Natural England Commissioned Report NECR170

The impact of phosphorus inputs from small discharges on designated freshwater sites

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Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Background

Nutrient enrichment of freshwater SSSIs is a widespread problem, with the majority of river and lake SSSIs in England affected. A range of measures have been taken to reduce this problem, many of which are ongoing. These have mainly focussed on reducing inputs from sewage treatment works and agriculture. For some time concerns have been raised about the potential role played by small domestic discharges, such as septic tanks, but it is only relatively recently that this issue has received significant attention.

To help address this issue, Natural England, with a contribution from the Broads Authority, commissioned the Centre for Ecology & Hydrology (CEH) in 2009 to conduct a review of the potential risk posed by small domestic discharges, such septic tanks, to freshwater SSSIs. This focussed on the risk of phosphorus (P) pollution to sites that are vulnerable to hyper-eutrophication.

The overall aims were to assess whether significant risks were likely to occur, either at a wide scale or under local circumstances and, if they were, to suggest approaches for refining the risk assessment across the freshwater SSSI series. In addition, CEH were asked to consider the options available for reducing the impact of P pollution from septic tanks where necessary. To this end, the study was divided into two main components:

- To review appropriate literature relevant to this issue.
- To conduct two desk-based case studies, in the River Avon SAC catchment and The Broads, to estimate the number of dwellings not on mains sewerage and to assess the possible P pollution arising from these.

The findings and recommendations from this work are contained within this report. The majority of the work was conducted in 2009, with updates in 2013 for certain topic areas such as:

- de-sludging of tanks to reduce P discharges;
- the effectiveness of reedbeds and wetlands to clean effluent; and
- the current status of registration schemes in England, Wales and Scotland.

Whilst some aspects of this report may have been superseded by more recent studies and developments, it still represents one of the most thorough reviews undertaken on this subject in the UK. As such, Natural England considers that it is worth publishing. However, the case studies in Chapter 6 make the precautionary assumption that all of the P estimated to be produced by septic tanks reaches the watercourse. More recent work undertaken by CEH for Natural England (NECR171) strongly suggests that this assumption is over-precautionary.

The work contained within this report has allowed Natural England to develop a focused research strategy to help better understand the risk posed by septic tanks to freshwater SSSIs. It has been used by the Environment Agency and Defra as a key source of information on this topic during a 2012 Public Consultation on septic tanks.

This report should be cited as:

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Further information
This report can be downloaded from the Natural England website: **www.gov.uk/natural-england**. For information on Natural England publications contact the Natural England Enquiry Service on 0845 600 3078 or e-mail **enquiries@naturalengland.org.uk**.

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Executive Summary

Eutrophication occurs when excessive amounts of nutrients enter freshwater systems as a result of human activity within the catchment. This results in a general deterioration in chemical and ecological water quality, which poses a threat to the conservation status of many protected waters. Serious eutrophication problems are usually caused by an excess of phosphorus (P), because this nutrient usually limits primary productivity in freshwater systems.

A major source of P in many catchments is domestic waste. In urban areas, discharges of P from this source are controlled by the use of large waste water treatment works (WWTWs) with P stripping capabilities to process domestic waste. However, in rural areas, sewage treatment facilities often comprise small, on-site, systems that are much less efficient at retaining P than larger works. These include septic tank systems, which may be very close to relatively clean and environmentally sensitive freshwater ecosystems.

In theory, septic tank systems should pose little threat to the environment, because much of the P discharged from the holding tank is removed from the effluent as it percolates through the soil in the drainage field or soakaway. However, based on available, albeit limited, information, it seems that many septic tank systems do not function properly because they are incorrectly sited and/or improperly maintained. Studies in Ireland have indicated that more than 80% of septic systems are probably not working efficiently. Anecdotal information indicates that the situation in England may be similar, though this has yet to be firmly established.

The literature review conducted as part of this study revealed cases where septic tank discharges have had a significant impact on downstream P concentrations, causing increases of up to 700% in some cases. P concentrations of up to 400µg P l^{-1} have been recorded in relatively rural areas downstream of septic tank systems. This has the potential to cause considerable ecological damage at the local scale, especially in sensitive areas where internationally important conservation sites may be threatened.

In terms of seasonality, it is likely that septic tank effluents have the potential to increase the P concentrations of receiving waters all year round, depending on the local circumstances. This is because the P-laden discharges from most tanks, apart from those that discharge directly to a watercourse, can enter drainage channels *via* two different routes. In spring/summer, and under low flow/dilution conditions, P seeps through the surrounding soil or soakaway to the receiving water at a low, but relatively constant, rate. In autumn/winter, and under high flow/dilution conditions, higher levels of P can be flushed from the tanks themselves and the surrounding soils during heavy rainfall (Jarvie *et al*, 2006).

When integrated at the catchment scale, the impact of septic tank discharges on P concentrations is less marked but evidence exists to suggest that it can still be important. Assuming that all of the P lost from septic systems in the catchment entered watercourses, this study estimated that such discharges may increase in-stream P concentrations at the catchment scale by as much as 15-20 µg P l^{-1} in rural areas of the Hampshire Avon catchment and by 44-86 µg P l^{-1} in parts of the The Broads. These values represent 17-19% and 73-86%, respectively, of the measured P content of the rivers that drain these subcatchments. Whilst the assumption that all of the P lost from septic tanks reaches watercourses has undoubtedly led to overestimates of the significance of septic tanks in the catchments studied, the results do support the need for further work in this area. A follow on project, also

funded by Natural England, is now looking at the capacity of soil soakaways to remove P from discharged effluent before it reaches a waterbody.

Based on the assumption stated above, in this study the amount of P estimated to be entering the rivers Wylye, Nadder, Bure and Ant from septic tank discharges was compared to that estimated to be coming from agriculture and WWTWs. Within the Nadder river system, the P discharged by septic tanks was estimated to be equivalent to about 20% of that coming from agricultural sources and 62% of that coming from WWTWs; within the Wylye river system the corresponding figures were 100% and 42%. In The Broads, however, the proportion of P from septic tank discharges was estimated to be even larger than this. Within the upper Bure catchment, septic tank discharges of P were estimated to be more than 12 times that from WWTWs and 1.1 times greater than that from agriculture. The corresponding figures for the upper Ant catchment were 17 times that from WWTWs and 9.5 times that from agriculture. While these findings strongly suggest that P discharges from septic tanks are not 'negligible' at the catchment scale, as has sometimes been suggested, it is important to stress that these calculations are based on worst case scenarios. To refine them, further research is needed on the factors affecting the loss of P from small domestic discharges and the extent to which P can then be trans-located to water bodies. Some limited work in this area has been conducted in North America, but this has limited applicability to the UK situation. It should also be noted that these proportions would have been even higher if this study had considered concentrations of soluble reactive (bioavailable) phosphorus (SRP), rather than total P, because sewage-related discharges have a much higher proportion of SRP than agricultural runoff.

