteristics is important for:

a) improving the current concepts incorporated into the implementation of the Water Framework Directive in relation to single chemical attributes (e.g. SRP concentration for rivers) and individual biological species;

b) removing risks of falsely assigning low weightings to sources on the basis of loads based on a narrow range of P forms, and not considering other key co-contaminants such as fine sediments and other nutrients; and

c) better explaining disconnections between expected recovery and catchment actions when tackling certain groups of pollution sources.

3. Review of septic tank literature

3.1 Background

On-site sewage treatment systems are widely used to treat domestic wastewater in rural and peri-urban areas of the UK where connection to mains sewerage networks is unavailable, impractical or too costly to implement (Environment Alliance PPG4, 2006; May et al., 2010). The majority of these on-site systems are septic tanks systems (STS), many of which are over 25 years old (e.g. 68% in the Clun catchment - Fildes, 2011).

The exact number of on-site STS within the UK is unclear, but it is has been estimated that 96% of households are connected to a sewage treatment works (Defra, 2002). This suggests that the remaining 4% are served by small private treatment works or STS. It has been estimated that there are about 1.2 million STS in England and Wales (Anthony, pers comm., ADAS UK Ltd.), 400,000 in Scotland and 500,000 in Ireland (OEE-OEA DWWTS Cross Office Team, 2012). Recently introduced registration schemes in many of these areas should be able to provide more accurate estimates of these numbers in the near future. Domestic wastewater contains a wide range of substances that can cause water pollution when discharged into the environment. These include nutrients, pathogens and suspended solids. Of these, P is of particular concern because, if it enters waterbodies that are P limited, it can cause serious eutrophication problems and failure to achieve EC Water Framework Directive (EC 2000) water quality targets.

The sources of P in domestic wastewater were recently reviewed by Defra (2008); these are summarised in Figure 1. In total, these sources amount to a P export from each dwelling of about 0.7 kg per capita^{-1} y^{-1} (Gilmour et al., 2008). Average influent P concentrations to sewage treatment systems from this source vary, but are broadly similar, with values of around 9-15 mg l^{-1} of TP having been reported from Ireland (EPA, 2000) and the UK (Gilmour et al., 2008). A large proportion of this TP is usually in the form of SRP; most of the remainder is in particulate form (Gilmour et al, 2008).

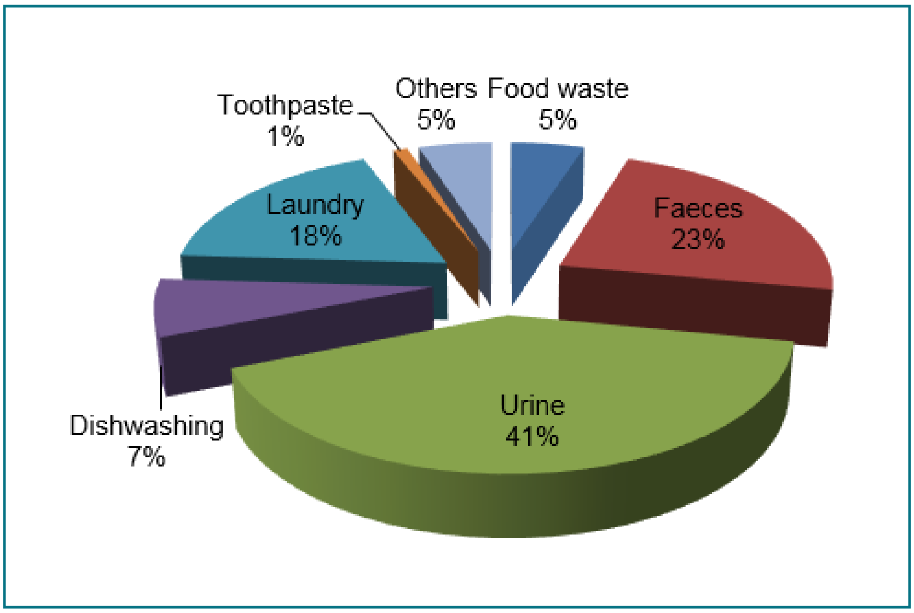


Figure 1. Phosphorus source apportionment in domestic wastewater (redrawn from Defra, 2008)

3.2 The septic tank

Most STS comprise two parts, the main tank and a secondary, soil-based soakway or drainage field. When wastewater enters the tank itself, primary suspended solids are removed by settlement, and oil, grease and fats float to the top. These processes result in a clear effluent that accumulates between these layers (Figure 2). This initial separation process is driven by gravity and by the relative densities of the different types of waste material (Canter & Knox, 1985). The design of the tank affects its performance. For the same volume, a tank with a greater area tends to settle solids more efficiently than one with a smaller area (Seabloom et al., 2005).

The anaerobic conditions that develop inside the tank promote the breakdown of waste material. This reduces the accumulation of solids (sludge) and helps reduce concentrations of some pollutants and pathogens in the resultant effluent (Viraraghavan, 1976; Canter & Knox, 1985). However, these breakdown processes do not remove P from the wastewater very effectively (Lawrence 1973; Bauer et al., 1979; Zanini et al, 1998; Van Cuyk et al, 2001; Beal et al., 2005). Instead, they tend to convert most of the organic P that enters the tank into SRP (Bouma, 1979; Wilhelm et al., 1994; Zanini et al., 1998; Beal et al., 2005), which then passes out of the effluent pipe into the soakway. So, contrary to popular belief, when a septic tank is working effectively, P concentrations in the effluent are very similar to those in the influent wastewater (Gill et al., 2001). It is only the form of P that changes. Effluent TP concentrations recorded in the literature range from 6 to 20 mg l^{-1}, with the average concentration being about 10 mg P l^{1} (Table 7).

Outputs of P from the tank itself can only be reduced by limiting the inputs (e.g. by using phosphate-free detergents) or by precipitating P within the tank (e.g. using additives such as aluminium sulphate (alum) or sodium aluminate – Long & Nesbitt, 1968). Phosphorus can be completely removed from the effluent when aluminium is present in excess (Canter and Knox, 1985).

Using additives such as those mentioned above can have additional benefits, because they also reduce the biochemical oxygen demand (BOD), suspended solids and coliforms in the effluent (Eberhardt & Nesbitt, 1968; Nilsson, 1969; Zenz, 1969). When additives such as alum are added to domestic wastewater it flocs and precipitates P; this precipitate then becomes part of the sludge that is removed by regular cleaning. It has been suggested that dosing of tanks can reduce effluent P concentrations by about 85% (Brandes, 1977).

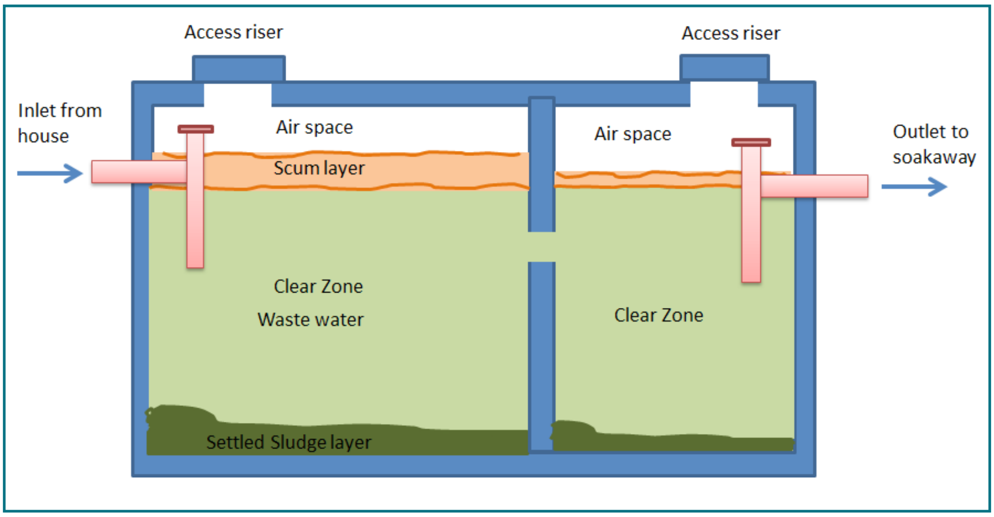


Figure 2. A typical two chamber septic tank where primary solids settle to the bottom of the tank and oil, grease and fat float at the top leaving a clear zone of effluent in between.

TP concentration in septic tank effluent (mg l-1)	Reference
7.0	Wood (1993)
9.7	Robertson et al. (1998)
8.0	Robertson (1995)
6.3	Robertson (2008)
7.5	Robertson (2012)
17	EPA (2011)
8.1	Gross (2005)
9.8	Lowe et al. (2009)
6.7	Patterson et al. (2001)
7	Cogger & Carlile (1984)
12	Canter & Knox (1985)
19.6	Brandes (1977)
Average = 9.9	

Table 7: Average TP concentrations in septic tank effluent

In some cases, a form of secondary treatment is installed to provide additional treatment to tank effluent before it is discharged to the soil soakaway. These are mainly aeration systems, which generate a controlled aerobic environment that accelerates microbial degradation of organic matter. Although not specifically designed to remove P, it has been suggested that this type of secondary treatment may result in a 15% reduction in effluent TP concentrations; this is achieved through bacterial assimilation, precipitation and adsorption (Metcalf & Eddy, 2003; Gill et al., 2009).

Although additives and aeration can reduce the amount of P in tank effluent, the number of septic tanks that have any form of secondary treatment to remove P from the effluent is relatively low (e.g. 14% in the Clun catchment - Fildes, 2011). So, unless site specific detail is available, it can be assumed that most tank effluent contains about 10 mg P l^{-1} when it is discharged.

In terms of the impacts of these discharges on the environment, it is the total load of P from the tank that is important rather than the concentration of P in the effluent (Eveborn et al., 2012). Converting effluent P concentrations to loads requires the rate of flow of effluent from the tank to be calculated. This can be derived from the number of people served by the tank and the average *per capita* water usage. In the case of Scotland, the average size of a household in rural areas is 2.45 people (http://www.scrol.gov.uk/) and the *per capita* water usage is about 150 l d^{-1} (Water UK, 2008), or 54,750 litres per year. So, it can be estimated that the TP load from an average tank serving a single household is about 10 mg P l^{-1} x 150 l d^{-1} x 2.45 = 3.675 g P d^{-1} or 1.341 kg y^{-1}.

3.3 The soil based soakaway or drainage field

Although a small percentage of tanks discharge directly to a waterbody (e.g. 4% in the Clun catchment - Fildes, 2011) and it is has been suggested that these deliver P at a rate of 0.63-0.72 kg $capita^{-1}$ y^{-1} (Pieterse et al., 2003), most deliver their effluent to a soil based soakaway via a percolation bed or trench based system. In the soakaway, further physical, biological and chemical treatment takes place before the effluent finally discharges to surface and/or ground waters (Bouma, 1979; Zanini et al., 1998; Environment Alliance, 2006). Soils are an important part of the treatment process because they form the last line of defence between the tank discharge point and receiving waters (Dawes & Goonetilleke, 2003).

When it leaves the tank, effluent is distributed through a network of perforated pipes within the soil soakaway system. Any suspended solids and organic material within the effluent gradually clogs soil pores as effluent loading rate exceeds its infiltration rate (Gill et al. 2004; Beal et al. 2005). As a result, a saturated zone at the base of the trench system is created that encourages massive growth of bacteria and microorganisms (biomat). Biomat bacteria and microorganisms provide much of septic tank effluent (STE) treatment in the soakaway, such as the decomposition of suspended material and organic matter (Beal et al., 2005; Dudley and May, 2007; Onsite Sewage Treatment Program, 2011; Siegrist, et al. 2012). The biomat is formed in the first few months of STS operation and is crucial in providing an even distribution of wastewater within the soakaway and in prolonging effluent retention time in the soil to maximise effluent treatment.

In a study by Magdoff et al. (1974) investigating P removal from wastewater in soil columns with and without a biomat, the authors observed that P concentrations in the soil column beneath the biomat were reduced to 2–6 mg l^{-1}, compared with 11–14 mg l^{-1} in the soil columns with no biomat. Postma et al. (1992) revealed that seasonal occupancy of dwellings relying on STS may reduce treatment efficiency and promote ground water contamination due to the incomplete formation, and sometimes the absence, of a biomat in drainage trenches (Postma et al, 1992). On the other hand, excessive biomat growth can result in ponding and the subsequent hydraulic failure of the drainage field (Potts et al, 2004).

As STE percolates through the soil soakaway system, P is removed by adsorption, cation exchange and precipitation. The P uptake potential of the soils into which the effluent drains depends on mineralogy and particle size (Jones & Lee, 1979; Zanini et al., 1998). Zanini et al. (1998) suggest that, in general, the ability of soils to immobilise P in septic tank effluents can be ranked as follows:

Fine grained, non-calcareous > Coarse grained non-calcareous > Fine grained calcareous > Coarse grained calcareous

This ranking is supported by the work of Robertson et al. (1998), who found that large plumes of PO_4-P developed at septic rank sites where the soakaway is mainly comprised of coarse grained calcareous sand.

The pH of the soil is also an important factor in determining P uptake capacity. Under acid conditions, phosphate ions are adsorbed onto the surfaces of soil iron (Fe) and aluminium (Al) to form insoluble aluminium or iron phosphate while, under alkaline conditions, phosphate is adsorbed by soil calcium to form calcium phosphate (Canter & Knox, 1986; Gill et al, 2004; Lusk et al, 2011). However, this is not simply a function of the natural pH of the soils. The natural capacity of calcite deficient soils to immobilise P from septic tank sources can be increased substantially by local increases in pH that are caused by effluent oxidation (Zanini et al., 1998). The movement of P through the soil column is minimal until soil sorption sites are occupied; then P movement through the soil is increased (Siegrist and Boyle, 1987).

In addition to the soil characteristics outlined above, local hydrological conditions also affect the ability of a soakaway to retain P. The distance between the effluent distribution system and the highest level of the water table (especially in winter) is particularly important (Canter & Knox, 1985; Environment alliance - PPG4, 2006), because this determines the degree of P uptake in the aerated soil (or vadose) zone. In many countries, planning regulations require a minimum of 1.2 m of undisturbed soil between the base of the percolation trenches and either the bedrock below, or the highest level of the water table (EPA, 2000; Gill et al, 2004). In Scotland, this value is 1m (Scottish Executive, 2001). It is unclear how widely this requirement has been met by existing systems in Scotland. However, in Ireland, it is believed that almost half of existing systems are situated in areas with inadequate soil percolation rates (OEE-OEA DWWTS Cross Office Team, 2012). A survey published by Gill et al. (2007) suggest that, in some areas of Ireland, the problem may be much worse, with almost 95% of STS failing on the basis of soil hydrological characteristics.

3.4 Delivery to waterbodies

Many studies have linked P contamination of surface waters to STS wastes (Withers et al. 2011; Bowes et al. 2010; Edwards and Withers 2008). Efroymson et al. (2007) states that septic tanks that are located within close proximity of watercourses and septic tanks with hydraulic failures will have direct impact on water quality. The contribution of P loading from septic systems that reaches surface waters has been estimated to be about 7% (ESB International, 2008) but, in reality, this figure is very dependent on local conditions.

In a properly maintained and functional STS, effluent P is effectively retained within the soil-based soakaway system and only a relatively low proportion reaches groundwater or surface waters. However, even if more than 90% of the P in the tank effluent is retained in the soakway, the concentration of effluent reaching a nearby waterbody would still be around 1 mg P l^{-1} – approximately the same concentration as the final effluent from a sewage treatment works with tertiary treatment. This is more than 30 times higher than the target P concentration of most receiving waters in rural areas.

The amount of P that is delivered to a nearby waterbody depends on the length of the PO_4-P plume that develops as the discharge percolates through the soil. This can be very short in soils that have a high capacity to immobilise P (Zanini et al., 1998), and much longer in other situations. For example, Robertson et al. (1998) recorded distinct PO_4-P plumes with concentrations of up to 6 mg P l^{-1} more than 10 metres away from domestic STS situated on sandy soils. For larger on site systems, Bussey and Walter (1996) found similar PO_4-P concentrations in discharge plumes that extended several hundred metres towards a groundwater fed pond in Cape Cod, Massachusetts.

Under suitable conditions, the movement of P through the soil is minimal until soil sorption sites have become saturated (Jones & Lee, 1979), after which it increases. For this reason, effluent plumes extend further as a system ages. Efroymson et al. (2007) confirmed that, as P is transported through the soil system, a significant amount of P is adsorbed onto clay soil particles or precipitated after being 'fixed' with soil iron, aluminum and calcium before it reaches surface water. Robertson et al. (1998) indicated that soil-fixed P may not be completely stable and, thus, P adsorbed onto soil surfaces can be re-mobilised (desorbed) over time, enabling it to reach surface and ground water at a later date. The authors recommended that, to increase septic tank P attenuation in soil, much larger setback distances between water courses and septic tank drainage field are required than are common within the UK. This is confirmed by Toor et al. (2011), who found that increasing the distance between the drainage field and a water course can reduce P transport to surface waters.

In general, P plumes from septic tanks tend to extend down gradient by about 1 m per year (Robertson, 2003). As most STS remain in place for more than 25 years (Fildes, 2011; May et al., 2014), this strongly suggests that the recommended safe setback distance from a surface waterbody of 10 m, which is currently in place within the UK (e.g. Scottish Executive, 2001), is probably insufficient to safeguard the quality of that water. However, long term P removal by soil based soakaways is not well understood and most of the data available appear to be unreliable (Eveborn et al., 2012).

In addition to the above, the impact of P delivery from STS on surface waters is also affected by the density of systems within the catchment upstream of any given point (Walsh and Kunapo, 2009). Arnscheidt et al. (2007) explored the potential for STS to pollute rivers under low flow conditions in three small catchments in Ireland and found that the percentage of time that a particular river exceeded a given P concentration increased with the density of STS upstream. Eighty percent of TP concentrations recorded in these rivers draining the Armagh and Monaghan catchments were over 0.2 mg l^{-1}.

One of the problems associated with estimating the P delivery from STS to surface waters is that most studies have focused on measuring P concentrations, rather than loads, and have measured these concentrations in deeper groundwaters used for water supply rather than shallower ground waters that are more likely to discharge into surface waters. There is a risk that the P that is discharged to shallow groundwaters and that immobilised in the soil, will be re-mobilised under wet conditions as the water table rises, especially where such discharges are to floodplain soakaways (Jarvie et al., 2006).

Septic tank systems have a mixed reputation in terms of their reliability (Beal et al., 2005). In many cases they are seen as unpredictable, with variable treatment efficiencies and failure rates. However, much of this is probably due to failure to follow design specifications during installation. For example, Gill et al. (2007) found that only 5.4% of the 74 STS that he examined in Ireland were properly installed in a suitable location; most failed on soil hydrological characteristics. Withers et al. (2011) suggest that STS are an effective method of wastewater treatment if they were designed, sited and maintained properly.

Most previous studies have looked at the concentration of P within the soil profile as the plume develops. However, this reflects not only P uptake/immobilisation processes, but also dilution processes. Eveborn et al. (2012) demonstrated, using a mass balance approach based on load rather than concentration, that P removal performance across STSs is probably much lower than has previously been reported.

4.0 Proposed P source methodology: Ecological Significance of P Sources approach

4.1 Introduction

4.1.1 Summary of factors to consider in an improved approach for ecological P impacts

The preceding review identified the following factors that need to be accounted for in any improved approach to model in-stream ecosystem impacts of P.

- Sources
- Transport factors
- Effect factors
- Bioavailability
- Timing of delivery
- In-stream retention and cycling
- Loads versus concentrations
- Scale

The following chapter describes how these factors are considered and incorporates key aspects into an approach enabling the scaling of existing source apportionment and P load model outputs to take account of within-waterbody exposure to, and uptake of, bioavailable P.

4.1.2 Issues and model need

Currently, there is a lack of understanding of why catchment restoration actions undertaken as part of Water Framework Directive (WFD) activities have failed to deliver the ecological improvements that would have been expected. This is in part due to a lack of mechanistic understanding, but also of appropriate data. These catchment actions, applied through the developing WFD river basin management plans, have been designed to tackle P sources, since dissolved P (DP) is a key chemical indicator in the WFD river classification tools and is assumed to be a limiting factor. With this current apparent decoupling of chemical improvements from good ecological status of rivers, there is a need for models to include some of the mechanisms known to bridge the gap between river P loads and ecological status. A number of these factors are discussed in Chapter 1 and summarised above.

Currently, catchment source apportionment models sum the total P loads from a range of catchment sources to evaluate:

- The cumulative impact of sources within catchments
- The sources that contribute most to loads, from which mitigation priorities within and between catchments is inferred
- Potential impairment of waterbody ecological status

For example, the method for load estimation and conversion to estimated average annual concentrations in the Diffuse Pollution Screening Tool is as described in Box 1.

Any proposed development of this approach has to make improvements to the following oversimplifications within such models:

- Phosphorus is generally considered as TP, not as components with differing bioavailability

- Phosphorus interacts with other physico-chemical aspects of the pollution source (such as C, BOD, N or sediment concentrations) and other waterbody conditions that determine impacts

- Sources will differ greatly in their characteristics, including P composition (bioavailability) and delivery (continuous cf. episodic)

Waterbody conditions (e.g. residence times) are difficult to separate from source characteristics in determining impact on ecology and these are not considered in source apportionment models.

Box 1-Summary of how the Diffuse Pollution Screening Tool phase (DPST) estimated mean [SRP] in rivers:

1. TP losses to water are estimated per land use, slope, connectivity, climate and soil type:

A. Sea and Estuary Water (Inland); Semi natural; Inland bare ground (assumed to have zero P export)

B. Rough grass; Forestry; Managed grass; Arable; Urban (have associated P exports)

2. TP losses are estimated using the Psychic model (Davison et al. 2008) for each land use, with incidental soluble P from animal manures and fertilisers using Vadas et al. (2004) and septic sources are based on per capita rates.

The DPST database gives a breakdown of P loads and routes to water for each local catchment as follows:

P_Urban Runoff (%) P_Septic Tanks (%) P_Point Stw (%) P_Road Runoff (%) P_Ag Runoff & Drainflow (%)
P_Ag Seep To Gw (%) P_Forest Runoff & Drainflow (%) P_Forest Seep To Gw (%)

3. The exported P load is dissolved in the annual excess rainfall to give the Perfect Mixer Average Concentration (PMAC) of TP for each local catchment (LC).

4. New time-weighted concentrations were estimated from predicted loads for each of the local catchments where observed values were available. This enabled accurate comparisons of modelled and observed concentrations in these calibration catchments. Information on Hydrology Of Soil Type (HOST) (Boorman et al., 1995) was used to estimate the river flow regime. Each HOST class has an associated Q95 value describing how the soil will affect a river's responsiveness to rainfall. Using this information, Hydrologically Effective Rainfall (HER)-weighted Q95 values were calculated for each catchment. These numbers could then be used in conjunction with information on HER to derive flow exceedance curves using a flow exceedance model (Anthony pers. comm.; Gustard et al. 1992). Pollutant loading from point sources can be assumed to remain constant during the year (although this is not the case for CSOs). The diffuse loading will, however, vary with runoff such that the point source contribution will be relatively higher during drier periods of the year. Having derived flow exceedance curves and assuming a constant concentration in diffuse runoff, it was possible to mix the constant load from point sources with the diffuse contribution varying across the year to calculate time-weighted concentrations.

4. A regression of the new, time weighted PMAC against observed mean SRP for some rivers where mean SRP was available was used to generate a calibration between PMAC and SRP for all rivers:

$\ln(\text{obs SRP}) = 0.714*\ln(\text{PMAC}) - 1.0478$ $r2=49$ $n=597$

This means that the time weighting for constant sources vs flow-related sources (e.g., erosion) has already been allowed for in the estimation of average [SRP]. Therefore in the DPST tool modification described here, we work from the original TP loads, not the PMAC or predicted risk of SRP failure.

4.2 Model description and data sources

4.2.1 Conceptual structure

The main challenge for improving the existing model is identifying and including a few over-arching processes for rivers that are the most ecologically relevant without adding unnecessary complexity to any resulting approaches. The approach developed here is, therefore, designed to provide the next simplistic stage to the above source apportionment models by coupling basic aspects of P form and concentration with river processes governing impact. This is not a highly mechanistic model, but a pragmatic approach to including expert judgement of the indicators of river processes that can be readily derived from catchment attribute data and basic bioavailability data from the literature. In doing this the principles in Table 8 have been applied.

To develop these principles, it is necessary to define the key sources of P that contribute to ecological processes. Our conceptual approach to P speciation and impact of sources is shown in Figures 3 and 4. The following sections describe the process of building the datasets and deciding on the appropriate process complexity. The steps in this process were:

1) Evaluating literature studies for concentrations of different forms P from different sources
2) Evaluating literature studies to determine the likely biovailability of P discharges from the sources and gap filling methodologies where data were not available
3) Compiling literature data on other source factors of transport, delivery and chemical quality
4) Reviewing and prioritising river processes that interact with P concentrations in determining ecological impact
5) Making rules for inclusion of simplistic river processes governing P impacts using readily available river or catchment attributes
6) Running a validation for P sources in a small, mixed land use catchment and comparing the revised source prioritisation with the current methodology, which accounts only for the total P load.

4.2.2 Derivation of model components

Data on catchment pollution sources were evaluated from scientific and grey literature during an initial review phase. This included 27 primary papers, review articles and reports. A decision was made not to include two potential P sources, namely agricultural and forestry seepage to groundwaters. These are often included for N modelling but considered less of a direct influence on river P concentrations in landscapes such as Scotland. No literature could be found on P fractionation or concentrations for industrial wastewaters so this was not included as a source here.

The following 11 sources (and 13 where arable and intensive grassland are separated) were chosen and are listed in approximate order of point to diffuse source behaviour:

Table 8: Principles for the improvement of methods seeking to predict ecological impacts of phosphorus sources in rivers.

Concept	Importance
Phosphorus bioavailability should be considered at the level of the different P forms, including SRP (dissolved unreactive P, i.e. organically complexed) and particulate P. The forms used are shown in Figure 3.	Different sources have varying proportions of different P forms; these vary in bioavailability between sources and in relation to drier/wetter conditions.
Source characteristics, such as delivery and composition of other nutrients (labile C and N) should be accounted for.	Phosphorus uptake is dependent on a range of physical and chemical conditions.
Basic waterbody characteristics should be considered alongside source characteristics	Factors like small catchment size and low baseflow index interact strongly with continuous source delivery to generate exposure to high concentrations in summer.
Impact, in the case of the current model approach, should be defined more specifically than conventional 'eutrophication'; exposure to elevated concentrations of bioavailable P forms that, according to river conditions could result in greater likelihood of P cycling, should be included.	Impact determined through a mechanistic ecological uptake model is too complex for the current application; a simpler set of principles for ecological exposure to available P should be incorporated instead.

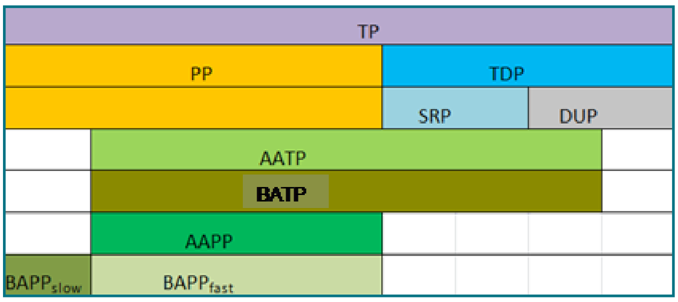


Figure 3. Pools of phosphorus (P) referred to in the River Effective P Sources (REPS) model. TP = Total P; PP = Particulate P; TDP = total dissolved P; SRP = Soluble reactive P; DUP = dissolved unreactive P; AATP = Algal-available total P, which we equate with BATP = biologically active total P; AAPP = algal-available particulate P, which we equate with biologically active particulate P, BAPP$_{fast}$; and algal non-available particulate P, which we equate with slowly available biologically active particulate phosphorus, BAPP$_{slow}$.

This process resulted in the conceptual structure of the model presented in Figure 4. This is termed the River Effective P Sources (REPS) model.

- Sewage treatment works (STW), no P stripping
- STW, with P stripping
- Combined sewer overflows (CSOs; assumed a 50/50 mixture of STW: no P stripping and urban surface runoff)
- Septic tanks
- Agricultural yard runoff including drains
- Freshwater aquaculture
- Rural road or track runoff
- Urban surface runoff
- Agricultural field drainflow (where possible this was subdivided into arable and intensive grassland)
- Agricultural field surface flow (where possible this was subdivided into arable and intensive grassland)
- Forest runoff

Data were compiled on source water characteristics:

- Total P concentrations of sources

- Distribution between: soluble reactive P (SRP; principally PO4), dissolved unreactive P (DUP; organically complexed P), particulate P (PP)

- Other chemistry: Suspended sediment, labile C, or BOD, NO_3+NH_4 concentrations

- Delivery behaviour: Discharge nature (continuous, semi, episodic), rainfall dependence, incidental loss risk (i.e. that is independent of rainfall)

4.2.2.1 *Phosphorus concentrations*

An inventory was compiled of pollutant source chemistry from 27 literature studies, especially focussing on studies where overall total P (TP) concentrations were distributed between the common P fractions of: soluble reactive P (SRP; generally given as the molybdate reactive form determined colorimetrically when filtered to <0.45 μm), dissolved unreactive P (DUP; generally being the difference between total dissolved P minus SRP) and particulate P (PP; generally TP minus TDP). Obtaining all these concentration data and the distribution of TP between these forms reduced the number of relevant studies, in some cases to one. The compiled data are shown in Table 9. Whilst this is recognised as not exhaustive it seems a reasonable assessment of contemporary literature relevant to the nature of sources likely to be found in the UK. In addition to P chemistry other source characteristics were also compiled.

4.2.2.2 *Bioavailability of phosphorus forms*

It was a key aim of the literature review to compile data on the bioavailability of P from source waters and these data were found in two studies of algal available P in batch cultures under similar conditions. Ekholm and Krogerus (2003) measured source water algal-available P under aerobic conditions over 14 to 28 days at 20°C using a 'dual culture' algal assay batch method with previously P-starved Selenastrum capricornutum (Table 10). Ellison and Brett (2006) measured maximum potential algal available P in aerobic conditions over 14 day batch assays at 24°C in the light. However, whilst the former study looked directly at the runoff from the sources, the latter authors characterised stream waters from streams where the catchment was dominated by a single source type (Table 11). An additional benefit of the approach of both studies was that bioavailable P (BAP) was apportioned both into a total pool and that associated with bioavailable particulate P (BAPP). Additionally Ellison and Brett (2006) determined the BATP in the streams under base flow and high flow runoff conditions.

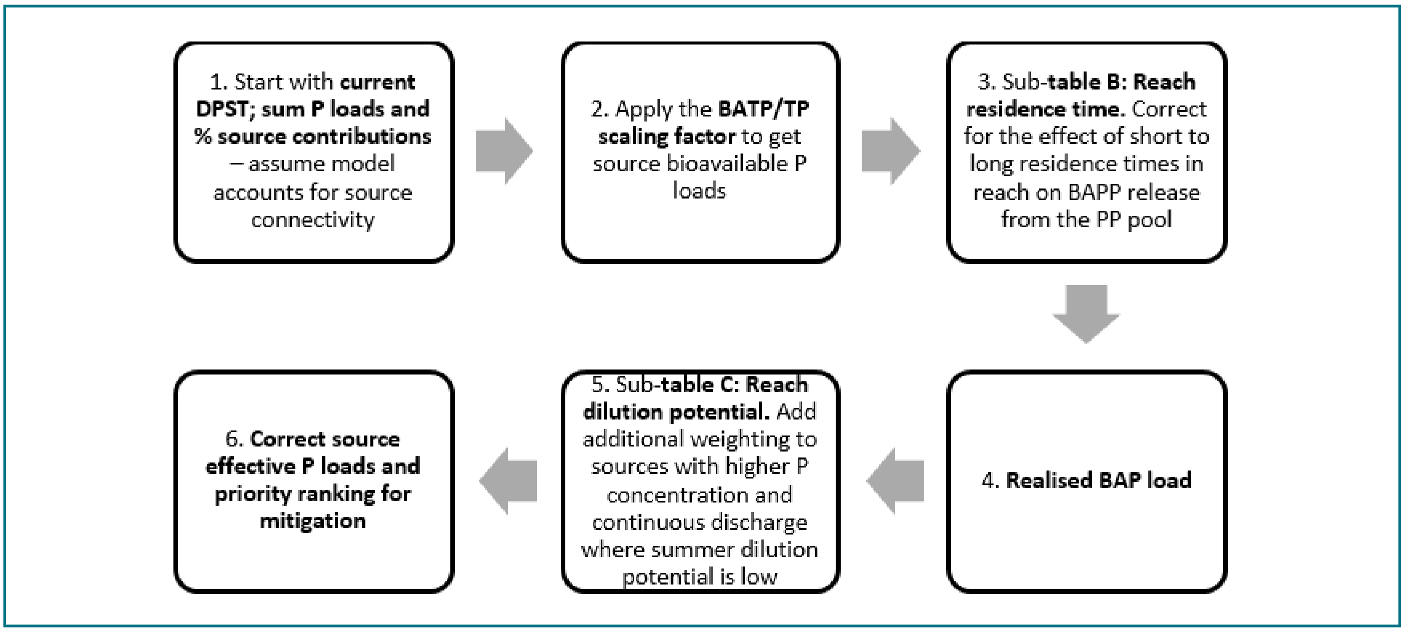


Figure 4. Conceptual structure of the river effective P sources (REPS) model.