One of the main problems in estimating P losses from septic tanks at the catchment scale is estimating their number and location. Systematic and comprehensive records of the distribution of septic tanks in a catchment are rarely available. In this study a method based on interpretation of aerial photography was used to investigate the distribution and number of septic tanks for the Wylye, Nadder, upper Ant and upper Bure catchments. The results suggested that potentially fewer than 10% of septic tank discharges are consented within the Wylye and Nadder catchments of the upper Hampshire Avon, while potentially fewer than 3% are consented within the upper Ant and upper Bure catchments of The Broads.

A range of methods for estimating the number and, in some cases, location of septic tanks are described. These include methods based on postcode data, large area statistics, local knowledge, aerial photography and population census returns. Most of these methods are labour intensive and so a centralised system for better understanding the location of all septic tanks would significantly help the process of understanding their impact on the environment.

The likelihood of any particular septic tank causing pollution problems depends partly on its location and partly on its condition and the way that it is managed. In terms of location nationally available datasets could be used to help assess the overall risk of septic tanks causing pollution problems in a particular area. Factors to take account of using this approach could include soil hydrological characteristics, topography, septic tank density and proximity to a watercourse. This would provide an initial screening system that would allow areas where septic tanks are most likely to cause pollution problems due to their location to be highlighted. Within these areas, the potential for individual tanks to cause water pollution problems could then be further assessed from their size, design and the way that they are managed. However, this information is often not readily available. Possible procedures for assessing risk at both the catchment/regional/national scale and at the site specific scale are proposed, but these require further development and validation for use within the UK scale.

A variety of options are available for reducing the potential of septic tank systems to pollute the environment. By far the cheapest, and probably the most effective, is encouraging owners to de-sludge their systems more frequently to prevent the overflow of untreated sewage. This would otherwise block the soakaway, reducing its capacity to remove P from the effluent. Some tanks may also need to be upgraded or replaced to ensure that they are fit for purpose. In areas where there are clusters of septic systems, it may be possible to replace them with first time sewerage schemes. Although such improvements can reduce the discharges from these systems in terms of some pollutants, e.g. suspended solids, whether they will reduce the amount of P that is discharged by the tank is questionable.

A key issue in trying to quantify septic tank discharges of P and their impacts on chemical and ecological water quality is that very few data are available for analysis. For this reason, the results of this report are based on a modelling approach that assumes that all of the P discharged by properly functioning septic tank systems eventually ends up in a nearby watercourse. Although, in a properly sited and maintained septic system, some of this P would be retained by soil adsorption and biological uptake in the drainage field, evidence exists to suggest that many septic tank systems are not working properly. On balance, given the available information, it was assumed that discharges to water from an 'average' septic tank system were probably about 0.54 kg P $capita^{-1}$ y^{-1}, a value that is similar to many of those reported in the literature. It is, however, recommended that this figure is re-evaluated when better information becomes available.

Although there is considerable anecdotal evidence that discharges of P from septic tank systems can elevate P concentrations in receiving waters to levels that are sufficiently high to cause ecological damage, there are few data available with which the extent of this problem can be determined. This study has compiled sufficient evidence to conclude that this topic warrants further investigation. Such investigations should comprise detailed field studies aimed at measuring and evaluating sources, loads and impacts of discharges from these systems on receiving waters. In an ideal world, all SSSIs would be assessed at a site specific level for potential contamination from septic tank discharges. However, realistically, resource constraints mean that this can only be achieved at a small number of locations. So, as a first step, it is important to be to identify, and focus on, sites that have been identified as being at greatest risk of nutrient enrichment from these discharges. A high level risk assessment procedure based on existing, national level, datasets has been suggested. However, it should be noted that some of the boundary values proposed for assessing the level of increased risk associated with different variables are derived from values obtained from studies in other countries. As such, they have not been validated for application within the UK. Also, the relative importance of these factors, an important issue when combining them to create an overall, single, risk assessment value, needs further investigation.

To properly assess the significance of septic tanks in relation to phosphorus pollution, it is important that they are evaluated at the local, catchment, regional and national scale. This requires the location of each discharge to be known.

Contents

1 Introduction, objectives and outline of report structure

Nutrient enrichment (eutrophication) of freshwater systems occurs when excessive amounts of nutrients enter waterbodies as a result of anthropogenic activity within the catchment. This leads to a general deterioration in both the chemical and ecological water quality. In rivers, these effects include excessive growth of algae and macrophytes, reduced biodiversity of aquatic plant communities, discolouration of the water (green or brown), and chemical changes such as fluctuating levels of pH (Hilton & Irons, 1998). Similar impacts are also found in standing waters. The impacts of eutrophication are also associated with microbial breakdown of the increasing amounts of organic matter that result from nutrient enrichment, which can lower the amount of oxygen available to organisms in both the sediments and the overlying water. This is especially undesirable, as it often results in high profile fish deaths in affected waters.

Water quality problems associated with human habitation have led to the introduction of large and small scale sewage treatment facilities across the developed world. Although most of the sewage associated with urban populations is dealt with by large waste water treatment works (WWTWs) some of which have phosphorus (P) stripping capabilities, this mains sewerage network does not extend far beyond the outskirts of small towns and larger villages in rural areas. So, most domestic effluent in these areas is treated by small, on-site, sewage treatment facilities such as septic tank systems, before being discharged into the environment (Hall, 2001). Although these systems may not pose a significant threat to water quality at the national level, especially in comparison with discharges from large WWTWs, they may have a significant impact on the ecological quality of nearby waterbodies at the local scale.

The problems associated with small discharges of effluent from on-site sewage treatment facilities are likely to be increasing. This is partly because of the level of rural development that is often associated with greater recreational use of the countryside by urban populations (e.g. Spey Catchment Steering Group, 2003). This is often associated with the conversion of farm buildings to holiday accommodation, the development of visitor's centres to encourage tourism, and the establishment of the temporary 'tented villages' at large outdoor music festivals. All of these activities bring more people into rural areas, creating more sewage that needs to be dealt with in an environmentally sensitive way. It should be noted, however, that rural development does not necessarily result in proportional increases in the load of P to waterbodies. In Northern Ireland, for example, Foy *et al*. (2003) found that, although the rural population within the catchment of Lough Neagh had increased between 1990 and 2000, the corresponding P load to the lake had decreased. This was due to a reduction in the use of P-based detergents over that period.