Table 9: Literature data compilation on the concentrations and proportions of phosphorus forms for catchment P sources. Abbreviations: TP, Total P; SRP, Soluble reactive P; DUP, Dissolved unreactive P; PP, particulate P.

Sources	P form concentrations in mgP L^{-1}					As % of total P			
			TP			TP			
			TDP			TDP			
	TP mean and range		SRP	DUP	PP	SRP	DUP	PP	Ref
STW: no P stripping	6.63	0.03-17.1	6.03	0.40	0.20	91	6	3	1
	2.9	<DL - 13.1							2
	10								8
			3.99						9
			4.9						15
	2.4	1.2 - 4.6							16
STW: P stripping	0.87	0.07 - 3.57	0.6	0.06	0.21	69	7	24	14
CSOs		1 - 11							8
Septic tanks	10.2	1.0 - 22.0							2
			7.07						10
	9.9	7.0 - 19.6							23
	7.5		7.5						24
	8.1								25
	8.6	7.2 - 17.0							26
	8.6	1.5 - 16.0	5.5	0.44	2.7	64	5	31	27
Agric yard runoff (undefined between livestock and arable yards)	30.8	0.02 - 247							2
	0.77	0.1 - 3.3							4
	2.71	0.08 - 15.0	1.11	0.18	1.42	41	7	52	3
	3.5	up to 51							7
	13.2		11.6 as TDP			88 as TDP		12	11
			3.9	1.4					12
FW aquaculture	0.13		0.06		0.07				13
Rural road runoff	2.03	0.11 - 16	0.38	0.12	1.53	19	6	75	3
Urban runoff	0.29	0.03 - 4.7							5
	0.3					45			6
		0.2 - 1.7							8
Agric field drains	0.72	0.01 - 4.14	0.13	0.06	0.53	18	8	74	3
Agric surface flow	0.97	0.31 - 2.24	0.21	0.09	0.67	22	9	69	3
Arable field drains					0.12				17
	0.78	0.5 - 1.4	0.04	0.03	0.71	5	4	91	21
Intensive grassland drains	0.1	0.03 - 1.15	0.05	0.01	0.04	50	10	40	18
	0.2	0.11 - 0.30	0.005	0.07	0.12	3	35	60	19
Arable surface runoff					0.62				17
	1.05		0.18	0.02	0.84	17	2	80	20
	0.81	0.6 - 1.3	0.04	0	0.78	5	0	95	21
	0.028		0.007	0.005	0.016	25	18	57	22
Intensive grassland surface	0.62	0.25 - 0.95						39	11
	0.11		0.054	0.0025	0.026	49	23	24	226.63
Forest runoff	0.19							94	11

Refs: 1, Neal et al (2005); 2, Edwards & Withers (2008); 3, Withers et al. (2009); 4, Edwards et al. (2007); 5, Kayhanian et al. (2007); 6, Mitchell (2001); 7, Edwards & Hooda (2007); 8, Ellis (1986); 9, Stutter et al. (2010); 10, Demars, pers. Comm.; 11, JEQ, 2005, 34:1224; 12, Edwards et al. (2008); 13, Sindilariu et al. (2009); 14, Neal et al. (2008); 15, Marti et al. (2004); 16, Ekholm et al. (2009); 17, Poirier et al. (2012); 18, Turner & Haygarth (2000); 19, Toor et al. (2004); 20, Roberts et al. (Submitted); 21, Uusitalo et al. (2001); 22, Grant et al. (1996); 23, Current review; 24, Robertson (2012), 25, EPA (2002); 26, Fayetteville; 27, Wastewater program, Idaho.

From these data across Tables 9, 10 and 11 the following calculation stages were employed to derive the source contributions to the total phosphorus concentrations in the waterbody, then scaled to bioavailable P forms:

From Table 9 the mean concentrations of individual P forms contributing to source total P using the following equation:

P form concentration = (mean P form % 100) * mean total P concentration.

It may be considered that some of the literature sources documented may have high P concentrations at lower flows and that a geometric mean (rather than the arithmetic mean used here) would be appropriate to not give undue weighting to large concentrations. This may be the case for yard runoff, but is less likely for STW effluents or field runoff, so therefore an arithmetic mean approach was used.

For each source, the % bioavailable P overall (%BATP) is equivalent to the AATP% and the % of bioavailable particulate P (%BAPP) is equivalent to the AAPP% (see Tables 10 and 11). In the remaining methodology and subsequent worked examples the terms are BATP and BAPP are used. For total P and particulate P the concentrations of BATP overall and of BAPP were calculated according to:

$BATP_{conc}$ = mean total P concentration * (%AATP/100), and
$BAPPc_{onc}$ = mean PP concentration * (%AAPP/100).

Ratios of the concentrations of BATP to overall total P concentrations were derived for two different conceptual pools of particulate P, namely a fast reacting pool being the BAPP and a slow reacting pool being the difference between total PP concentrations and BAPP concentrations:

$BAPP_{fast}/TP = BAPP_{conc}/TP_{conc}$
$BAPP_{slow}/TP = (PP_{conc}-BAPP_{conc)}) / TP_{conc}$

The summary of the source P chemistry is given in Table 12. The main outcome of deriving the BATP values was to gain (a) the BATP/TP ratio as a way to scale total P loads from the source apportionment modelling and (b) to determine the bioavailable particulate P (BAPP) component apportioned between fast and slow pools of bioavailability, as this is used in relation to different river conditions to scale the amount of realised PP contributing to overall bioavailability. The rationale for the split between the fast and slow acting pools of PP was that the algal batch tests generally represent PP available to algae in experimental periods of < 3 weeks. Where the residence time of PP in stream reaches is greater than this period, some of the PP not determined as BAPP in these short term batch studies may become bioavailable and that some of this should be conceptually brought into the BAPP pool under appropriate reach conditions.

Figure 5 shows the results of applying these P fractionation rules in terms of the contributions of these P forms to the source waters. Differences in Figure 5 (a) and (b) between the mean total P concentrations and the bioavailable P largely results from the varying reactivity of the PP pools, as depicted in Figure 5 (c) and (d). For example, STW: no P stripping comprises 91% SRP (assumed to be 100% bioavailable) so the overall BATP is very similar to the total P concentration. Another example is Agricultural yard runoff, where although PP comprises 52% of the total P concentration the overall BATP concentration remains large due to a large contribution of BAPP (46%) of the total PP pool. In contrast, forest runoff has low overall P concentration but negligible BATP as it is dominated by PP (94%) and the BAPP is small (5%). Therefore PP amount and bioavailability exerts a large control on overall P bioavailability between different source waters, resulting in between source differences in ecological impact.

Although for a limited number of data points Figure 6 makes an independent verification of this compilation methodology of ascribing PP bioavailability by comparing data for some of the source waters with a ratio for the reactive inorganic fraction of PP derived by Owens et al. (2002). The relationship in Figure 6 is linear and highly significant ($p<0.001$), giving some validation of the procedures. However, the positive intercept on the x axis indicates that a portion (44%) of the analytically-defined pool of chemically-labile inorganic P determined by Owens et al (2002) would not contribute to the algal available particulate P pool (AAPP and thereby $BAPP_{fast}$ in the current classification) as determined by the culture methods of Ekholm and Krogerus (2003) and Ellison and Brett (2006). This probably indicates a relatively strong chemical extractant and that some of the chemically-labile P would be classed in the $BAPP_{slow}$ pool due to protection from algal uptake afforded by associations with mineral surfaces.

Table 10: Algal-available total P and algal-available particulate P determined by Ekholm and Krogerus (2003) for source runoff waters.

Algal-available P				
	AATP (% of TP)		AAPP (% of TP)	
P source	Mean ± C.I.	Min-max	Mean ± C.I.	Min-max
Septic tank outflows	89 ± 6	74-98	41 ± 31	0-88
STW, no P stripping	83 ± 11	61-103	41 ± 31	0-100
Dairy house	69 ± 32	27-93	46 ± 72	0-100
STW, P stripping	36 ± 10	0-67	25 ± 8	0-54
Field runoff	31 ± 8	15-50	17 ± 5	9-33
Fish farms	29 ± 14	9-72	20 ± 16	0-76
Forest runoff	16 ± 8	0-55	4.5 ± 8.8	0-58

Table 11: Algal-available P determined by Ellison and Brett (2006) for stream waters dominated by single land use types. Within the table concentrations for TDP + PP do not equal TP due to the geometric means taken across samples by the authors.

Sources	Flow	P concentrations			Bioavailable P		
		TP	TDP	PP	AATP	AAPP	AAP
			µg/L		µg/L		%
Forest	Baseflow	30	18	10	20	6	
	Storm	78	25	52	31	9	20
Mixed	Baseflow	55	35	19	40	16	12
	Storm	1000	34	65	49	22	29
Urban	Baseflow	69	50	16	61	50	22
	Storm	255	57	187	92	48	73
Agric	Baseflow	133	66	31	73	29	19
	Storm	313	111	145	149	81	22

Table 12: Calculation of concentration proportions of total P and bioavailable total P across individual P forms, using calculations based on previous Tables 9-11.

	% Bioavailable P			Mean P concentrations (mg P/L)								Concentration ratios of P forms to TP		
				TP	n[4]	Specific P form				Bioavailability				
	% BATP[1]	% BAPP[2]	Ref[3]			SRP	DUP	PP	n[5]	BATP	BAPP	SRP/ TP	BAPPfast/ TP[6]	BAPPslow/ TP[7]
STW: no P stripping	89	40	a	6.63	4	6.03	0.40	0.20	1	5.90	0.08	0.91	0.01	0.02
STW: with P stripping	36	25	a	0.87	1	0.60	0.06	0.21	1	0.31	0.05	0.69	0.06	0.18
CSOs	63	30	c	6.00	1	4.08	0.18	1.74	1	3.78	0.52	0.68	0.09	0.20
Septic tanks	89	41	a	8.82	6	5.64	0.45	2.77	1	7.85	1.13	0.64	0.13	0.19
Agric yard runoff	69	46	a	10.20	5	4.18	0.68	5.34	1	7.04	2.46	0.41	0.24	0.28
Freshwater aquaculture	29	20	a	0.13	1	0.06	0.00	0.07	1	0.04	0.01	0.46	0.11	0.43
Road runoff	37	24	b	2.03	1	0.38	0.12	1.53	1	0.75	0.37	0.19	0.18	0.57
Urban runoff	70	46	b	0.30	2	0.13	0.00	0.16	1	0.21	0.07	0.45	0.25	0.30
Agric drain flow	31	17	a	0.72	1	0.13	0.06	0.53	1	0.22	0.09	0.18	0.13	0.61
Agric surface flow	31	17	a	0.97	2	0.21	0.09	0.67	1	0.30	0.11	0.22	0.12	0.57
Arable drain flow	31	17	a	0.78	1	0.04	0.03	0.71	1	0.24	0.12	0.05	0.15	0.76
Grassland drain flow	31	17	a	0.15	2	0.04	0.03	0.08	2	0.05	0.01	0.26	0.09	0.42
Arable surface flow	31	17	a	0.63	3	0.10	0.04	0.49	3	0.20	0.08	0.16	0.13	0.65
Grassland surface flow	31	17	a	0.37	2	0.18	0.08	0.09	1	0.11	0.01	0.49	0.04	0.20
Forest runoff	16	5	a	0.19	1	0.01	0.01	0.18	1	0.03	0.01	0.03	0.05	0.89

Key: 1, %BATP represents the % of the total P concentration that is bioavailable; 2, %BAPP represents the % bioavailable particulate P; 3, Denotes the study from which the BATP values are derived either: a, Ekholm & Krogerus (2003) or b, Ellison & Brett (2006) or c, values for CSOs represent the mean of STW: no P stripping (Ekholm & Krogerus, 2003) and urban runoff during storm flow conditions (Ellison & Brett, 2006); 4, n = number of studies from which mean TP concentrations were derived; 5, n = number of studies from which the distribution of P form concentrations were derived; 6, BAPPfast/TP = the concentration ratio of $BAPP_{conc}/TP_{conc}$; 7, $BAPP_{slow}/TP$ = the concentration ratio of slowly reacting PP (as defined by total $PP_{conc} - BAPP_{conc}$) / TP_{conc}

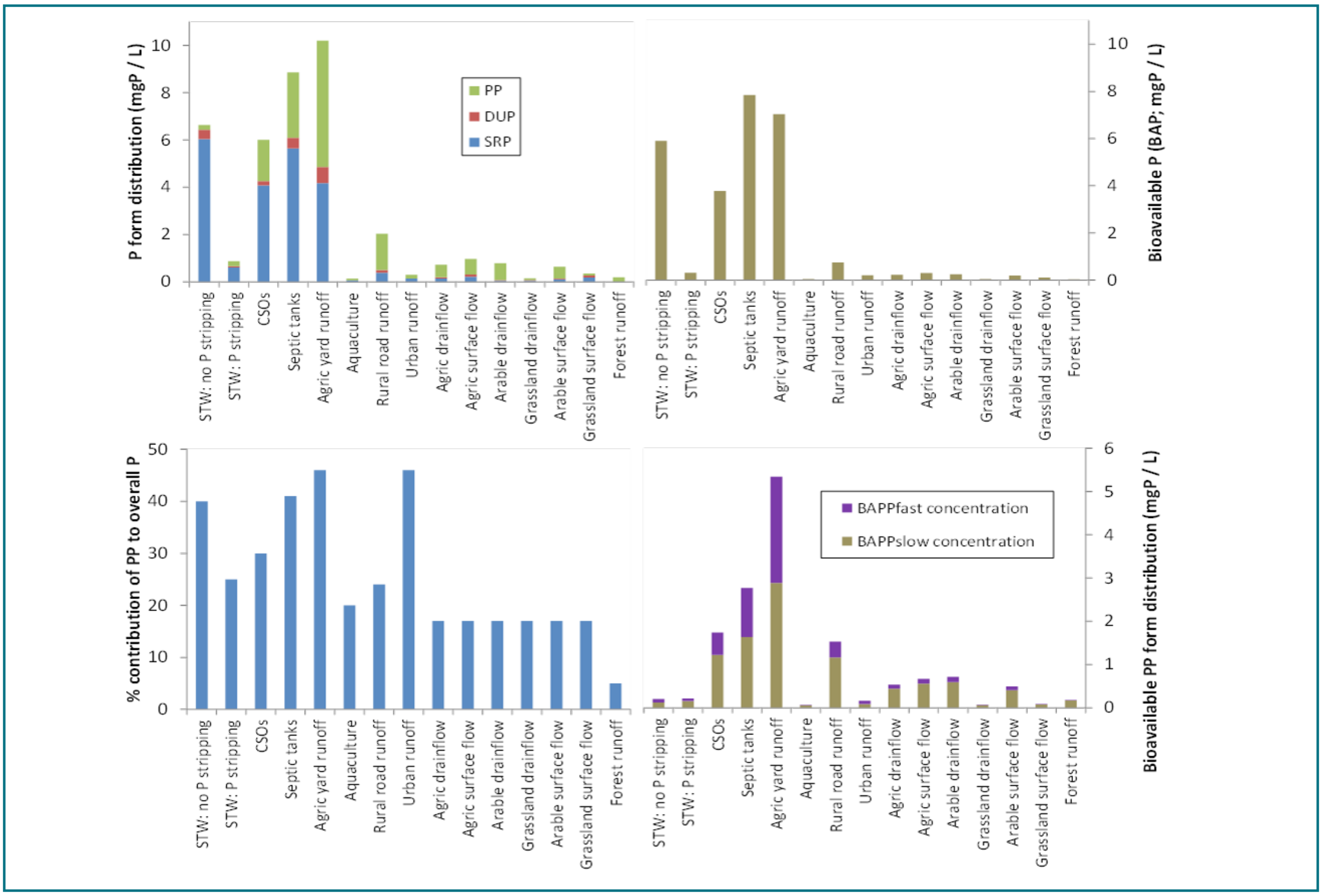


Figure 5. Compilation of different phosphorus concentrations across P forms and bioavailable P derived from literature studies. (a) Mean total P concentrations (bar height), and P form contributions to total P, for different pollution source waters. (b) Overall mean bioavailable P concentrations. (c) The percentage of particulate P form to overall total P concentrations. (d) The distribution of the particulate P into a fast acting pool (BAPP) and a slow acting pool (PP-BAPP).

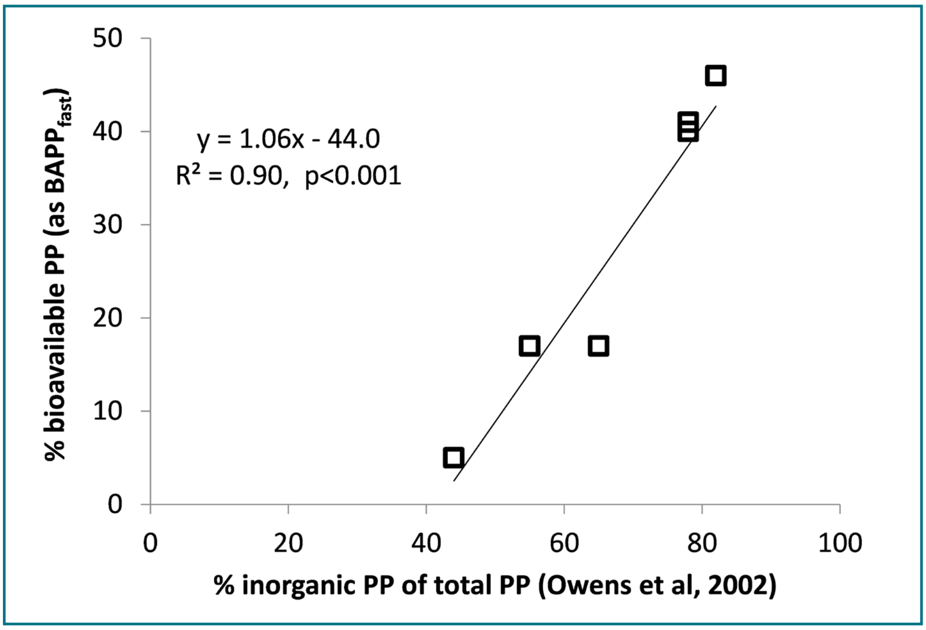


Figure 6. Relationship between the % of PP that is bioavailable (in the $BAPP_{fast}$ pool) calculated using the procedures developed in this study and a chemical assay of inorganic reactive PP as a % of total PP reported by Owens et al. (2002).

4.2.2.3 *Evidence for the magnitude of contributions to P impacts from the BAP_{Pslow} pool*

The original data from Ekholm and Krogerus (2003) carried forwards into the methodology defines small contributions of bioavailable PP (here termed $BAPP_{fast}$) for agricultural and forest runoff (17% and 4.5% of the total PP pool, respectively). Hence, for these diffuse pollution sources, PP is dominated by less bioavailable PP, likely to be turned over much more slowly in aquatic ecosystems (here termed $BAPP_{slow}$). Sub-table B brings up to 50% of this BAPPslow into the 'available' pool according to river conditions. The justification for this comes from observations from sediment tracing studies in two agricultural headwaters: the Tarland Burn and Lunan Water in NE Scotland (Stutter, Unpublished data). In two separate studies, sediment sources were taken from field topsoil transects parallel to (i) a small stream in Tarland with mixed arable and intensive grassland land use and (ii) a small stream with arable land use in Lunan. Soils were compared with stream bed sediments for a range of properties including organic carbon content and acid ammonium oxalate extractable Fe, Al and P, having first been sieved to common particle sizes of <250 μm for Tarland (by dry-sieving) and 150 μm for Lunan (by wet-sieving). The hypothesis of this study was that the release of soluble reactive P to waters associated with incoming eroded soils (i.e. exchange from particle to reactive dissolved P in the water column) is equal to the difference between soil P minus river bed sediment P, and that the exchangeable pool of P was best represented by that associated with the sorption surfaces of amorphous Fe and Al complexes (P_{ox}; as determined by extraction with acid ammonium oxalate).

The results of this study showed that for Tarland arable topsoil had P_{ox} contents of 1431 ± 179 mgP kg^{-1}, whilst the stream bed sediment had 805 ± 358 mgP kg^{-1}. The Lunan arable soil had P_{ox} contents of 1036 ± 66 mgP kg^{-1}, whilst the corresponding stream bed sediment had 364 ± 245 mgP kg^{-1}. These values indicated that 43% and 65% of the P_{ox} at Tarland and Lunan, respectively, was missing between the parent soil from which the eroded sediment had originated and the stream sediment. It was expected that, since these samples were taken in summer that the sediments would represent residence times in the stream on the order of a month or more. It was also assumed that the arable soil dominated the erosion sources in Tarland (and in Lunan was the sole land use in the catchment). Therefore this supported the assumption that approximately 50% of the sorbed P associated with stream bed sediments may be brought from the BAPPslow pool into a pool considered 'available' in this modelling approach. This is an area that ought to be refined in subsequent approaches as these site-specific data would be weak for a national approach.

4.2.3 Interactions of pollution source and river characteristics

Table 13 shows a broad assessment of different river physical and habitat conditions that could interact with nutrient concentrations in determining the ecological processing of P and the resulting degree of ecological impact. These are presented as a means of considering the many processes implicated in ecosystem P cycling. However, to account for all these would necessitate a very complex mechanistic model of chemical-physical/hydromorphological-biological/habitat processes and this is beyond the scope of this current approach. Our goal is to bring a next step in source apportionment modelling that allows the ecologically-relevant factors of P delivery from source waters to be combined with some basic river conditions to provide an improvement in P source modelling. Hence, this approach concentrates on the basic physical conditions of:

- Residence time for processing of P forms
- Scouring and energetics of the reach that would retain or mobilise the variable bioavailable fractions associated with particulate P
- Dilution of P concentrations associated with diffuse and point P sources differing in delivery between continuous and episodic nature

It was therefore decided that this approach would comprise factors that related to ecological exposure to elevated bioavailable P forms and basic effects of in-waterbody processing rates, concentrating on source P chemistry, delivery and reach flow dynamics. It was assumed that the results could then be run through an additional stage comprising important attributes such as temperature and riparian condition and that this P bioavailability exposure component and a subsequent more mechanistic ecological processing component would together be used in the validation against, and future prediction of, ecological metrics. Two examples are presented in Figures 7 and 8 of the interactions of two different pollution sources with two different waterbodies. The interactions are conceptualised into a matrix of four components, namely the pollution and delivery characteristics of the source as well as the flow and internal processing capacity of the waterbody.

Factor	Impact on continuing P retention		Rationale	Possible indicators
	negative impact	positive impact		
Flow regime	Low water level, slower flow, especially in summer	Flushing/ scouring flows	Slower moving waters and those with longer residence times optimise conditions for greater biological processing of nutrients. Incidence of low flow relates to limited ability to dilute point sources	Baseflow index; Catchment soil HOST class; Catchment area; Stream order; Average catchment slope
Water depth; water clarity	Greater depth, Less clarity		Greater depth and/or less clarity of the water reduce light penetration and limits growth of photosynthetic organisms and plants. Greater depth limits the processing capacity of the bed sediment and benthic periphyton on the water column (via area:volume ratio) and exchange with suspended sediment (delivered or remobilised) becomes more important	Stream order; Stream form (pool, rifle, glide); Catchment soil texture (i.e. % clay)
Temperature	Higher temperature		Nutrient processing rates increase rapidly with temperature, especially processes associated with heterotrophs. Longer growing period overall. Water temperature responds more rapidly to rising air temp in low order streams	Mean catchment altitude; Region; Stream order

Table 13: Broad consideration of the interactions of river physical and habitat condition with nutrient concentrations in determining ecological impacts.

Table 13 continued: Broad consideration of the interactions of river physical and habitat condition with nutrient concentrations in determining ecological impacts.

Factor	Impact on continuing P retention		Rationale	Possible indicators
	negative impact	positive impact		
Riparian condition & stream shading	Degraded riparian habitat	More shading, woody debris	Higher biodiversity riparian habitats add resilience to waterbody nutrient processing capacities and lead to greater and more prolonged P retention capacities, especially broad leaved woodland. Reduced light through shading by riparian vegetation limits growth of photosynthetic organisms and plants, but can provide woody debris and labile C inputs via leaves.	Riparian habitat condition score, % of riparian area in natural woodland
Grazer abundance		More grazers	Increased abundance of grazers removes photosynthetic organisms	Trophic state
Other nutrient supplies (C, N, Si)	Greater labile nutrient concentrations		C, N, P and (Si in the case of diatoms) must be present in an appropriate balance and in available forms for eutrophication to develop. Greater C input leads to increased prevalence of benthic anoxia leading to P desorption.	Flow weighted mean DOC, DIN, Si concentrations

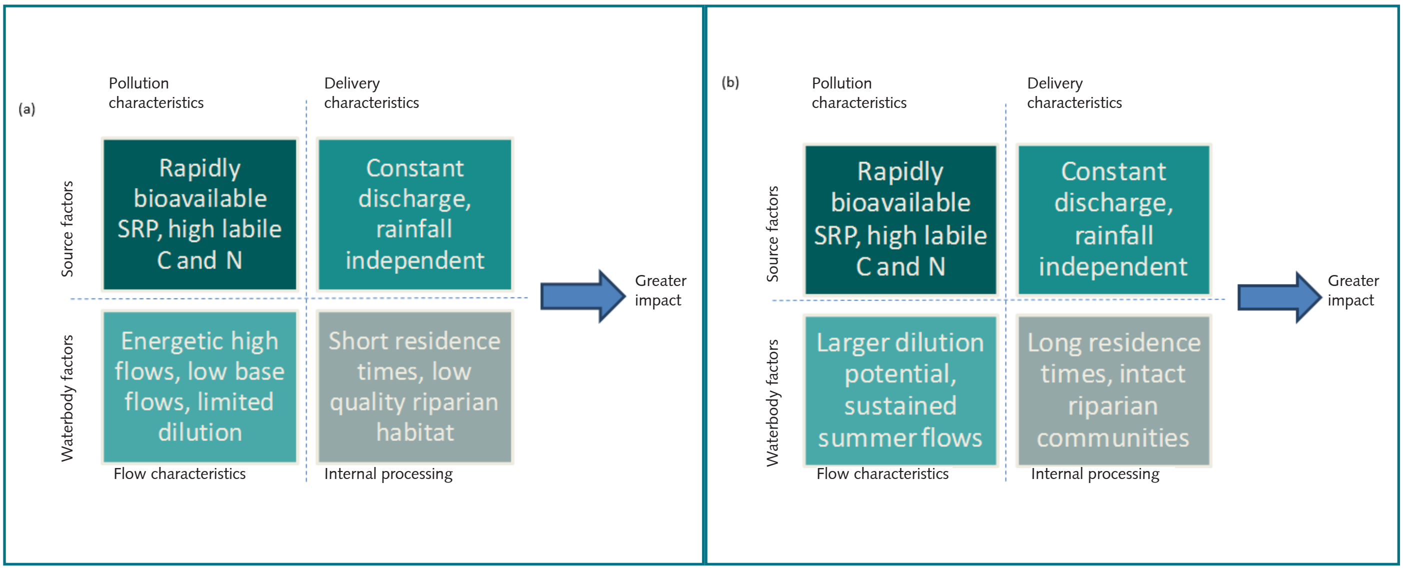


Figure 7. Demonstration of the concept of how waterbody and pollution source characteristics interact to determine ecological impacts associated with phosphorus. The figure illustrates how a **small sewage treatment works** would have a larger ecological impact discharging into (a) a smaller tributary river, of low baseflow index and with additional pressures of intensive agriculture, and a smaller ecological impact with the same discharge into (b) a larger, mixed land use, ground water fed (high baseflow index) mainstem river.

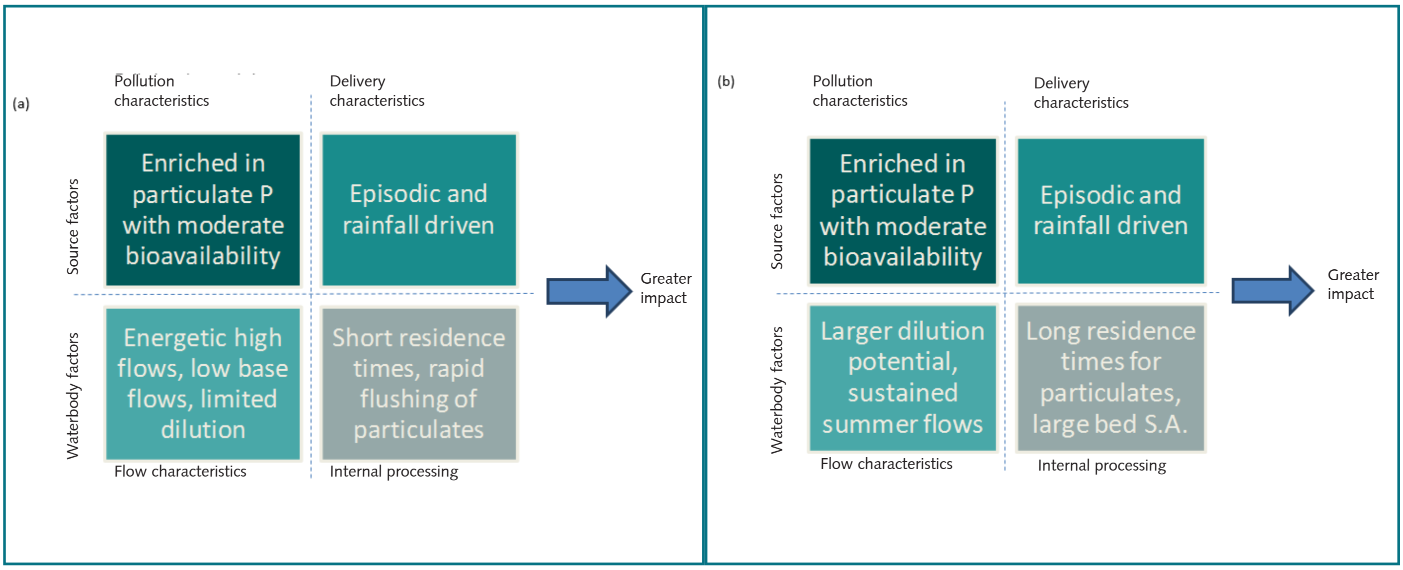


Figure 8. Demonstration of the concept of how waterbody and pollution source characteristics interact to determine ecological impacts associated with phosphorus. The figure illustrates how **intensive arable runoff sources** would have a smaller ecological impact discharging into (a) a small tributary stream, of low baseflow index, and a larger ecological impact with the same discharge into (b) a larger, mixed land use, ground water fed (high baseflow index) mainstem river. S.A. is surface area.

4.2.4 Implementation of source and river interactions

These interactions between source and waterbody characteristics are implemented via a series of sub-tables acting to process characteristics of source and waterbody into risk factors. The first step was the derivation of a set of basic scores for the discharge nature, concentration and other pollutants associated with the source waters (Table 14). The discharge nature reflects the delivery of the waters being continuous for some point sources such as sewage treatment works and aquaculture, these score 3 for high risk due to the potential for continuous inputs, independent of rainfall, to give ecological impact in low dilution, baseflow conditions in critical summer ecological periods. Episodic delivery is shown by rainfall driven sources such as agricultural surface flow, road and urban runoff and these are classified as 1 for lower risk due to the delivery only during high flow events at periods of dilution, often outside of ecologically-sensitive periods. Combined sewer overflows classify as episodic as they tend to occur when urban surface runoff overwhelms STWs and drainage during storms. Agricultural drain flow has delivery characteristics between episodic to semi-continuous since they may flow all year but increase greatly during autumn to spring conditions on wet soils whilst having a lag in flow relative to rainfall. Septic tanks are considered as episodic to semi-continuous since they have some continuous flow but many older systems flush with rising soil water tables and this is even more episodic if connected to roof runoff. The impacts of these discharge characteristics have been previously discussed in detail by Withers and Jarvie (2008).

The risk categories for phosphorus concentrations are considered here to be low risk (score 1) for <2.5 mg BATP/L, medium risk (score 2) for 2.5 to 5.0 mg BATP/L and high risk (score 3) for >5.0 mg BATP/L. Bioavailable P concentration is used in this scoring since it is meant to combine with delivery risk to give the score Concentration × Delivery that determines the ecological impact risk interaction with waterbody dilution and residence time (via Sub-tables A and C). The "*other pollutant*" score uses a mixture of data and expert knowledge to attribute a low to high risk of impact from the processing of P mediated by availability of labile organic carbon and dissolved inorganic N. However, this is already accounted for in using the bioavailable P values from the batch algal assays so the decision was taken not to further scale for this in the final P impact outputs.

The full process of the model in scaling the source P loads has been summarised in Table 8. The steps are described below and relate to the descriptions of the scaling factors in Table 14 and the sub-tables provided in Table 15:

1) The primary input to the process is the output from the current Diffuse Pollution Screening Tool (DPST) methodology (see Box 1). This provides for each waterbody an inventory of the contributing sources and their overall total P loads. The assumption is made that these DPST loads already have been scaled for source connectivity. Hence, the River Ecological P Significance (REPS) method described in steps 2-6 is a scaling factor on these DPST outputs, not a catchment source-transport model.

2) Onto these source-apportioned total P loads, the BAP/TP ratios are applied to convert the total P into ecologically-relevant bioavailable P loads.

3) Sub-table B is then used to scale the contribution of bioavailable particulate P according to waterbody factors. The waterbody input data for this step are the Base Flow Index (BFI) and the average catchment slope. These are designed to be readily-available catchment datasets from GIS, for example, the BFI can be calculated from Hydrology of Soil Types (HOST; Boorman et al. 1995) GIS data. The concept for these data are that high BFI denotes stable summer flows, damped from rainfall that would indicate high residence times for particulate P in spring to summer. The presence of low slope angles would accentuate this propensity to stable base flow in summer and a lack of scouring flows. In such cases the Sub-table B acts to make progressively more of the $BAPP_{slow}$ pool available (i.e. assuming sediment residence times greater than the 14-28 days batch incubations of the algal assays that define the $BAPP_{fast}$ pool). Conversely high slope angles and low BFI denote flashy hydrology and greater likelihood of scouring flow events in summer that would limit residence times. At the extreme category of this risk, Sub-table B acts to remove some of the contribution from the BAPPfast pool (i.e. assuming sediment has less residence than the 14-28 days of the algal assays.