In 1991, the Urban Waste Water Treatment Directive (UWWTD) was introduced by the European Union (EU, 2000), with member states translating it into local legislation in 1998. The UWWTD set the level to which sewage must be treated before being discharged into a waterbody in an environmentally sensitive area. As a result, many larger WWTWs across the UK were upgraded, reducing their P output by about 80 per cent (e.g. May *et al*., 1996; Bowes *et al*., 2005). This, together with additional reductions in P loads from large WWTWs that were discharging to Natura 2000 sites and SSSIs, which have been driven through the requirements of the Habitats Regulations and the Countryside and Rights of Way Act, has sometimes led to the belief that most of the remaining P input to waterbodies – especially in rural areas - comes from agriculture. In reality, however, this is untrue because the UWWTD applied only to WWTWs serving populations of more than 10,000 people Also, reductions in P loads from WWTWs that enter sites designated as Natura 2000 and Sites of Special Scientific Interest (SSSI) has usually gone no further than implementing an emission standard of $1mg\ l^{-1}$ at selected works, with significant WWTWs in continuity with some sensitive SSSIs receiving no P stripping whatsoever. To achieve appropriate water quality targets at many aquatic SSSIs it seems likely that further reductions in both agricultural and sewage-related inputs will be required. There is now mounting evidence that P loads from small discharges such as septic tanks may be significant in terms of the nutrient management of some receiving waters. May *et al*. (1996), for example, estimated that, since the WWTW at Keswick was upgraded in 1993, about 18 per cent of the P load to Bassenthwaite Lake has probably been coming from septic tank discharges within the catchment.

Although many previous studies have attempted to quantify the relative importance of P loads to waterbodies from small point source discharges (Table 1.1), most of these assessments have been based on incomplete, and often unreliable, information. As a result, many authors have simply noted such discharges as being 'important and worthy of attention' (e.g. Babtie Group, 2001; Spey Catchment Steering Group, 2003), and there is still a widely held belief that these discharges have little impact on P concentrations in receiving waters. More recently, however, Jarvie *et al.* (2006) and Arnscheidt *et al*. (2007) have suggested that P-laden effluent from small point sources such as septic tanks may still pose a greater risk to the quality of UK river water than P from diffuse agricultural sources in some catchments.

There appears to be a growing body of evidence that indicates the ecological impact of small discharges of P may be more significant in some situations than was once thought (e.g. Carvalho *et al*. 2005; Smith *et al*., 2005; Jarvie *et al*., 2006). As a result, Natural England is concerned that inputs from small discharges could threaten the attainment of "favourable condition" of some SSSI rivers and lakes, especially where they are concentrated in small rural catchments and in the headwaters of rivers. In these locations, their effect may be disproportionately greater than further downstream where dilution capacities are greater.

As many small discharges are not consented, they have not been considered in the EA Review of Consents (RoC) process for sites designated as Special Areas of Conservation (SACs) or Special Protection Areas (SPA). Instead, their contribution has usually been included in the diffuse pollution load estimates that are generally attributed to agricultural sources. Even where consented, many small discharges have been considered to be 'trivial' and, as such, have not been included in the RoC process. Nevertheless, determining action to safeguard and restore both SAC/SPAs and SSSIs may require some action to be taken on controlling these small discharges. This could include the introduction of first time rural sewerage schemes, innovative P removal technologies and/or use of reedbed systems, reduction of inputs or diversion, including removal of tanked effluent to approved disposal sites.

Table 1.1 Estimated P input to waterbodies from septic tank systems within their catchments.

Waterbody	**Estimated P input to waterbody from septic tanks** (tonnes y^{-1})	**Proportion of external P input to waterbody attributable to septic tank discharges** (%)	**Reference**
Bassenthwaite Lake	2.3	18.0	May *et al*. (1996)
Black Beck	0.25	40 - 76	Hall (2001)
Llyn Tegid	4.6	3	Millband *et al*. (2002)
Loch Earn	0.07	1.2	Weller (2000)
Loch Flemington	0.02	17.5	May *et al*. (2001)
Loch Leane	1.5	12.0	Kirk *et al*. (2003)
Loch Leven	1.0	14.0	Dudley & May (2007)
Loch Ussie	0.03	22.0	May & Gunn (2000)
Lough Conn	1.58	5	McGarrigle & Champ (1999)
Lough Derg	25.8	12	Kirk *et al*. (2001)
Lough Erne	-	12.0	Foy (*pers. comm.*)
Lough Neagh	56.0	14.0	Foy (*pers. comm.*)
River Boyne	5.6	8	MCOS (2002)
Loweswater	23	10	Maberly et al (2006)
River Liffey	1	3	MCOS (2002)
River Suir	-	7	MCOS (2002)
All standing waters in Northern Ireland	118	5	Smith *et al*. (2005)
All waterbodies in Northern Ireland	130	8.5	SNIFFER (2006b)
All waterbodies in Scotland	142	2.6	SNIFFER (2006b)

1.1 Aims and objectives of this study

The aim of this report is to provide an indication of the potential scale and nature of the problem of small discharges of P for designated sites and to start the process of identifying management solutions to reduce impacts, where required. This will be used to determine the possible scale of such discharges as a factor influencing the condition of designated freshwater SSSIs more generally and to raise awareness of the issue. It will also be used to inform and influence site management strategies and to identify research needs and advice for future improvement programmes (e.g. Water Framework Directive Programme of Measures; water company asset management plan (AMP) process). The scope of this work is limited to the impact of P on freshwater habitats.

The study is divided into two main tasks with the following objectives:

Task 1

- review appropriate literature concerning the impact of small discharges of P in relation to eutrophication;
- review existing guidance on managing small discharges;
- summarise the conditions under which small discharges may result in significant impacts at local and/or site level;
- describe the range of options available to deal with small discharges;
- suggest an approach that could be applied across aquatic SSSIs to establish the scale and likely impacts of small discharges.

Task 2

- undertake two desk-based case studies to investigate the scale and potential impact of both consented and unconsented small discharges on water quality;
- develop an outline risk assessment process for prioritising remediation measures.

The structure of this report reflects these objectives.

2 On-site options for waste water treatment

2.1 Introduction

Large numbers of properties in rural areas of the UK are not connected to mains sewerage systems. These rely on on-site water treatment systems, such as cess pits, septic tanks and private sewage treatment works (or package treatment works). Septic tanks, either privately or publicly owned (DCMP, 2006), are the most common on-site sewage treatment systems in these areas (Wood *et al*., 2005). For this reason, their function, effectiveness and possible failings are discussed in detail in this chapter.

2.2 Septic tanks

2.2.1 Design, function and regulation

Septic tanks, which are believed to have originated in France in the 1870s, were probably introduced into the United Kingdom (UK) and the United States (US) in the 1880s (Canter & Knox, 1985). They comprise a buried, two-chamber, tank (Figure 2.1) that is designed to maximise the removal of solids, pathogens and other pollutants from the waste that they receive (Goldstein & Wenk, 1972; Viraraghavan, 1976; Canter & Knox, 1985). In a properly functioning septic tank system, scum floats to the top (Figure 2.2), other solids settle to the bottom and the clarified effluent (Figure 2.3) percolates into the surrounding soil. It is generally assumed that effluent from these systems is then 'cleaned up' as it passes through the surrounding soil, thus posing little threat to the environment (Wood *et al*., 2005).