The result of this is the realised BATP load adjusted for residence time and sediment internal processing.

4) A final adjustment is then applied via Sub-tables A and C to increase the scaling of sources acting to deliver more concentrated flows of bioavailable P with a continuous discharge characteristic. Table 14 shows how the source characteristic of delivery (episodic, semi-continuous and continuous) was translated into ascending risk classes (1, 2 and 3, respectively). Additionally, the source characteristic of P concentration and form (BATP categories of <2.5, 2.5 to 5.0 and >5.0 mgBAP/L) has been translated into ascending risk classes (1, 2 and 3, respectively). In Sub-table A, these are brought together to give a Concentration × Delivery score. The rationale for this is that there is a disproportionate risk of P impact for continuous delivery sources that are highly concentrated in BAP. Sub-table C then acts to bring this source term of Concentration × Delivery risk together with waterbody indicators of propensity of summer low flows, namely BFI and stream order. The rationale is that low BFI and low stream order (i.e. a surrogate for small catchment size and lower percentage runoff) maximise the risk of low summer flows. It is recognised that such extreme flows often do not last for long periods but are critical for ecological exposure to the most rapidly bioavailable P inputs when these come from continuous delivery (non-rainfall dependent), concentrated sources. In Sub-table C, the adjustment is made to the SRP component of the total P load in recognition of the rapid bioavailability of this P form.

Following this final adjustment the output is the final corrected source ecological P weighting.

Table 16 summarises the processes that are included in the approach via Tables 14 and Sub-tables A, B and C (Table 15). A wide range of factors are considered, according to the results of the literature review chapter and Table 13, with the rationale for inclusion of otherwise of that group of processes.

Table 14: Derived risk scores for pollutant source characteristics of discharge, concentration, discharge × concentration and other pollutant components. Scores 1 to 3 represent an order of low to high risk of source impact.

	Delivery		P concentration			
Sources	**Nature[1]**	**Score**	**BATP mg/L**	**Score[2]**	**Concentration × delivery score[3]**	**Other pollutant score[4]**
STW: no P stripping	C	3	5.90	3	3	2
STW: with P stripping	C	3	0.31	1	22	2
CSOs	E	1	3.78	2	3	3
Septic tanks	E, S	2	7.85	3	22	3
Agric yard runoff	E	1	7.04	3	1	3
Freshwater aquaculture	C	3	0.04	1	1	2
Road runoff	E	1	0.75	1	1	1
Urban runoff	E	1	0.21	1	1	1
Agric drainflow	S,E	2	0.22	1	1	1
Agric surface flow	E	1	0.30	1	1	2
Arable drainflow	S, E	2	0.24	1	1	1
Grassland drainflow	S, E	2	0.05	1	1	1
Arable surface flow	E	1	0.20	1	1	2
Grassland surface flow	E	1	0.11	1	1	2
Forest runoff	E	1	0.03	1	1	1

Key: [1]The Nature of delivery: E - Episodic, S - Semi-continuous, C - Continuous; [2] P concentration score based on the bands 1=<2.5 mgBAP/L, 2=2.5 to 5.0 mgBAP/L, 3=>5mgBAP/L; [3] Concentration × delivery score based on Sub-table A; [4] Other pollutant score based on expert knowledge of associated dissolved inorganic N and labile C concentrations.

Table 15: Sub-tables A, B and C used to scale source phosphorus according to a basic set of waterbody interactions.

Sub-table A. Defining concentration × discharge score

Discharge	Concentration score		
score	1	2	3
1	1	1	2
2	1	2	3
3	2	3	3

Low risk top left to high risk bottom right

Sub-table B. Defining PP pool availability according to waterbody residence time

	Mean catchment slope		
BFI	>13%	8-13%	<8%
<0.3	-10%$BAPP_{fast}$	no change	+10%$BAPP_{slow}$
0.3-0.7	no change	+10%$BAPP_{slow}$	+30%$BAPP_{slow}$
>0.7	+10%$BAPP_{slow}$	+30%PP_{slow}	+50%$BAPP_{slow}$

Low risk top left to high risk bottom right

Actions: (i) for high scouring, subtract some of the fast PP pool (i.e. accounted for in the 20 day algal P bioavailability tests), (ii) for high residence time, make some of the remaining slow PP pool bioavailable.

Sub-table C. Defining concentration × discharge score effect on share of ecological impact score

	Stream order		
BFI	>5	3 to 5	<3
>0.7	no change	no change	Add 10% of SRP load for score 3
0.3-0.7	no change	Add 10% of SRP load for score 3	Add 30% of SRP load for score 3
<0.3	Add 10% of SRP load for score 3	Add 30% of SRP load for score 3	Add 50% of SRP load for score 3, add 20% of SRP load for score 2

Low risk top left to high risk bottom right

Actions: for smaller rivers/streams with low BFI (i.e. risk of extreme summer lows) increase the weighting of SRP loads for sources designated as high P concentration with continuous discharge (defined in Sub-table A)

Table 16: Summary of the factors considered and included in the model approach.

Factor	Is this incorporated?	Data requirements	Other considerations
Source load, transport and connectivity	No, considered part of a preceding model output	Output of average annual total P load proportioned over catchment sources	The DPST model structure needs to be validated itself and assessed to ensure no process double accounting with this proposed procedure.
Flow regime	Yes, via: The waterbody hydrological energy (assessed in Sub-table B) to determine residence and amount of realised bioavailable particulate P; The potential for low summer flows (assessed in Sub-table C) to determine the dilution potential of concentrated continuous source inputs of SRP	Catchment baseflow index (BFI) and mean slope angle Catchment BFI and stream order	?
Water depth; water clarity	No, as: The interactions between light penetration and P cycling in rivers; and between water column to bed area, are considered too complex	N/A	
Temperature	No, this interaction is considered too complex to add a specific catchment modifier.	N/A	Some account is made in scaling up the impact of continuous concentrated P sources in waterbodies considered to have risk of low summer flows, as such flow conditions caused warmer waters.
Riparian condition and stream shading	No, this interaction is considered to be high risk of lack of appropriate data availability.	N/A	
Grazer abundance	No, this ecological process is considered too complex.	N/A	
Other nutrient supplies (C, N, Si)	Not as a further modifier	N/A	Included via the batch algal assays used to determine the bioavailable P (BAP) fraction
Chemical reactivity of sediments	Yes, via: The split of overall BATP into total and two conceptual pools of particulate P bioavailability ($BAPP_{fast}$ and $BAPP_{slow}$); The scaling of the amount of P realised from the BAPP by interaction with waterbody residence time factors (in Sub-table B).	Literature data on the split of BATP across P forms Catchment baseflow index (BFI) and mean slope angle	An antagonistic process is the supply of fresh eroded sediment surfaces for P sorption with erosive flows. However, due to bed sediment factors (surface area, redox, geochemistry) this is considered difficult to predict.

4.3 A worked example for a small river catchment

A validation of the approach is given below for the Tarland catchment, a tributary of the River Dee in northeast Scotland. Tarland is a 50 km^2, mixed land use catchment. The current status under the Water Framework Directive classification system is good for SRP status, but moderate for ecological status. The total P source apportionment has previously been determined as part of a national screening of WFD waterbodies using the Diffuse Pollution Screening Tool (DPST). The output of this model calculated a 0.24 kgP / ha / year total P load, indicating 1200 kgP annual load at the catchment scale. This was apportioned between sources as: 6% urban runoff, 13% septic tanks, 6% STWs, 2% road runoff, 2% forestry, 70% agriculture. The following assumptions were applied to the catchment data:

- The 70% total P contribution from agriculture was split: 10% yard, 32% arable, 28% intensive grassland (on the basis of the land cover areas of 1875 ha cropland and 1630 ha intensive grassland) and an assumed 70/30 distribution between surface and drain flow pathways.

- For Sub-table B the baseflow index (BFI) was set to category 0.3 to 0.7 and slope category <8% giving a BAPP modifier of +30% $BAPP_{slow}$.

- For Sub-table C the BFI was category 0.3 to 0.7 and the stream order category 3 to 5 giving an SRP dilution modifier of +10% of SRP load for Concentration × Discharge score 3. The data for this validation is presented in Table 17, with Figures 9 and 10 showing the absolute and proportional changes in the P loadings with progressive steps of the approach.

The raw output data of the Diffuse Pollution Screening Tool (DPST) for total P loads shows contributions in order of: arable equivalent to improved grassland surface flow > septic tanks > farm yard runoff > arable equivalent to intensive grassland drain flow > STW ≈ urban runoff > road runoff ≈ forest runoff (Figure 10a). The greatest change in the absolute and proportional P loading comes with step 2, the conversion of total P to BATP using the TP/BATP ratios for each source. The impact of this is clear in Figure 9 and Figure 10b in large reductions of the effective P load associated with the diffuse sources of arable and intensive grassland surface and drain flow, road and forest runoff. Following this scaling to BATP, septic tanks become the dominant effective P source (from 17% to 24%) with relative increases in yard runoff, STW and urban runoff. In step 3, the moderate residence time and scouring potential acts to add 30% of the $BAPP_{slow}$ pool back into the effective P contributions, so gains are then made for sources more concentrated in PP associated with soil erosion such as arable surface and drainflow and forest runoff. Step 5 then makes only a small further difference by adding 10% of the SRP loads for sources STW and septic tanks accounting for the limited dilution potential of the waterbody in summer. The results of steps 3-6 make little overall difference to the effective P load relative contributions for the sources between Figure 10 parts (b) and (c). However, in terms of the absolute effective P loads this is increased through the modifiers of steps 3-6 from 572 kgP / year after step 2 to 729 kgP / year final effective P load. In summary, the approach increases the importance of the effective bioavailable P contributions from point sources relative to diffuse sources and would indicate the greater priority in addressing point sources to improve ecological waterbody status.

4.3.1 Model uncertainties

There are a number of accumulating uncertainties in the procedure. Probably the greatest are inherited from the pre-approach source apportionment model output. The uncertainties in this of overall catchment P load and the proportional distribution between sources is undefined by the DPST model used in this example. However, previous work with the DPST has compared predicted and observed P data for a limited number of streams with appropriate monitoring data. We would propose to extend this approach of validation in future tuning of the proposed model with river P concentration and ecological water quality data.

Of the steps taken in the current approach, Table 10 gives the confidence intervals for the BATP scaling factor derived from the study of Ekholm and Krogerus (2003). These give the only quantified errors in this procedure as direct error in the determination of the BAPPslow pool on which Sub-table B acts is not given. It could be inferred by scaling the BAPP confidence intervals in Table 10 but this is not done. Hence, the error bars given in Figure 9 show the 95% confidence error (as positive error only for clarity) using Table 10 data and an assumption of confidence intervals of ±0.1 for rural track and urban surface BAP. The errors shown may be compared to the difference between the raw DPST P load and the BATP scaled P load. For diffuse P sources (road, urban surface, agricultural field and forest runoff) the errors are small compared with the difference between BATP and total P loads. Where BATP is a large proportion of total P load for STW and septic tanks, the error is equivalent to the difference. For farm yard runoff the error is greatest and positive error matches the differences between total and BAP-scaled P loads.

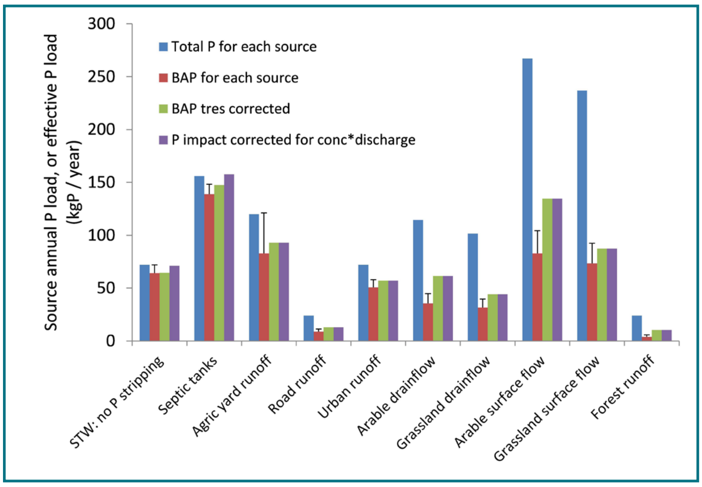


Figure 9. Effect of the successive scaling factors, building from unscaled on the left (blue bars) to fully scaled on the right (purple bars), acting on total P load from the Diffuse Pollution Screening Tool output to determine effective bioavailable P via three stages of the model approach. Error bars for BATP indicate the 95% confidence intervals taken from data in Table 10 (only positive error bars are shown for clarity). Tres is the residence time (T_{res}).

Table 17: Model data and output for scaling P sources for impacts in the Tarland catchment.

Model compiled data										P load data for Tarland and effect of modifiers						
										KgP/yr =1200			Sub-table B: add 30%PP_{slow}		Sub-table C add 10% of SRP load for cat 3 only	
Sources	TP	BAP	SRP /TP	BAP /TP	$BAPP_{fast}$	$BAPP_{slow}$	Source delivery score	Source Conc Score	Conc × delivery score	%	Total P for each source	BATP for each source	Residence time corr	BATP t_{res} corrected	Corr for conc*discharge	Effective P
STW: no P stripping	6.60	5.90	0.91	0.89	0.01	0.02	3	3	3	6	72	64	0	64	7	71
STW: with P stripping	0.90	0.31	0.69	0.36	0.06	0.18	3	1	2	0	0	0	0	0		0
CSOs	6.00	3.78	0.68	0.63	0.09	0.20	1	2	2	0	0	0	0	0		0
Septic tanks	8.80	7.85	0.64	0.89	0.13	0.19	2	3	3	13	156	139	9	148	10	157
Agric yard runoff	10.2 0	7.04	0.41	0.69	0.24	0.28	1	3	2	10	120	83	10	93		93
Aquaculture	0.10	0.04	0.46	0.29	0.11	0.43	3	1	2	0	0	0	0	0		0
Road runoff	2.00	0.75	0.19	0.37	0.18	0.57	1	1	1	2	24	9	4	13		13
Urban runoff	0.30	0.21	0.45	0.70	0.25	0.30	1	1	1	6	72	51	6	57		57
Arable drainflow	0.80	0.24	0.05	0.31	0.15	0.76	2	1	1	10	114	35	26	61		61
Grassland drainflow	0.20	0.05	0.26	0.31	0.09	0.42	2	1	1	9	102	31	13	44		44
Arable surface flow	0.60	0.20	0.16	0.31	0.13	0.65	1	1	1	22	267	83	52	135		135
Grassland surface flow	0.40	0.11	0.49	0.31	0.04	0.20	1	1	1	20	237	73	14	87		87
Forest runoff	0.20	0.03	0.03	0.16	0.05	0.89	1	1	1	2	24	4	6	10		10

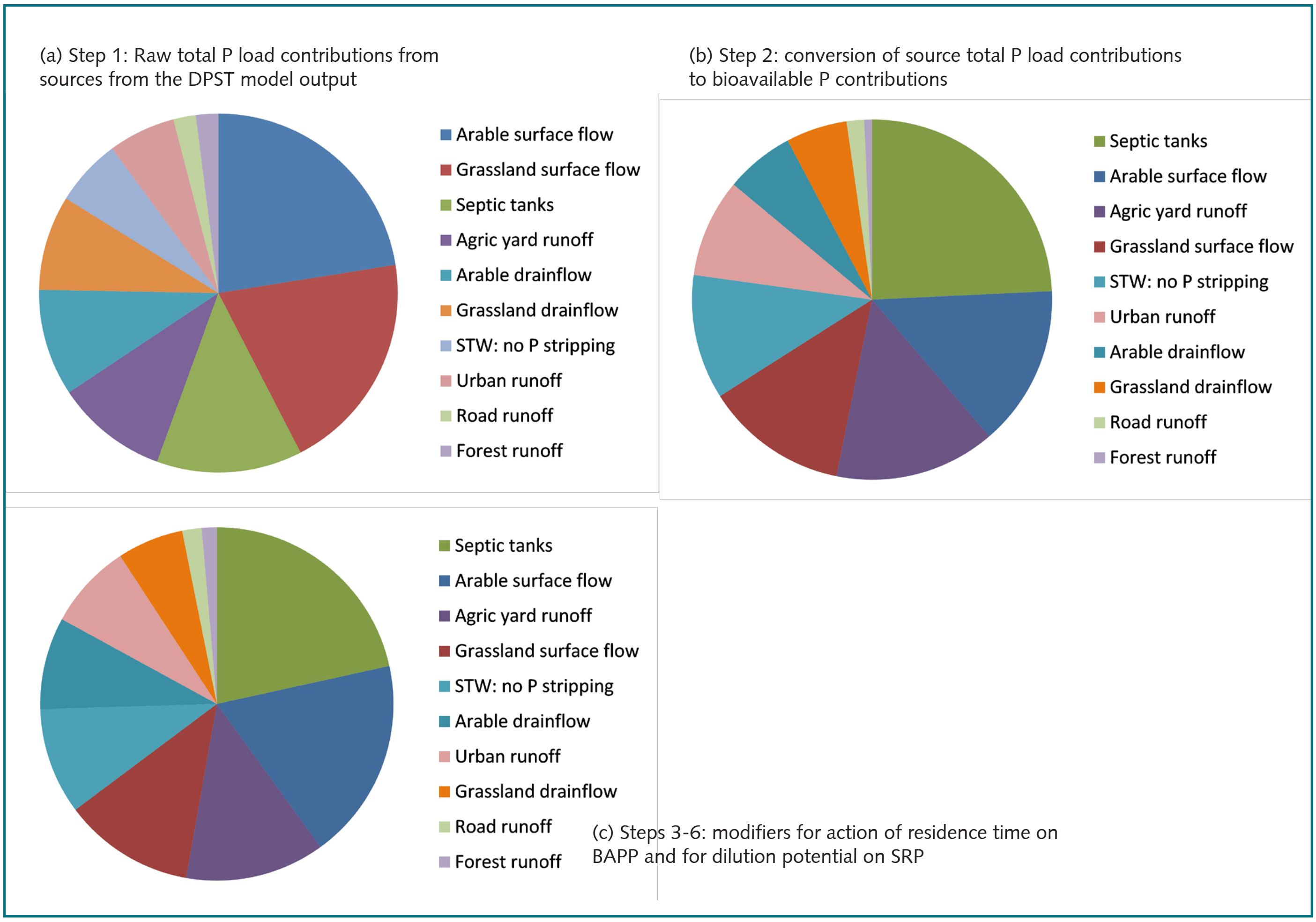


Figure 10. Effects of modifiers on the contributions of sources to P loads going from (a) the raw DPST total P loads, to (b) the bioavailable P loads, to (c) following the interactions with waterbody conditions, according to the steps depicted in Figure 3. In each case the order of the legend reflects decreasing P load contributions.