Although the original septic tanks were built of brick or concrete, most modern systems comprise a bulb-shaped, self-contained, fibre-glass unit. Other than that, their basic design and function has changed very little over the years. Many older properties still rely on their original, Victorian, septic tanks for the treatment of waste water which are, in many cases, under-sized for modern patterns of domestic water use, such as frequent bathing and the use of domestic appliances, and may not have been upgraded when the property that they serve was upgraded or developed (Selyf Consultancy, 2002). So, they tend to overflow and discharge untreated waste into the environment. This is a particular problem during heavy rainfall, because many older systems receive roof runoff as well as domestic waste. Selyf Consultancy (2002) surveyed 124 septic tanks within the catchment of Llyn Tegid, Wales, and found that more than half of these tanks were affected by this problem. In addition, many of older, unconsented systems discharge effluent directly to watercourses without secondary treatment; in the River Irvine catchment, Scotland, for example, 82% of septic tanks were found to discharge in this way (Aitken *et al*., 2001).

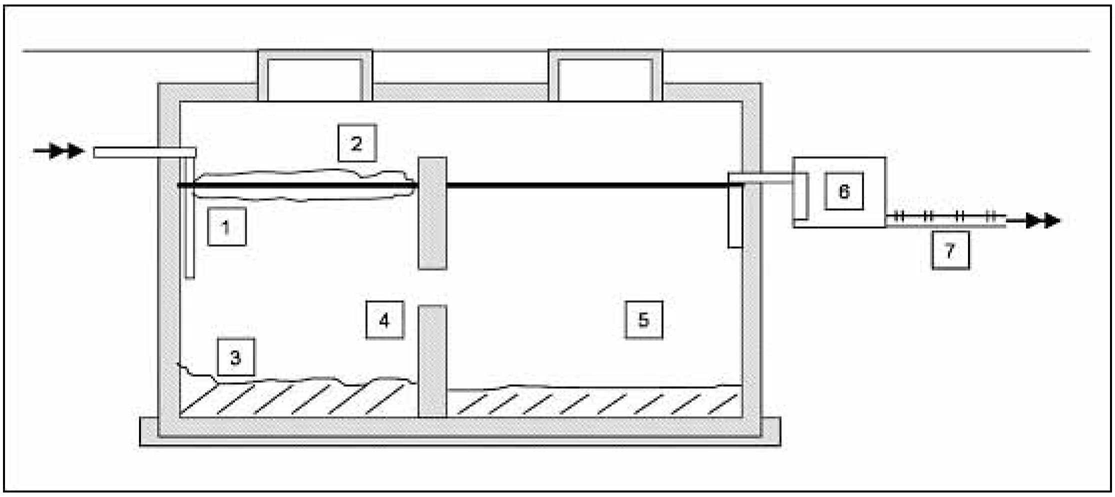

Figure 2.1 A standard septic tank design. 1 - inflow; 2 – floating scum; 3 - settled sludge; 4 - connection between chambers; 5 - secondary chamber; 6 outflow and effluent inspection chamber; 7 – soakaway or drainage system (reproduced from Hilton et al., unpublished).

Detailed information on the functioning and effectiveness of septic tanks can be found in reviews published by Canter and Knox (1985) and Beal *et al*. (2005). Most of the literature seems to suggest that septic tank systems are an effective means of processing household waste (especially sewage) in rural areas, if they are functioning correctly. However, this depends on the design of the system, how it is managed and the environment in which it is placed (Cotteral & Norris, 1969). Most of the problems caused by septic tank discharges relate to the last two of these issues, i.e. how a septic is managed and where it is placed.

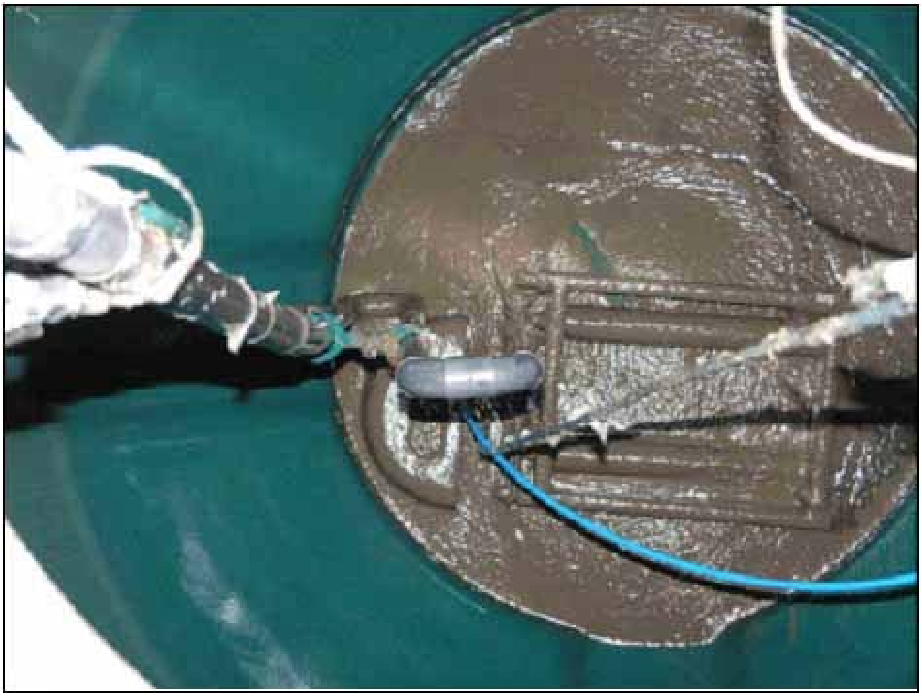

Figure 2.2 Floating scum layer inside the reactor vessel of a 'Balmoral' septic tank as viewed from above. The tubing forms part of the "air lift" pump system that transfers effluent between the chambers and pumps clarified effluent out of the system (Image courtesy of J Malley, National Trust).

In most cases, the operation and maintenance of a system is unregulated and left to the householder. This often results in tanks not being emptied regularly (Kirk *et al*., 2003), which has been shown to be a key factor in determining their impact on nearby water quality (Arnscheidt *et al*., 2007). There are many reasons why septic tanks are not emptied as often as they should be. In some cases, people are reluctant to pay the costs of having their tanks emptied. In other cases, the owners believe that a septic tank does not need emptying if it is working correctly. This is a very widespread misconception in many parts of the UK and there have been many attempts to address this with public information campaigns. An example of such an approach, in relation to reducing septic tank discharges to Loch Leven, Scotland, can be found at http://news.bbc.co.uk/1/hi/scotland/tayside_and_central/6912062.stm. Although it is recommended that the sludge or 'septage' that accumulates within a septic tank should be removed at 2-3 year intervals, depending on their design capacity, in reality tanks are often emptied much less frequently. Aitken *et al*. (2001), for example, found that the most common frequency of de-sludging in a catchment in Ayrshire, Scotland, was every 5 years or more. In a similar study of 24 septic tanks in the Lough Leane catchment, Ireland, few tanks were found to have been emptied regularly (Kirk *et al*., 2003). As a result, 88 % were found to be full of sludge to the outlet and not functioning efficiently.

Figure 2.3 Clarified effluent from a septic tank.

In recognition of the problems that can be caused by septic tanks in rural areas of England and Wales, government guidance contained within DETR Circular 03/99/ WO 10/99 'Planning requirements in respect of the use of non-mains sewerage incorporating septic tanks in new development' discourages the installation of new systems by requiring a hierarchy of other drainage options to be considered and discounted, in the following order, before any decision is made to install such a system:

1. Connection to the public sewer.
2. Package sewage treatment plant (which can be offered to the Sewerage Undertaker for adoption).
3. Septic tank.

If it is concluded that a septic system is the only viable option, building regulations are in place to ensure that new systems are placed in suitable locations and that discharges meet certain water quality standards. For example, they must also be constructed in such a way that they can be properly maintained in good working order and they must provide sufficient storage capacity to meet the demands of the population that they serve. These requirements are summarised in the building regulations for England and Wales on drainage and waste disposal (The Building Regulations 2000 – Approved Document H), as follows:

1) Any septic tank and its form of secondary treatment shall be sited and constructed so that:
   (a) it is not prejudicial to personal health;
   (b) it will not contaminate any watercourse, underground water or water supply;
   (c) there is adequate access for emptying and maintenance; and
   (d) it will function sufficiently well for the protection of health in the event of a power failure, where relevant.

2) Any septic tank, holding tank or cesspool shall be:
   (a) of adequate capacity for its intended use;
   (b) constructed so that it is impermeable to liquids; and
   (c) adequately ventilated.

In addition to the above, a number of siting restrictions apply to each development. These include a specified minimum distance from habitable buildings and from any site where groundwater is abstracted for drinking. As far as we have been able to ascertain, restrictions on P concentration or load in the effluent from these systems does not usually form part of the discharge consent process in England and Wales (Cormac Quigley, Environment Agency, *pers. comm.*); only biochemical oxygen demand (BOD), suspended solids (SS) and ammonia (NH_3) discharges are regulated. This is in contrast to the situation in Scotland where consented effluent discharges must remain below agreed P discharge concentrations of about 2 mg P l^{-1} and property developers creating new discharges in some P sensitive catchments must be able to show how they will reduce the overall input to the receiving water by 125% of the expected output from the proposed new source (LLCMP, 1999). If they cannot do this, approval will be refused by the local planning authority. Unfortunately, there are less stringent controls over P concentrations in effluent from existing systems due to lack of resources to monitor large numbers of small discharges and follow up those that are not complying with their discharge consents. A recent study in Scotland showed that some existing tanks complied with their discharge consents in only one of nine times that they were monitored, and others failed to comply at all (Roxburgh & Anderson, 2006). The total phosphorus (TP) concentrations in the effluents measured ranged from 6 to 15 mg l^{-1}.

2.2.2 When things go right

In a properly functioning septic tank system, the influent waste water contains about 25 mg P l^{-1} as TP of which about 9 mg P l^{-1} (i.e. 36%) is soluble orthophosphate (OP) (Bauer *et al*., 1979). Within the tank, anaerobic digestion converts influent P to OP, such that > 85% of the TP in septic tank effluent is OP (Bouma, 1979).

When waste water enters a septic tank, solids settle in the first chamber where they undergo anaerobic digestion by bacteria. Much of the particulate P in the original waste is retained in these solids, but some is converted to soluble P. This process increases the concentration of soluble P in the effluent, making it (~ 10 - 12 mg P l^{-1}, Viraraghavan, 1976; Viraraghavan & Warnock, 1976) greater than the concentration in the influent waste water (~ 9 mg P l^{-1}; Bauer et al., 1979). So, in terms of soluble P, the liquid that passes out of the tank and into the soakaway or, in some cases, directly into the stream, is 33% richer in dissolved P than that in the influent. That said, there is an overall reduction in TP concentration between the influent and the effluent which, in a properly functioning system, is estimated to be about 50% (Gold & Sims, 2006).

The effluent from septic tanks usually passes into a secondary treatment system such as a soakaway or drainage field, a package treatment system, or a passive treatment system such as a reed bed, before discharging into a waterbody. The relative merits of these different systems in relation to their ability to remove P are unclear, and a comparison of their effectiveness is needed to inform policy positions on new developments and authorisations.

When effluent from a septic tank is discharged to a soakaway, the dissolved OP is at least partly adsorbed onto soil particles in the unsaturated zone. This reduces the concentration by about 25% - 50% (Canter & Knox, 1985). In contrast, waste water treatment plants, which are not designed to remove P (Gill *et al*., 2009), probably remove only about 15% of the OP, which is usually achieved by bacterial assimilation, precipitation and adsorption (Metcalf & Eddy, 2003). The effectiveness of reed beds in removing P from effluent water is unclear, but it is generally believed that these systems are best suited to ‘polishing’ effluent once it has already passed through another type of secondary treatment system. It should be noted, however, that some septic tank effluents discharge directly into waterbodies, especially in the case of older systems.

Even if the P in effluents from septic systems is adsorbed onto the surrounding soil particles in the short term, little is known about the capacity of soils to adsorb nutrients indefinitely, or how the poor maintenance of septic systems affects the ability of soakaways to remove soluble phosphate (see Beal *et al*., 2005, for review). In general, it seems likely that soils in the drainage field probably lose the ability to retain soluble nutrients over time and could, under certain circumstances, change from a sink to a source of P (Beal *et al*., 2005). In any case, recent research has shown that this P attenuation in soils is not irreversible, so the P remains mobile and likely to contaminate downstream waterbodies for many years even if the septic tank itself is removed (Robertson, 2008). Robertson (2008) also showed that the groundwater plume of discharge from a septic tank system was still relatively high in P content (i.e. ~ 2 mg l^{-1}) at a distance of 16 m from the source and that this plume had advanced towards a nearby waterbody by about 1m per year since the tank had been installed.

2.2.3 When things go wrong

If any of the elements within a septic tank system fail, the system will no longer perform as expected and the efficiency with which pollutants are removed from the effluent will be reduced. One of the most frequently recorded causes of septic tank failure is poor maintenance. To function properly, septic tanks need to be emptied of accumulated sludge every 2-3 years. However, there is a general lack of awareness of this amongst many septic tank users (e.g. Kirk *et al*., 2003; DCMP, 2006). This is mainly due to the common misconception that a properly working septic tank does not need to be emptied. However, if the sludge is not removed periodically, the effective volume of the tank is reduced. This reduces the residence time of the effluent, which, in turn, reduces the amount of processing that the effluent receives. As a result, more undigested material, which would otherwise have been deposited as solids in the tank, may reach the soakaway or be discharged directly into a waterbody. Increased amounts of solids passing through the tank and directly into the soakaway may block the biomat, potentially causing hydraulic failure.