4.4 Conclusions

This chapter has provided a description of the basis for a scaling methodology for ecological significance of P from catchment sources that can be used alongside current catchment source apportionment models to improve conceptual understanding of the interactions of source and waterbody characteristics in determining P ecological impacts and explaining and predicting ecological responses for catchment P pollution mitigation planning, prioritisation and assessment of improvement post-intervention. This could provide a valuable step in explaining some of the deviations between predictions of ecological improvements made in catchments as a result of the River Basin Management Plans and observations of change in SRP and ecological metrics. The key aspects of the approach are:

- The model links to outputs from existing methodologies like the DPST but could be used as a scaling factor in sequence with other catchment source-transport model outputs as an improved P source-ecological receptor module.

- The model includes a simplistic basis for source chemistry and discharge behaviour with aspects of waterbody condition. This is recognised as fairly basic but provides an important conceptual step in incorporating interactions between waterbody summary factors for residence and internal processing to Pollution source water factors of P concentration, bioavailability and source delivery nature.

- Factors included in the model approach summarise key P impact components described below, bringing the ability to incorporate simplistic ecological principles in P exposure without over-parameterisation:

    o The principle of bioavailability of P from source waters is incorporated via algal assays taken from the literature and such batch assays encompass not only overall P bioavailability, but also the bioavailability of particulate P and co-limitation of other macronutrients (labile C, dissolved inorganic N) in the assays.

    o The principle of source concentration delivery vs in-waterbody dilution is incorporated via a sub-table and negates the requirement for a difficult conversion from pre-modelling inputs (e.g. from the Diffuse Pollution Screening Tool) as source P loads, into concentrations for the impacts determination, then back to loads for comparison with standard model outputs.

    o The principle of timing of pollution delivery relative to ecologically-sensitive summer periods is implied in the delivery nature and negates the need for complex hydrograph division into monthly flows to compare with source delivery.

4.4.1 Recommendations for future work

The next step is that the methodology should be verified by an evaluation of firstly average P concentrations and secondly ecological impacts data across a range of catchments with diverse pollution sources, that have been processed by methodologies such as the DPST, and for which suitable waterbody P monitoring data exist. This may be used to validate and tune the methodology, for example by updating the weightings used in Sub-tables A, B and C.

During this model validation stage there are a number of conceptual factors to address:

- To explore the risk of double accounting of impacts of BFI as DPST does consider this in the relationship derived between observed SRP concentrations for a subset of rivers and the modelled P loads considered exported in the average annual effective rainfall (see point 4 of Box 1).

- Improving the incorporation of factors of scale. The proposed methodology here may, on validation, work better for smaller (first to second order) streams than larger rivers due to increasing weightings needing to be used for impacts of the BAPPslow pool, for example acting through Sub-table B. Downstream reaches of larger river systems may have residence times, and hence retention and realised bioavailability corrections, beyond the current scaling factors used. It may be appropriate to explore procedures for summing bioavailability of the different P sources across a number of contributing headwaters, then modifying retention and dilution effects for downstream impacts on the summed headwater contributions.

5.0 Development of the septic tank P source prioritisation rule base

5.1 Background

The risk of P pollution of surface waters by septic tank discharges depends on a wide range of site- and catchment-based characteristics. Broadly, these are:

- tank size, structure and level of maintenance
- soil and hydrological characteristics of the leach field
- geographical location

The relative contributions of these factors to system failures and consequent pollution problems are discussed below. The data that are potentially available to evaluate them are summarised in Table 32.

Many attempts have been made to quantify the relative importance of the various environmental risk factors that relate to P discharges from septic tank systems. These include a model described by CMHC (2006), which is based on the 'DRASTIC' model proposed by Aller et al. (1987) and tested by Kinsley et al. (2004) and Kinsley & Joy (2006). In outline, the CMHC (2006) model identifies a range of factors that may affect the likelihood of septic tank effluent contaminating waterbodies and enables them to be combined. These factors include system age, soil type, septic tank density, depth to the water table, aquifer conductivity and proximity to surface water. CMHC (2006) subdivided each factor into five levels of risk (where 0 = no risk and 5 = high risk) and gave each a weighting that reflected its relative importance in relation to pollutant delivery (Table 18).

Risk factor	Description	Weighting (% of total)
R_1	System age	30
R_2	Soil type	15
R_3	Septic tank density	15
R_4	Depth to high water table	15
R_5	Aquifer conductivity	5
R_6	Connectivity to surface water	20

Table 18: Relative importance of different wastewater risk factors in relation to pollutant delivery to water (*after CMHC, 2006*).

This approach enables the overall risk value (RISK) associated with individual septic systems, or group of systems, to be calculated by summing the products of risk rating (RISK_RATING) and weighting (WEIGHTING) for each risk factor (1 - n), as follows:

$$RISK = \sum_{1}^{n} (RISK\_RATING_n \times WEIGHTING_n)$$

The risk ratings for each risk factor, as suggested by CMHC (2006) and modified for use in the UK by May et al. (2010), are summarised below.

5.2 Tank factors

5.2.1 Discharge of P to the environment

The amount of P that is discharged is very important in determining the impact that septic tanks will have on that environment. This is determined by multiplying the effluent P concentration by the rate of discharge.

When a septic tank is working efficiently, almost all of the P in the influent wastewater is converted to soluble phosphorus and discharged from the outflow pipe (Canter & Knox, 1985). So, effluent P concentrations are very similar to influent P concentrations, although relative contributions of the different P fractions vary. Effluent TP concentrations recorded in the literature range from about 6 to 20 mg P l^{-1}, with an average of about 10 mg P l^{-1} (see 'Review of septic tank literature': Table 7).

The amount of P discharged to the environment depends on the concentration of P in the effluent and the effluent discharge rate. Because of the nature of septic tanks, i.e. they are sealed systems with an inflow and outflow pipe, the effluent discharges at the same rate waste water enters. This value, which can be approximated by the per capita rate of water usage, is about 144 litres per day in Scotland (as calculated from water demand, household consumption and population figures published by Scottish Water, 2010). If it is assumed that about 10% of household water consumption is lost to evaporation during cooking, laundering and gardening activities, the likely influx of wastewater to a septic tank system is about 130 litres per capita per day. The corresponding discharge per household can be estimated by multiplying this value by the average size of a rural household, as estimated from national statistics derived from census data. These data indicate that the population of Scotland is about 5.2 million, with 960,000 (18%) living in 'rural' settlements (i.e. those with less than 3,000 inhabitants) (http://www.scotland.gov.uk/Publications/2011/09/29133747/2; http://www.scotland.gov.uk/Resource/0041/00417824.pdf), most of which are likely to have septic systems to process domestic waste water. There are 2.37 million households in Scotland (National Records of Scotland, 2011), so the average size of a household in Scotland is about 2.2 people. Using the figures given above, the average volume of domestic waste water likely to be discharged into a septic tank, per household, is approximately 285 litres *per* day. It may be possible to calculate regional figures more accurately if data are available.

5.2.2 Bioavailability of effluent

The P in septic tank effluent is mainly (>90%) soluble reactive phosphorus (SRP), so the bioavailability of the effluent P is high. However, most of this will be adsorbed in the soil soakaway in a properly installed system. Where systems are close to waterbodies or have direct hydrological connectivity to them (e.g. field drains or direct discharges), high concentrations of SRP have been recorded downstream of discharges (e.g. 60 µg l^{-1} Bergfur et al., 2012; up to 400 µg l^{-1} May et al., 2010). This can result in very high in stream P concentrations locally, especially under low flow conditions (Arnscheidt et al., 2007). The biological impacts of such discharges have been examined by Bergfur et al. (2012) but are difficult to resolve and may be greatest where the receiving waters are P limited (Bowes et al., 2007). UKTAG (2008) set P standards for rivers (see Table 19), but these have recently been revised after a consultation in 2012 (UKTAG, 2013).

5.2.3 Size of tank

The size of a septic tank for the population that it serves is critical in terms of determining whether or not it is likely to cause pollution problems. Hydraulic overloading can cause septic tank systems to fail. To function correctly, the storage tank needs to be large enough to achieve a fluid retention time of at least 24 h and to store any accumulated sludge safely for a period of at least 2 years before needing to be emptied (Canter & Knox, 1985). If a tank is too small, wastewater passes through it too quickly for the breakdown processes to work effectively. So, solid material is discharged. This causes hydraulic failure of the soakaway and consequent pollution problems. Under-sized tanks are common, often resulting from of an increase in the size of, or change in the use of, a property without corresponding improvements to the waste water management system. In older systems, increases in water usage due to changes in lifestyle can also overload a septic tank.

It is recommended that each tank has a volume of 2.7 m^3 for up to 4 people, plus an additional volume of 0.18 m^3 per person for each additional user (The Building Regulations 2000). This value is based on the expectation that the tank will be de-sludged annually, as a build-up of sludge reduces the effective volume of the tank. It should be noted, however, that the relationship between tank size and P discharge concentration is unknown and,

Type	Altitude (m)	Akalinity (mg l^{-1} $CaCO_3$)	High	Good	Moderate	Poor
1n	< 80	< 50	19 (13-26)	40 (28-52)	114 (87-140)	842 (752-918)
2n	> 80	< 50	13 (13-20)	28 (28-41)	87 (87-117)	752 (752-851)
3n	< 80	> 50	36 (27-50)	69 (52-91)	173 (141-215)	1003 (921-1098)
4n	> 80	> 50	24 (18-37)	48 (28-70)	132 (109-177)	898 (829-1012)

Table 19: Current P standards to protect WFD ecological status in rivers (UKTAG, 2013). Values now use a site-specific standard, rather than the previous type-specific standard. Note that these are expressed against the previous typologies using a median of the revised standard with possible range in terms of annual mean Soluble Reactive P (Total Reactive P) at a sampling point (µg l^{-1}).

therefore, difficult to quantify. Proposed risk ratings associated with tank size are shown in Table 20.

Table 20. Risk ratings associated with tank size per household (after May et al., 2010)

Size	Risk rating
Small (≤ 2 m^3)	5
Medium (> 2 m^3 - ≤ 3.4 m^3)	3
Large (> 3.4 m^3)	1

5.2.4 Frequency of de-sludging

To function correctly, a septic tank must be de-sludged every 1-2 years. Less frequent emptying can cause solids to build up in the holding tank which, in turn, reduces the effective volume. This causes the tank to overflow and discharge untreated waste. The relationship between rate of de-sludging and P discharge rate and concentration is unknown. Proposed risk ratings associated with rate of de-sludging are shown in Table 21.

Table 21. Risk associated with different de-sludging intervals (after May et al., 2010)

De-sludging interval	Risk rating
> 5 years	5
≥ 2 - 5 years	3
< 2 years	1

5.2.5 Behaviour of households

It is likely that household behaviour has an impact on the amount of P discharged from septic tanks, because the level of P discharge is roughly equal to the level of input. The level of P in waste water is affected by such factors as whether low P detergents are used and the meat content of the diet. Urine and faecal material from vegetarian households may have a 50% lower P content than that from non-vegetarian households (Cordell et al., 2009). The relative source apportionment of P in domestic wastewater is summarised in Figure 1. Information on household behaviour is not available at national scale, so it would be difficult to take these factors into account in a national scale model. However, these values could be varied to explore the possible impacts of different behaviour scenarios.

In addition to the above, while some domestic cleaning products are safe for use in households that are served by septic tank systems, other products – especially those containing bleach – can damage the bacteria that degrade wastes inside the holding tank. This is likely to reduce the efficiency with which the tank breaks down waste material, but the relationship between use of 'septic safe' products and effluent quality is unknown. A proposed risk rating for frequency of use of 'septic tank safe' cleaning products is shown in Table 22.

Table 22. Risk rating for frequency of use of 'septic tank safe' cleaning products (after May et al., 2010).

Use of septic tank safe cleaning products	Risk rating
Never	5
Occasionally	3
Always	1

5.2.6 Secondary treatment of effluent

Secondary treatment of tank effluent can be introduced by householders before the effluent is discharged to the soakaway. Such treatments may involve dosing flocculants such as aluminium, ferric iron, ferrous iron and calcium salts, although the effectiveness of such treatments can be compromised by changes in pH. In some cases, aeration is used to encourage microbial growth and P removal, e.g. as incorporated into biodisc systems. There is no evidence that any of these secondary treatment systems are effective at removing P from the effluent in the long term, with effluent samples collected from simple septic tanks and package treatment plants (PTPs) all appearing to have similar P concentrations (May *pers. comm*., Brownlie *pers. comm*., Environment Agency *pers. comm*.). As it is assumed that the effluent from PTPs is less polluting than that from ordinary septic tanks, they are more likely to discharge directly to water courses than standard septic tanks (Tobin, 2012). More recent evidence suggests that this assumption may be flawed (May et al., 2014).

5.2.7 Seasonality of discharge

Because septic tanks depend on microbial breakdown to process wastewater effectively, it is very unlikely that they function well if the level of use is not relatively constant. So, tanks that are only used seasonally, such as those associated with holiday accommodation, caravan sites and rural visitors centres, are unlikely to function effectively. The impact of seasonal use on septic tank effluent quality is unknown.

5.2.8 Tank receives roof runoff

Many older septic tanks within the UK receive roof runoff in addition to standard domestic wastewater. This increases the hydraulic load to the tank significantly and reduces the waste retention time and level of processing. Whether or not a septic tank system receives roof runoff is an important factor in determining whether the system is likely to overflow due to hydraulic failure (Canter & Knox, 1985). Systems that receive roof runoff will almost certainly overflow during heavy rainfall, in contrast to those that do not receive roof runoff. The effect of this on effluent quality and quantity is unknown, but risk ratings have been suggested in relation to this. These are shown in Table 23.

Table 23. Risk ratings associated with septic tanks receiving roof runoff (after May et al., 2010).

Receives roof runoff?	Risk rating
Yes	5
No	0

5.2.9 Type of installation

Older septic tanks are often less well designed than newer systems and, as such, function less efficiently. The design life of many older systems was probably only about 10-15 years when they were originally installed (Canter & Knox, 1985), but many have been in constant use for much longer (Fildes, 2011). In contrast, most recently installed systems have been constructed from stronger and more watertight material that should last for up to 50 years (Canter & Knox, 1985).

There are many types of septic tanks. Older ones are likely to be built of brick or concrete. More recently installed tanks are likely to be made from glass reinforced plastic (GRP) or polyethelene. The age and type of installation may affect the P content of the effluent discharged. CMHC (2006) suggest that the relative risk of failure of septic systems increases with age, with systems over

30 years old being up to 12 times more likely to cause water pollution problems than those less than those less than 10 years old. However, there are no national level data available on the age and type of existing installations and the specific effects of tank design and construction on effluent quality and quantity are unknown and cannot be quantified.