Care should be taken in the disposal of sludge from septic tanks when they are emptied, as it may contain large amounts of P, possibly 170 to 350 mg P l^{-1} (USEPA, 1980). This could pose a serious threat to water quality if it enters the environment from a faulty or improperly managed tank, or if sludge is disposed of inappropriately. In parts of Ayrshire, for example, 61% of farmers spread the sludge from their septic tanks onto nearby farmland (Aitken *et al*., 2001), which has the potential to cause serious contamination of nearby watercourses. Even if emptied and taken away, there is a risk that a small WWTW could suddenly be overloaded by the sudden introduction of a large quantity of sludge from a tanker if it does not have sufficient holding tank capacity to store the sludge and process it over time. This, too, can cause pollution problems within the catchment or, indeed, within another catchment altogether (Wood & Gibson, 1974).

Hydraulic failure is another common problem in septic tank systems. It occurs when infiltration rates are lower than the loading rate of effluent into the soakaway or drainage system (Beal *et al.*, 2005) and results in effluent being discharged onto the soil surface (Figure 2.4). This situation is most common in older or poorly designed septic systems, or those that are being used beyond their original design capacity. The latter situation usually arises where the use of clothes and dish washing machines has replaced manual cleaning, or where the number of people using a septic tank system has increased.

When hydraulic failure occurs, septic tank effluent by-passes the soakaway system and flows directly into surface waters, generally a nearby ditch or stream, as observed within the Llyn Tegid catchment (Selyf Consultancy, 2002). If this occurs, both undigested organic matter (including bacteria and viruses) and large amounts of inorganic nutrients enter the water body directly and without being processed. A common response to hydraulic failure by septic tanks users is often to excavate a ditch from the soakaway to the nearest watercourse (Jordan, *pers. comm*., regarding septic tank systems in Northern Ireland), thereby by-passing the soakaway altogether. One of the authors of this report is also aware of a situation in Scotland where the effluent from a septic tank systems with hydraulic failure was ‘plumbed’ directly into a nearby watercourse to provide a cheap solution to dealing with a ‘soggy patch’ in the garden. So, it seems possible that these types of response to hydraulic failure may be commonplace across the UK.

Figure 2.4 Effluent discharging onto the soil surface as a result of hydraulic failure in a septic tank system; note the algal bloom that is developing on the nutrient rich puddle that is forming at the base of the slope.

Many failures of septic systems are the result of poor maintenance, which results in leaking tanks, broken or missing pipes, blocked inlets and outlets, leaking or missing filters or blocked soakaways (Selyf Consultancy, 2002). Other problems may be caused by the addition of inappropriate chemicals, which reduce the rate of bacterial decomposition and, consequently, the effectiveness of the tank. Selyf Consultancy (2002) found that few owners took their septic tank into account when selecting domestic cleaning products, a problem that was even more marked in holiday accommodation where occupants were unaccustomed to using septic tanks and tended to flush chemicals and inappropriate objects down the toilet for disposal. Such objects included condoms that were found blocking filters. The report concluded that the effectiveness of many septic systems could be improved with better management and improved local education for users.

2.3 Other on-site waste water treatment options

Where septic tank discharges are causing contamination of watercourses, other options for reducing levels of pollution may be useful. These are outlined below. It should be noted, however, that all new discharges from any waste water treatment systems will need to be subject to the appropriate regulatory control by the relevant regulatory agency.

2.3.1 Secondary treatment of effluent

Secondary treatment of septic tank effluent, such as using peat bed filters (Patterson *et al*., 2001) or adsorbing P onto aluminium or iron-based additives (EPA, 2008), could be used to reduce the amount of P in tank effluent before it enters the drainage field. However, it has been found that nitrification during the secondary treatment process reduces the pH of the effluent, which can adversely affect pH-dependent reactions that fix P within the soil matrix (Gill *et al*., 2007). For this reason, secondary treatment prior to discharge to the drainage field is unlikely to reduce the overall discharge of P from the system. While this process decreases the P load from the tank to the drainage field, it also reduces the P removal efficiency of the latter. So, the overall improvement is probably negligible (Holman *et al*., 2008).

2.3.2 Package sewage treatment plants

Package treatment plants produce a higher quality of effluent than a septic tank system alone, because they use micro-organisms to break down organic matter in the effluent before it is discharged to soakaway. Depending on the type of system used, it has been suggested that these treatment plants can reduce the P content of discharged effluent by up to 99% (Heistada, 2006). However, before any firm conclusions can be drawn more research is required that compares effluent from these sources to standard septic tank systems. This is especially true where package treatment plants discharge directly to a water course. In this situation, a standard septic tanks system that discharges to a drainage field may retain more P than a package treatment system that discharges directly to a waterbody.

There are many types of package treatment plant to choose from, but it is important that any system that is installed complies with local and European regulations on effluent quality. Some systems have a settlement tank and can replace an existing septic tank system while others need to be attached to a separate settlement tank or septic tank. Both primary and secondary package treatment systems will improve effluent quality, but require a constant supply of sewage to maintain the level of microbial activity required to operate effectively. This means that they are unsuitable for seasonal use, e.g. by holiday accommodation, unless combined with a flow management system.

2.3.3 Cesspools

A cesspool is a covered and contained system for storing waste and, because it has no outlet, the system is entirely dependent upon frequent emptying to avoid overflow. It is estimated that an average household would fill a tanker sized cesspool in about two weeks. As such, these systems are only practical for temporary storage of waste. It should also be noted that their use is discouraged in England and Wales and banned in Scotland (Environment Alliance, 2006).

2.3.4 Waterless toilets

Waterless toilets, such as chemical toilets or composting toilets, reduce pollution problems under some circumstances. Chemical toilets are only suitable for providing temporary facilities, e.g. at campsites, on building sites or at large outdoor events. The accumulated waste must be disposed of in a responsible manner and cannot be discharged to a watercourse, surface drain or to groundwater. In contrast, composting toilets use natural processes to convert waste matter into compost and are useful in remote locations, although they do require maintenance to ensure that the composting system remains effective. Neither of these systems is likely to replace the current system of septic tanks in unsewered areas.

2.3.5 Reed beds and constructed wetlands

Reed beds and constructed wetlands can be used to improve the quality of effluent discharged by a septic tank or a package treatment system, but cannot be used as a primary source of sewage treatment. It should be noted, however, that these systems can change from a sink to a source of P if environmental conditions change (Gabriel *et al*., 2008) or if P binding sites become saturated (Mann, 1990). A number of projects are currently underway to critically evaluate the role of constructed wetland for removal of P, including the reversion from a P sink to a source. An example is that recently conducted by the Wildfowl and Wetlands Trust and the NERC Centre for Ecology and Hydrology at Slimbridge, Gloucestershire (Duenas *et al*., 2007). Here, initial results showed the wetland removing 20% of the total phosphorus (TP), 5.2% of soluble reactive phosphorus (SRP) and 100% of particulate phosphorus (PP) from the influent water which had a P content of about 0.78 mg l^{-1} TP, 0.35 mg l^{-1} SRP and 0.43 mg l^{-1} PP. After 10 years of operation, those values had fallen to 10% of the total phosphorus (TP), -40% of soluble reactive phosphorus (SRP) and 50% of particulate phosphorus (PP) in the influent water. So, the reed bed had become a source of SRP. Regular changing of the substrate and cutting/removal of plant biomass is required to keep these systems in good working order.

3 Evidence based review of the impacts of septic tank discharges on receiving waters

3.1 Introduction

Discharges from septic tanks and other small sources of P to nearby water courses have rarely been measured or documented and most of the information from the UK linking them to established water quality problems is anecdotal. However, the Centre for Ecology and Hydrology (CEH) and many other organisations hold small amounts of, mostly unpublished, data that support this assertion. These data are summarised below. Some of these case studies also provide evidence that exceptionally large discharges from some of these sources may be driven by high rainfall events. This is an important observation, as this type of event is often missed by routine monitoring and may become increasingly common with climate change.

3.2 Impacts at the site specific scale

3.2.1 Example 1: Eye Brook

The first example is part of a study on the Eye Brook, Leicestershire, which formed part of a Defra-funded project called ‘Phosphorus from Agriculture Riverine Impacts Study’ (PARIS) (Stoate, 2008; Dils, *pers. comm*.). Samples were taken up and downstream of two septic tank locations within the catchment. In general, median SRP concentrations were found to be three to four times higher downstream of septic tank locations than upstream, while median TP concentrations were approximately double (Figure 3.1). Upstream of these locations, farming was the only source of P.

Overall the project found that inputs to streams from septic tank discharges were more concentrated than runoff from agricultural land and were delivered more continuously, with the mean concentration for SRP from this source being about 551 $\mu g\ l^{-1}$. So, although, at an annual scale, agricultural sources contributed more than 95% of the in-stream P load, the much smaller contribution from septic systems was a major contributor to the seasonal P load under base flow conditions. It was concluded that this was because septic systems are a continuous source of P that is not diluted by water from the surrounding land during low summer flows.

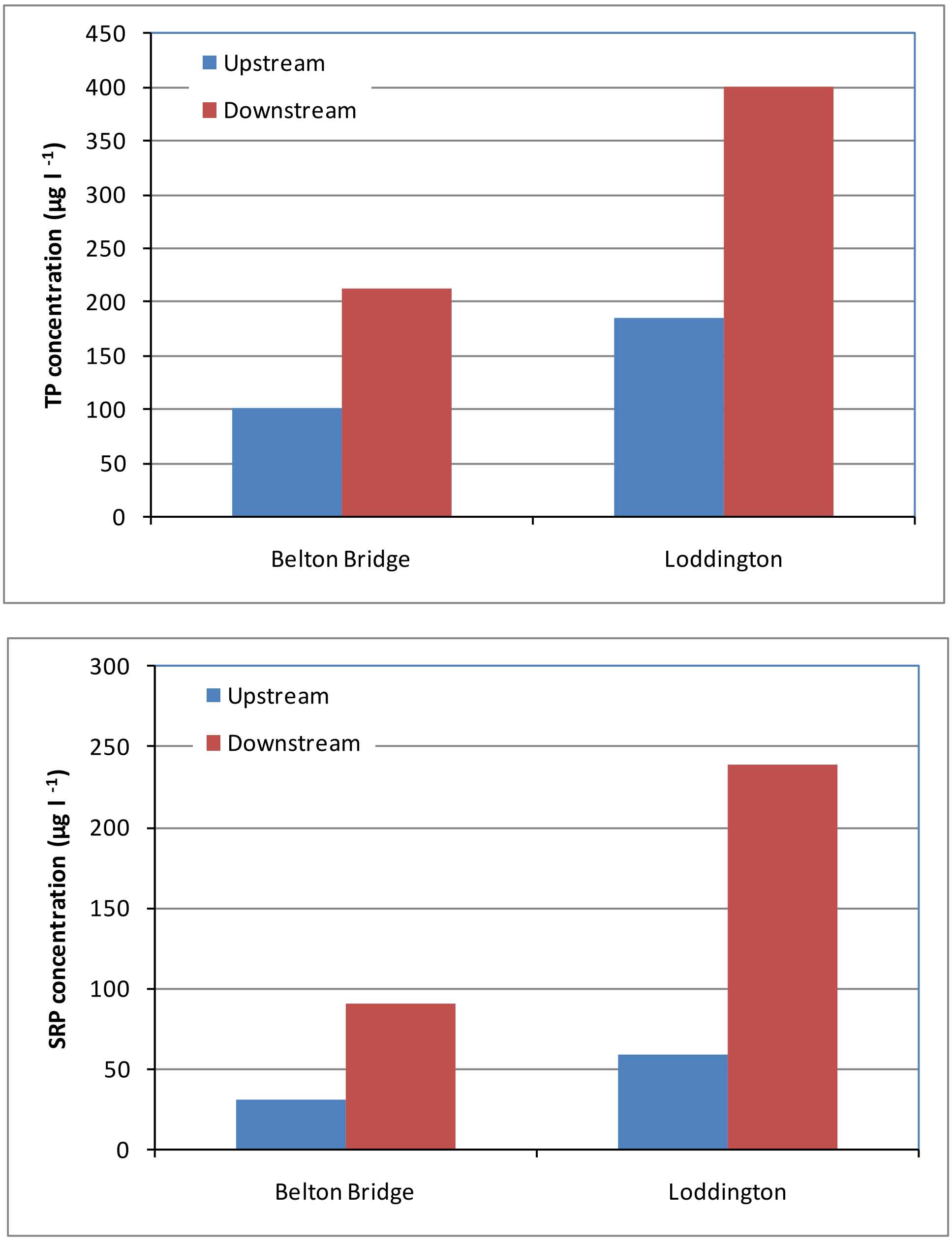

Figure 3.1 Median concentrations in total phosphorus (TP) and soluble reactive phosphorus (SRP) concentrations upstream and downstream of septic tank discharges in (a) Belton Bridge stream and (b) a stream near Loddington village, October 2006 to October 2007.

Overall, the main conclusions from this project were that, in rural headwater catchments, the human population is large enough to cause significant nutrient pollution and that septic tanks behave as multiple point sources rather than diffuse sources. In addition it was observed that, although the impact from each tank was probably small, the cumulative impact from all tanks across a rural catchment was probably significant.