5.2.10 Level of maintenance/integrity of tank

Level of maintenance affects the integrity of the tank and probably has a high impact on the amount of P discharged to the environment. This is because badly maintained systems leak untreated or partially treated effluent. Leaks can occur as a result of serious structural damage, such as large holes or cracks, or more minor problems, such as small cracks or broken seals. The effect of this on effluent quality and quantity is unknown but risk ratings have been suggested in relation to these problems and are summarised in Table 24.

Table 24. Risk ratings associated with condition of tank (after May et al., 2010)

Condition	Risk rating
Cracked; broken	5
Small crack; broken seal	3
Watertight; in good repair	1

5.3 Leach field factors

5.3.1 Direct connection to watercourse

The level of hydrological connectivity to a watercourse, including a ditch or field drain, can have a very large impact on the amount of P that is likely to be delivered to water. In tank systems that are installed correctly, most of the P that is discharged to the soil soakaway seems to be adsorbed by the soil within a few metres of the point of discharge (May et al., 2010). However, the amount of P from systems with increased hydrological connectivity, including tanks that discharge directly to water, is probably much higher. At present, the relationship between hydrological connectivity and P delivery to water is unclear, except in systems that discharge directly. The number, or proportion, of direct connections varies from catchment to catchment. However, it has been suggested that values range from 4% (Fildes, 2011), through 17% (Dudley & May, 2007) to 50% (Tobin, 2012) with an average of about 34%. Most systems that discharge directly to water are PTPs. It has always been assumed that PTPs discharge much lower amounts of P than simple septic tanks. However, recent monitoring data suggest that this is untrue (May et al., 2010; Browlee *pers. comm*.).

5.3.2 Age of leachfield

The age of the septic tank system also affects the capacity of the soil within the drainage field to adsorb P from the discharged effluent. This is because, after many years in the same location, soils can become saturated with P making them less able to remove P from the effluent that they receive. The older the installation, the more likely the soakaway is to be affected by this problem. The approximate age of a property is a good indicator of the age of the septic tank and soakaway in most cases. The extent to which the age of the soakaway affects the P saturation of the soil and, consequently, the degree of P removal from the effluent is discussed under 'P sorption and leaching characteristics of soil', below. In summary, there is an increased risk of P loss once the degree of P saturation of the soil (DPS_{Ox}) exceeds 20%. In addition, it should be noted that very young leachfields may have a reduced ability to retain P until a fully functioning biomat has been formed. Risk factors associated with system age are shown in Table 25.

Table 25. Risk ratings associated with system age (after May et al., 2010).

System age (years)	Risk rating
< 10	0.4
10-29	2.1
≥ 30	5

5.3.3 P sorption and leaching characteristics of soil

Septic systems need to be located on suitable soil types, ideally well drained sandy soils (Canter & Knox, 1985). Other types of soil, such as gravel, cobble or clay, are much less suitable for use as drainage fields because they drain either too quickly or too slowly for effective pollutant removal to take place (Canter & Knox, 1985).

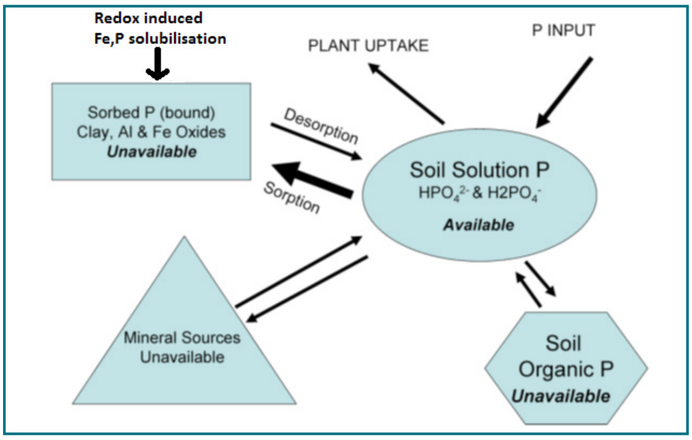


Figure 11. Phosphorus reaction processes in leachfield soils (*after Sinclair et al.*, 2012)

The velocity of transport of P, both in the unsaturated (vadose) zone constituting the leachfield and in the groundwater zone where lateral transport away from the leachfield towards surface water occurs, is strongly retarded compared to water transport in suitable soils. The magnitude of this retardation depends on chemical precipitation, soil pH, the adsorptive surfaces present, and the redox potential of the leachfield and the groundwater plume. Figure 11 (after Sinclair et al., 2012) shows the nature of the reactions that bring about this retardation.

A simple way of estimating the retardation is to assume a linear relationship between the soil solution concentration and the adsorbing surfaces but, unfortunately, the sorption process is highly non-linear and chemical precipitation also has a significant impact on P retention. Soil solution chemistry modelling to achieve such predictions is highly complex and would need to be underpinned by a detailed research and development project. So, as an alternative, we propose to utilise the "change point" concept to identify the amount of effluent that can pass through a leachfield before significant leaching begins (McDowell & Sharpley, 2001; Nair et al., 2004). This concept defines the degree of P saturation as follows:

$DPS_{Ox} = [(P_{Ox})/(Fe_{Ox}+Al_{Ox})] \times 100\%$ (Equation 1)

where:

DPS_{Ox} = Degree of P saturation (%)
P_{Ox} = Oxalate extractable P
$Fe_{Ox}+Al_{Ox}$ = Oxalate extractable Fe and Al

Nair et al. (2004) found that P availability for leaching increased rapidly once DPS_{Ox} exceeded 20%.

Table 26. Assumptions used to estimate the time until significant P leaching occurs from a new, well-functioning vertical leachfield.

Leachfield area per person	10	m^2
P input per person	500	g/year
Design depth of leachfield	1	m
Soil density	1000	kg/m^3
P saturation fraction at which P leaching starts	20%	

Soil association	P sorption potential (mg P/kg soil)	Time to P loss from leachfield		Risk factor
		Years	Discounted impact	
Millbuie	1434	5.7	0.678	4
Minto	1604	6.4	0.648	4
Auchenblae	1811	7.2	0.613	4
Darvel	2077	8.3	0.570	3
Thurso	2244	9.0	0.545	3
UndifAlluvium	2259	9.0	0.543	3
Kintyre	2289	9.2	0.538	3
Alluvialsoil	2316	9.3	0.534	3
NorthMormond	2343	9.4	0.530	3
Eckford	2420	9.7	0.519	3
Ordley	2455	9.8	0.515	3
Sorn	2460	9.8	0.514	3
Stirling	2578	10.3	0.498	3
Arkaig	2585	10.3	0.497	3
Strichen	2626	10.5	0.491	3
Kippen	2642	10.6	0.489	3
Forfar	2650	10.6	0.488	3
Mountboy	2657	10.6	0.487	3
Balrownie	2674	10.7	0.485	3
Tynet	2682	10.7	0.484	3
Countesswells	2693	10.8	0.482	3
Tomintoul	2701	10.8	0.481	3
Corby	2705	10.8	0.481	3
Kilmarnock	2785	11.1	0.471	3
Boyndie	2787	11.1	0.470	3
Rowanhill	2883	11.5	0.458	3
Foudland	3013	12.1	0.442	3
Panbride	3165	12.7	0.425	3
Ettrick	3195	12.8	0.421	3
Yarrow	3458	13.8	0.392	2
Tarves	3864	15.5	0.351	2
Sourhope	4469	17.9	0.298	2
Insch	6038	24.2	0.195	1
Darleith	6451	25.8	0.174	1

Table 27: Estimated time to P loss from the leachfield based on the P sorption capacity of the Scottish soil series, as determined according to content of oxalate extractable Fe + Al (data derived from Sinclair et al., 2012).

Sinclair et al. (2012) recently carried out an assessment of the P leaching risk from the main agricultural soil associations of Scotland. Using the assumptions given in Table 26, we have estimated an indicative time for significant P leaching to begin from a well-functioning, vertical leachfield on Scottish soils (Table 27).

5.3.4 Discounting of mitigation impacts

Because P is a conservative element, its long term attenuation in septic tank leachfields is likely to be zero even if it is precipitated and/or strongly adsorbed. In environmental economics, the usual practice for dealing with deferred benefits from expenditure is to assume a discount rate for those benefits against which initial costs of implementation can be compared. Using this approach in the context of retardation of a P plume, we can identify an appropriate discounting rate to represent the benefits of mitigation by considering the delay in improvements that would occur if a mitigation measure was adopted. For example, adoption of a mitigation measure that recovers P from the septic tank would result in a step change in P levels exported to the leachfield. However, the benefit of that mitigation measure would not be felt until it had resulted in a step change in outlet SRP. We can therefore consider the net benefit of mitigation of 1 unit of P as equal to:

$$\text{Unit NPB} = 1/(1+i)^{T^{sw}} \qquad \text{(Equation 2)}$$

where:

Unit NPB = discounted benefit of mitigation of 1 unit of input to the leachfield;

i = discount rate (taken as 7%);

T^{sw} = Retardation time from Table 27.

If the leachfield is functioning correctly, the travel times in Table 27 will, effectively, strongly discount the environmental impact of the P leaching. Taking Risk factor 5 as connoting zero delay and, therefore, zero discounting of impacts, and Risk factor 1 as connoting a Unit NPB of < 20% with a discount rate of 7% per year, Equation 2 allows a semi-quantitative Risk factor for a well-functioning leachfield soil to be determined. Table 27 gives a proposed categorisation of these risks. For soil series and sites not represented, it is proposed that the same relationship between P sorption potential and risk category could be used as demonstrated in Table 28.

Table 28. Relationship between P sorption potential and risk category for soil series not shown in Table 27.

P sorption capacity (mg/kg)	Risk factor	Example soil association
> 6000	1	Darleith, Insch
3400 -6000	2	Sourhope, Yarrow, Tarves
1800 -3400	3	Rowanhill, Countesswells
900 – 1800	4	Auchenblae, Minto, Millbuie
<900	5	?

5.3.5 High or seasonally variable water table

The height of the water table is an important factor in determining the likelihood that P discharged from septic tank systems will pollute water. In Scotland, planning regulations require a minimum of 1 m of undisturbed soil between the base of the percolation trenches and the bedrock below or the highest level of the water table (Scottish Executive, 2001). Although there are no site-specific data available on the height of the water table, Hydrology of Soils Types (HOST) data (Boorman, 1995) can be used to indicate likely water table heights in different geographical areas. For sites where the water table height is likely to be less than 1 m below the base of the percolation trenches, it should be assumed that soil uptake of P is minimal and that all of the effluent P discharges to groundwater. In newer systems, at least, distribution pipes are likely to have been laid at least 500 mm below the soil surface (The Building Regulations, 2000) so the water table needs to be at least 1.5 m below the soil surface for the soakaway to function correctly.

5.3.6 Soil hydraulic properties

The hydraulic properties of the soil soakaway are very important in determining the rate of uptake of P from the tank effluent before it reaches a waterbody. This is recognised in the installation requirements for septic tank systems, as controlled by UK building regulations (The Building Regulations 2000). In summary, these require the leach field to have certain hydrological properties that are assessed during soil percolation tests. Small soakaways are designed to work effectively even with a rainfall rate of 10 mm in 5 minutes, which is assumed to be the worst case scenario for a storm event with a 10-year return cycle. It should be noted, however, that only newer systems will have been installed under these regulations.

There are no national scale data available to provide the results of soil percolation tests for individual installations. However, soil hydraulic properties can be derived from HOST Classes data (Boorman et al., 1995) or the baseflow index (BFI). Both of these datasets are available at 1 km^2 resolution.

Table 29. Risk ratings for impact of soil type and hydrological characteristics (HOST class) on likelihood of contamination of nearby waterbodies by septic tank effluent (after May et al., 2010).

Risk rating	HOST classes
5	9, 10, 12, 13, 14, 18, 19, 20, 21, 22, 23, 24, 25, 27
3	4, 6, 7, 8, 11, 15, 17, 26, 28, 29
1	1, 2, 3, 5, 16,

Alternatively, a risk index for P discharge from the leachfield to surface water (E) could be estimated as follows:

$$E = (1\text{-}BFI) \times (1\text{-} E_{bypass}) + BFI \times (1\text{-} E_{matrix}) \qquad \text{(Equation 3)}$$

Where:

BFI = base flow index = ground water flow/excess rainfall for each soil or 1km^2;

E_{bypass} = efficiency of removal when the absorption capacity of the leachfield is bypassed (e.g. due to high water table, biomat failure, high inputs of water from rain/floods, rapid pathways for shallow groundwater flow, such as tile drains etc);

E_{matrix} = efficiency of removal as the leachate travels through the soil matrix.

For even a modest setback distance of a leach field from a surface water (e.g. 12 m, the median value of the range of setback distances in the high risk category of Table 30), the saturated zone travel time is likely to be an order of magnitude greater than the travel time over 1 m of vadose zone leachfield, shown in Table 27. Consequently, discounting of the impact using Equation 2, it is reasonable to assume that the discounted impact of P transported through the soil matrix in the groundwater plume, is close to zero and, hence, that E_{matrix} is almost 100%. In contrast, when bypass routing occurs it seems reasonable to assume that the travel time of P via this route is likely to be short, with the net present benefit of its mitigation high and the retardation low. It is, therefore, suggested that a reasonable assumption would be $E_{bypass}=0$. This means that the efficiency of P removal in the groundwater plume is simply the value of the BFI. As part of other water balance modelling projects, the BFI has been estimated for the dominant soils/HOST classes for each 1 km^2 of Scotland. As this database forms part of the existing screening tool (SNIFFER, 2006), this index is readily available and its values could be seasonally adjusted to allow for more rapid transport in winter months, if required.

5.3.7 Soil redox

Robertson et al. (1998) found that, in reducing zones, P concentrations in groundwater plumes were buffered to moderate concentrations whereas, in oxidising zones, greater contrasts in P concentrations were exhibited. However, the enhanced concentrations of ferrous iron being transported mean that when a reduced plume is exposed to oxygen (as near discharge points), P precipitation and adsorption will be strongly enhanced, for example adsorbing on ochre deposits emerging from waterlogged soils. It is rarely the case that soil redox conditions can be predicted with any certainty but, given the potential for released iron compounds to re-precipitate, it does not seem appropriate to distinguish sites based on redo without more substantial geochemical evidence and modelling.

5.3.8 Soil pH

Soil pH has the potential to be strongly influenced by the oxidation of ammonium and organic compounds in the leachfield, which releases acidity (Robertson et al., 1991, 1998). Where soils are well buffered (e.g. calcareous soils) this will not necessarily alter P sorption properties too strongly. However, if a pH decrease occurs, this will lead to decreased P concentrations because strengite and variscite precipitate P more efficiently at low pH. If soils remain at a high pH (> 7.5), calcium phosphate minerals will limit P solubility. In view of these considerations, we propose to increase the risk factor in Tables 27 or 28 for well-buffered soils with neutral pH (5.5-7.5), to reduce the risk factor by 1 for initially acid (pH < 5.5) or initially alkali (pH > 7.5) soils, and not to modify the risk factor for poorly-buffered (non-calcareous, loamy sand) soils with a neutral pH. The maximum for the soil risk factor (RF_{soil}) would remain at 5.

The soil risk factor is then applied to the proportion of P applied to soil by calculating a factor which is $RF_{soil}/5$.

5.3.9 Organic matter

The presence of organic matter in soils may modify the potential for adsorption of P onto iron and aluminium sesquioxides. For example, Maguire et al. (2001) found a relatively small impact of organic matter on the sorption of P by acidic soils in N. Ireland. Kang et al, (2008) found a much lower slope relating P sorption maxima to organic matter content in soils above about 50 mg kg-1. However, as most septic tank systems are designed to function using subsoils, where organic matter contents are much lower, it is probably not necessary to alter the estimates of P sorption capacity set out in Table 27, except in soils with very high organic matter. Here, the impact is principally one of water retention and its influence on hydrology, which is covered by the HOST classification and by the risk factor for high or seasonably variable water table described above.

5.3.10 Distance to waterbody

Septic systems that are close to waterbodies are more likely to cause pollution problems than those that are situated further away. Older installations are more likely to cause contamination of waterbodies at greater distances than newer installations, because the effluent plume moves towards a downstream waterbody at a rate of about 1m per year (Robertson, 2003). Within the UK, regulations set minimum distances of 10 m from a watercourse or permeable drain, 50 m from a groundwater supply abstraction point, and 15 m from a building (The Building Regulations, 2000).

Where the OS grid reference locations of tanks and leach fields are known, i.e. for registered systems, it is possible to determine the distance from a watercourse or drain if drainage network data are available at sufficiently high resolution. However, registered systems comprise only 10% of all tanks. The exact locations of the remainder are unknown, but many can be located approximately using postcode data and sewered area maps (May et al., 1999; May et al., 2010). It is unlikely, however, that older systems will have been installed in compliance with the regulations outlined above. Risk factors associated with distance from a watercourse are proposed in Table 30.

Table 30. Risk rating for impact of proximity to a watercourse on contamination of waterbodies by septic tank effluent (after May et al., 2010).

Distance from waterbody	Risk rating
0 - < 25 m	5
25 - < 100m	4
100 - < 250m	3
250 - < 500m	2
≥ 500 m	1

5.3.11 Slope

Slope affects the way that the drainage field functions. It has been suggested that septic systems should only be sited where the slope of the land is less than 20%, and preferably less than 5%, because this affects the percolative (transmission through) and infiltrative (inflow) capacity of the soil (Cotteral & Norris, 1969). Cotteral and Norris (1969) also recommended that systems should be placed on a concave rather than convex slope, whenever possible, and that they should not be sited at the base of a slope or in areas that are subject to seasonal flooding. For modelling purposes, slope can be determined from a 50m resolution digital terrain model (Morris & Flavin, 1990; 1994). However, the relationship between slope and P delivery to lower parts of the catchment is unknown and requires further research. A range of risk factors associated with the slope of the terrain are suggested in Table 31.

Table 31. Risk rating for impact of slope on contamination of waterbodies by septic tank effluent (after May et al., 2010).

Slope	Risk rating
0	1
> 0 - < 5%	2
5% - <15%	3
15% - <25%	4
≥ 25%	5

We combine the risk factors associated with proximity to watercourse and slope (Tables 30 and 31) by taking the higher value of the two risk ratings. This we term the riparian risk factor $RF_{riparian}$. This is then applied to the proportion of P applied to soil and transported during high flows by calculating a factor which is $RF_{riparian}/5$. High risk sites thus show no attenuation of P in flows above the BFI, but low risk sites show significant attenuation of P at BFI.

5.3.12 Hydraulic gradient

Hydraulic gradient affects the flow path of the water, and its related solute and particulate load, through the catchment. This can be derived from slope using available hydrological models but, in some areas, accuracy may be affected if flows are augmented by upslope water sources such as springs.

5.4 Conclusions

This chapter has explored the risk factors associated with septic tanks that determine their impact on the environment in terms of P pollution. These are:

- Tank size, structure and level of maintenance
- Soil and hydrological characteristics of the soil soakaway
- Geographical location

Their relative importance has been evaluated and methods of estimating them from readily available data have been considered. In relation to the above, Figure 12 illustrates how the potential delivery of septic tank P discharges to water could be incorporated into an updated DPST model. Data availability is a key issue in this process. This is especially true of site specific factors, such as system structure and maintenance, and household behaviour, which affect the level of P discharge from the tank itself and for which there are no readily available data sources (Table 32). For this reason, these risk factors have not been incorporated into the proposed model development. Instead, annual discharges of P and water from an average tank have been estimated, based on published values. It is proposed to include these to estimate the P discharged from each tank to the environment.

The main focus of the proposed improvement to the DPST model has been on the factors that affect P retention in the soil soakaway and can be estimated from readily available data (Table 32). These are summarised in Figure 12. They include the effect of the local hydrology on P transport; the effect of soil P characteristics on P retention; and the likelihood of tanks discharging directly to water, thereby bypassing the soil soakaway system which would, otherwise, provide the last line of defence between the tank discharge and the receiving waterbody.

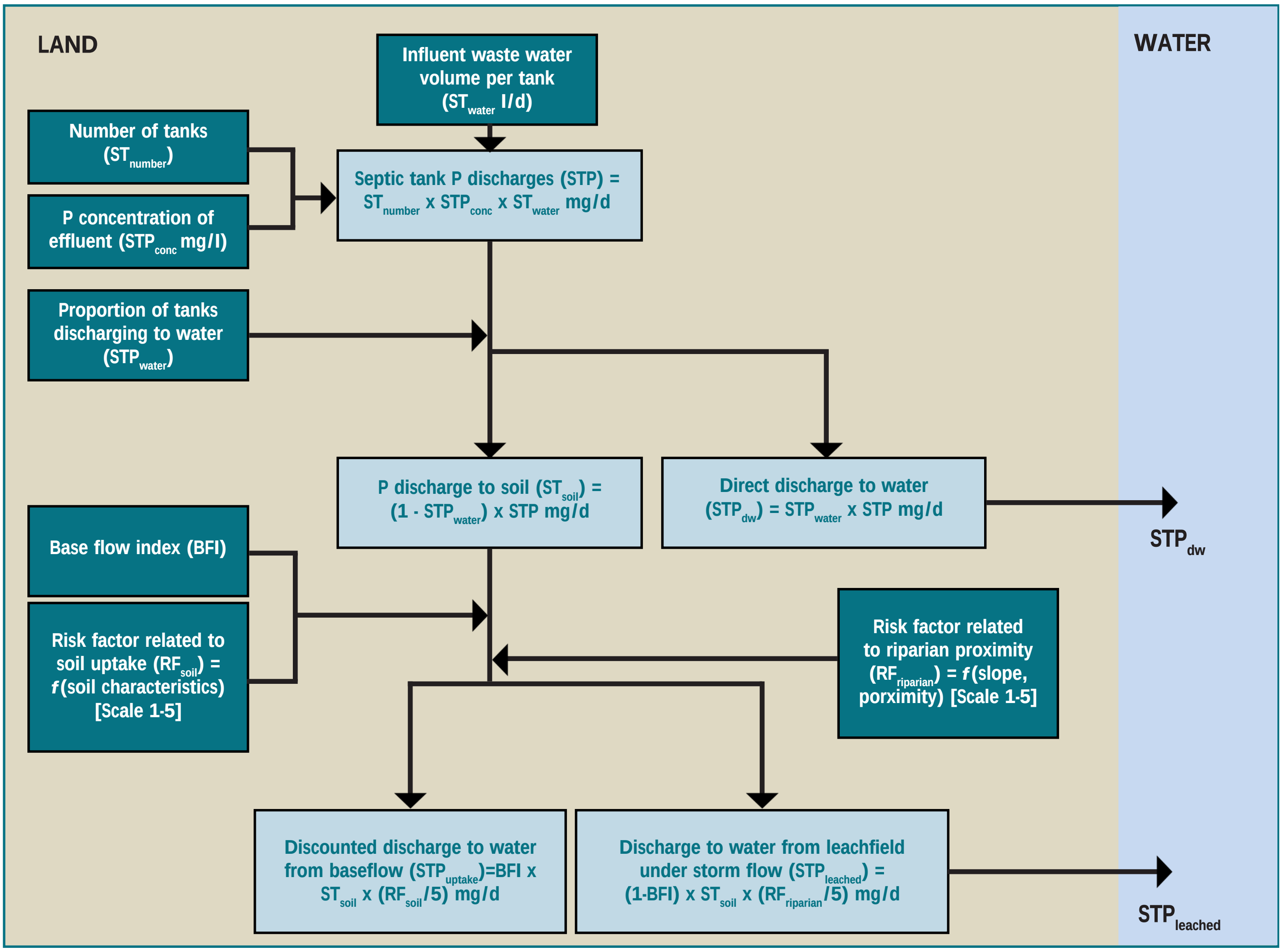
System component	Factor affecting P discharge to water	Level of importance	Can it be incorporated into model?	Data available	Level of uncertainty
Tank	Discharge concentration	Very high	Yes; use constant discharge value (with error) for all tanks	(i) Effluent concentration about 10 mg P l-1 (ii) Effluent volume about 130 litres per capita (iii) Average size of rural household = 2.2 people	(i) Range 6.3-19.6 mg P l-1 (ii) Unknown (iii) Unknown
Tank	Bioavailability of effluent P	Low	No	No data; assume > 90% of effluent P is soluble reactive and has high bioavailability (Canter & Knox 1985)	Unknown
Tank	Size of tank	Medium	No; could use constant value	Tank capacity = (180* (number of people - 4)) + 2700 litres (The Building Regulations 2000)	Unknown
Tank	Frequency of de-sludging	Unknown. Affects processing efficiency and accidental discharge of solids.	No	No data on frequency of emptying at national level.	Unknown
Tank	Household behaviours (product use, diet, etc.)	Medium	No	No national level data; could be derived from sewage P source apportionment data; recent changes in P concentrations in cleaning products need to be addressed.	Unknown
Tank	Secondary treatment (in tank/before discharge to soil)	Unknown	No	No data at national level; impact on P discharge unknown.	Unknown
Tank	Seasonality of use/ discharge	Variable; may be important in some cases	No	No national or site level data; effect of seasonal use on effluent quality unknown.	Unknown
Tank	Tank receives roof runoff	Very high; results in high flushing and discharge rates	No	No data; effect of increased flushing on effluent quality unknown.	Unknown
Tank	Installation type	Low	No	No data available at national level; could be derived from age of house, if known, in most cases.	Unknown
Tank	Level of maintenance; integrity of tank	High; badly maintained tanks leak	No	No data available at national level, but > 90% likely to be badly maintained.	Unknown

Table 32: Factors affecting P discharges to water from septic tank systems and possible sources of data to support the modelling of P delivery from these sources.

Table 32 continued: Factors affecting P discharges to water from septic tank systems and possible sources of data to support the modelling of P delivery from these sources.

System component	Factor affecting P discharge to water	Level of importance	Can it be incorporated into model?	Data available	Level of uncertainty
Leach field	Direct connection to watercourse	Very high	Yes	Use average number of direct connections = 34%	17% (Dudley & May, 2007 - Scotland) to 50% (Tobin, 2102 - England)
Leach field	Age of leach field	Medium; affects biomat development & soil P uptake	No	No detailed data available at national level. Derive from age of property; enhanced risk once DPS_{Ox} for 1m of leachfield exceeds 20%. Low P loss for newer leachfield (cf. Table 27).	Unknown
Leach field	Low P sorption soils	High	No	No data available. Use risk table provided (Table 30).	Unknown
Leach field	Soil hydraulic properties	High	Yes	National level data could be derived from HOST classes (Boorman et al., 1995) and BFI – both available at 1 km^2 resolution. Use BFI as proxy for efficiency of P removal	Unknown
Leach field	High or seasonally variable water table	High	Yes	HOST Classes data can provide information on likely water table height at 1km^2 resolution (Boorman et al., 1995). Use BFI as proxy for efficiency of P removal	Unknown
Leach field	Soil redox	High; governs sorption reactions	Yes; assume constant?	Don't use risk modifier for this parameter	Unknown
Leach field	Soil pH	High; precipitation reactions with Ca, Al controlled by pH	Yes; assume constant?	Risk factor 3: well buffered soils with neutral pH (5.5-7.5) Risk factor 2: initially acid (pH < 5.5) or initially alkali (pH > 7.5) soils Risk factor 2: poorly buffered (non-calcareous, loamy sand) soils with a neutral pH.	Affected by NH4 reactions
Leach field	Organic matter	Yes, limits P sorption and maintains soil moisture	Yes; assume constant?	No modifier as subsoils usually used.	Unknown
Location	Distance from waterbody	Yes	Yes	Minimum setback distances enforced by The Building Regulations, 2000. Use OS locations of tanks and leach fields for registered systems (i.e. about 10% of tanks); derive remainder from postcodes and sewered area maps (May et al., 1999; 2010).	Medium uncertainty; may not apply to older systems. High uncertainty if using data with low resolution.
Location	Slope	Medium	Yes; simple surrogate for hydraulic gradient.	Digital terrain model at 50m resolution (Morris & Flavin, 1990, 1994).	Poor if low resolution
Location	Hydraulic gradient	Medium	Yes	Derive from slope.	May be affected by upslope water sources, e.g. springs.

Figure 12: Diagram of how risk factors associated with potential delivery of septic tank P discharges to water could be evaluated and incorporated into an updated DSPT model (see text for details).



6.0 Overall conclusions

SEPA make regular use of the Diffuse Pollution Screening Tool (DSPT) for apportioning of sources within catchments and identifying targeted approaches to mitigation of P loads. However, as it stands, this approach is open to challenge on the basis that not all P loading has the same ecological significance, due to the form or timing of the P loading. We have developed a simple methodology for identifying the most ecologically significant sources of phosphorus in a river waterbody catchment. Whilst this methodology is transparent and fully explained within this report, it remains to be tested and hence validated. The methodology has involved assessing the transport of P to running waters, the bioavailability once in the water, and appropriate seasonal and spatial adjustment of effects on ecology. The use of normalised factors to adjust the source strengths has the potential to allow adjustment of mitigation targets more effectively to reduce soluble P concentrations and diatom impacts.

The report shows that sources vary widely in the proportion of P which is soluble (from >80% for septic tanks to <20% for forest and arable crop runoff). Moreover the high particulate P content of sediment loads in wet periods has a lower bioavailability. If sediments do not release P during the biologically active period, then catchment measures aimed at controlling agricultural P loading will not have the desired effect of reducing eutrophication risk, even though such measures may control sediment transfer effectively. However, there is evidence for some of the P in sediment sources undergoing rapid desorption following re-suspension and dilution. If this occurs in winter months the soluble P released will not strongly impact ecology, and in summer months the sediment still retained in streams may act as a P sink, especially for significant septic sources upstream.

The review of septic tank literature suggests that unless site specific detail is available, it can be assumed that most tank effluent contains about 10 mg P l^{-1} when it is discharged and that the load from rural households will average about 1.3 kg P $year^{-1}$. Where septic tanks are discharging to a soakaway/leachfield, biomat bacteria and microorganisms provide much of septic tank effluent treatment in the soakaway, including some P immobilisation. The ability of soils to immobilise P in septic tank effluents is probably more significant, and can be ranked as follows: Fine-grained, non-calcareous > Coarse-grained non-calcareous > Fine-grained calcareous > Coarse-grained calcareous. Phosphorus plumes from septic tanks tend to travel along downslope gradients by about 1 m per year. As most septic tank systems (STS) remain in place for more than 25 years, the recommended (UK) safe setback distance of 10 m is probably inadequate. This is compounded by the wide occurrence of improper hydrological installation of STS. Hence our approach to assessing the impact of siting on septic tank loading to water is to discount according to timescales of STS impact and recovery when mitigated. Using a P leaching risk framework developed in another project (Sinclair et al., 2012), we devised a flow chart methodology for assessing the attenuation of STS impacts for P delivery, as a function of discharge route (direct or leachfield), soil type, Base Flow Index, slope and proximity to water. This can be applied to the DPST estimates of septic tank loads.

Systems for determination of concentration/exposure/duration are gaining favour as a means of linking pollutant concentrations to impacts. These also need to take into account scale, as released P may impact on slower downstream reaches where more retention occurs than the reach where the P was released. Combining source and waterbody characteristics is also important. One reason for this is that single chemical attributes (e.g. SRP concentrations; on which the current framework relies heavily) do not necessarily link with diatom responses in streams and rivers as well as other key co-contaminants do (such as fine sediments and other nutrients). An additional interaction is that source and water body attributes influence ecological exposure to P (and variably between different P forms), but also timescales of recovery following mitigation.

The existing DPST procedure for estimating impacts on SRP calculates a perfect mixer average concentration (PMAC) in which flow weighting (using HOST class) of loads from diffuse sources are combined with invariant loads from point sources. We therefore need to be careful not to double account for the differential pattern of release, although weighting for ecological significance is still appropriate. For this reason, in the presented case study of our ecological P scaling procedure to a small catchment we chose to make use of the raw TP loads/ water body dataset in the DPST, not the PMAC. There is some further development needed to finalise a working procedure for ecological P exposure that can be applied to current source apportionment methodologies, as well as in calibration of some scaling factors in the approach.

The conceptualisation of P pools in the project distinguishes TP = Total P; PP = Particulate P; TDP = total dissolved P; SRP = Soluble reactive P; DUP = dissolved unreactive P; AATP = Algal-available total P, which we equate with BATP = biologically active total P; AAPP = algal-available particulate P, which we equate with biologically active particulate P, $BAPP_{fast}$; and algal non-available particulate P, which we equate with slowly available biologically active particulate phosphorus, $BAPP_{slow}$, only effective in downstream reaches (under conditions of greater residence times). The TP loads from the DPST are adjusted to BATP according to the BATP/TP ratio derived from the literature for different sources, then corrected for residence time, dilution potential by the water body characteristics (energetic high flows with low baseflow vs larger dilution potential and sustained summer flows), catchment (high vs low slope) and source characteristics (rainfall dependent discharge vs constant discharge). The additional water body data for this adjustment are the base flow index (BFI) and average slope, both of which are readily obtained.

The outcomes of this CREW ECO-P methodology still need to be tested against catchment data, for both first order (e.g. Tarland) and non-first order streams. This would also allow a treatment of uncertainty, and the generation of a risk-based approach to compliance with appropriate UKTAG standards. DPST loads are available for Scottish WFD river water bodies both for the whole catchment upstream after allowing for retention and for the local waterbodies. Methodology for retention calculations in complex catchments has improved since DPST was constructed, so we suggest it would be better to build up a retention picture based on the local catchment loads and more recent retention methodology. Further work is also needed to explore the risk of double accounting where retention is considered in complex catchments. It should also be noted that the proposed methodology can only be applied to the existing DPST database, not any updates based on further process understanding or data availability.

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