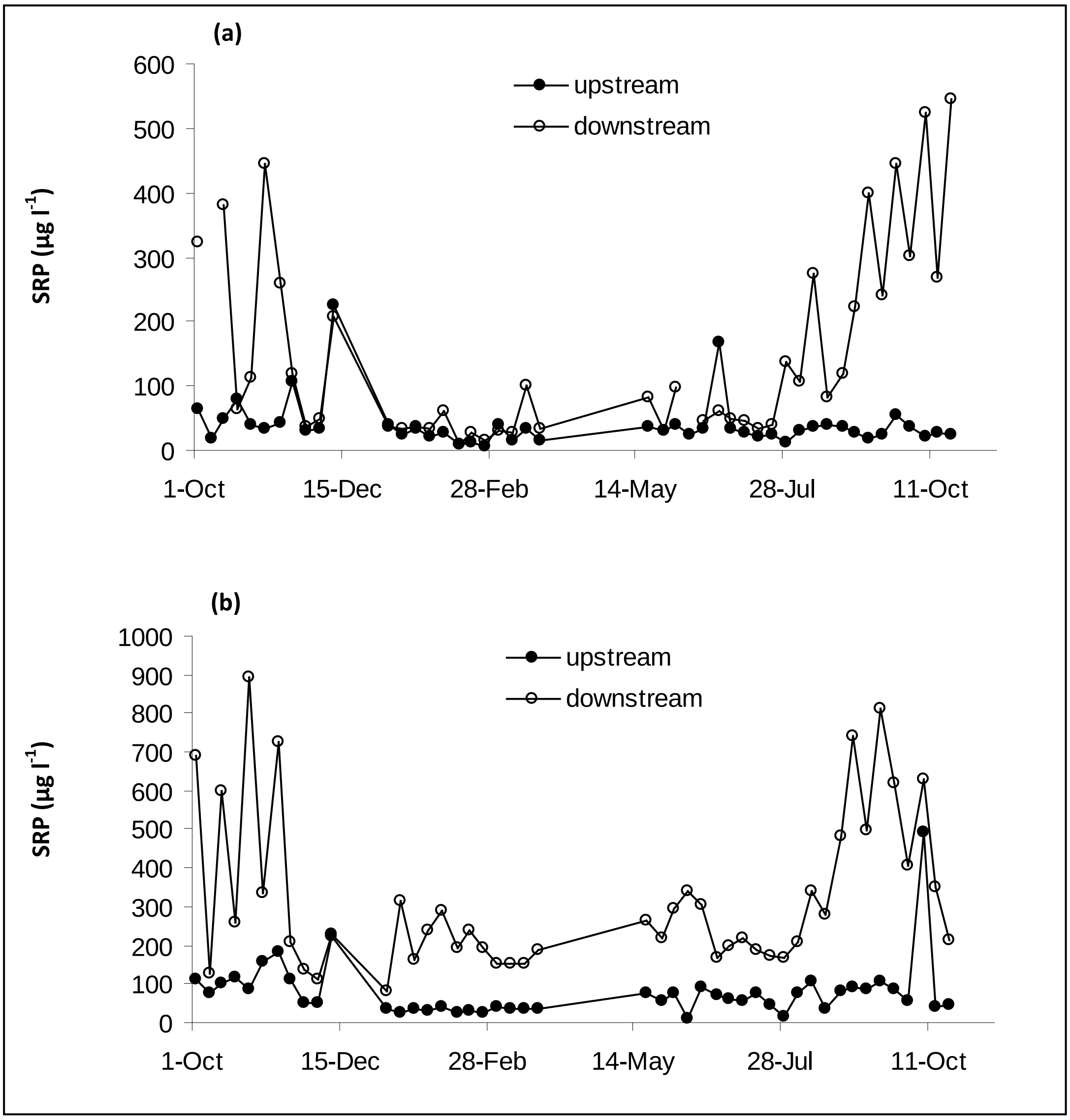

Figure 3.2 Soluble reactive phospshorus (SRP) concentrations upstream and downstream of septic tank discharges in (a) Belton Bridge stream and (b) a stream near Loddington village, October 2006 to October 2007.

3.2.2 Example 2: Loweswater

The second example is from a small feeder stream to Loweswater, Cumbria, (Figure 3.3). Here, OP concentrations were monitored from October 2004 to September 2005 (Maberly *et al*., 2006). Although annual mean OP concentrations in most of the inflows to the lake were low (i.e. < 10 µg P l^{-1}), this stream had a much higher OP concentration (i.e. ~ 24 µg P l^{-1}). Further investigation showed that the stream was receiving effluent from a faulty septic tank.

When combined with stream discharge rates, the mean daily OP load from this tank over the whole year was estimated to be approximately 8 g P d^{-1} or 2.9 kg P y^{-1}. However, during a storm event in December 2004, a single value of 122 g P d^{-1} (i.e. about 4% of the annual P load) was recorded. This highlights the importance of rainfall driven discharge events in delivering nutrients to watercourses from some small discharges. In this case, this was a septic tank of outdated design that received roof runoff as well as sewage and, therefore, tended to overflow during heavy rainfall.

3.2.3 Example 3: Loch Leven

The third example comes from the Loch Leven catchment, Scotland (Brownlee, 2008). Here, P concentrations were measured at five sites along the Greens Burn on 1 July 2008 (Day 1) and 22 July 2008 (Day 2). This stream passed a small cluster of buildings between sites 4 and 5 and, as it passed this location, P concentrations increased sharply (Figure 3.4). This strongly suggested that there was a small discharge of P-laden effluent between these sites, probably associated with the farm steading or an on-site sewage treatment facility. Although this example serves to illustrate the level of increase in P concentration that may be associated with small rural discharges, it is unclear exactly what proportion of the increase at this site was associated with the septic tank discharges, alone.

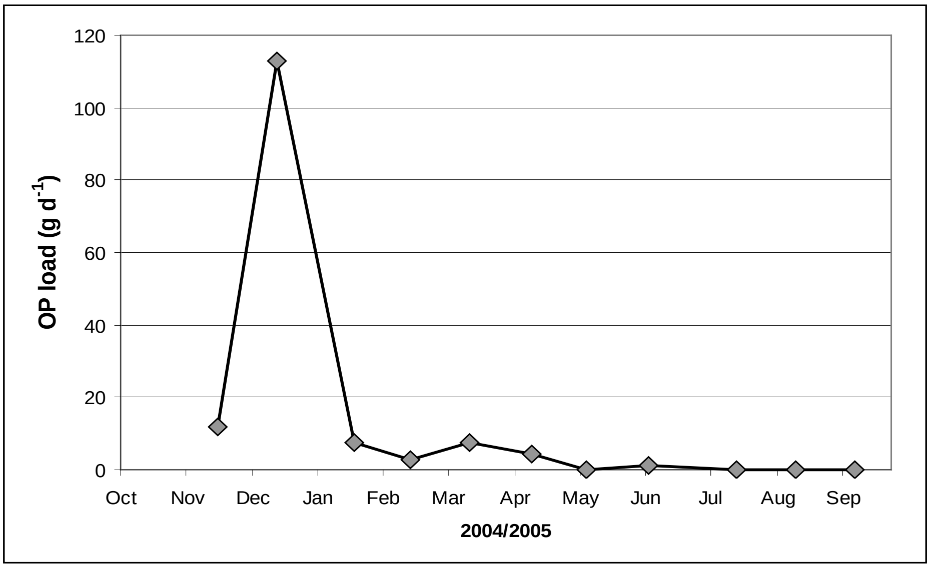

Figure 3.3 Seasonal changes in orthophosphate (OP) loads in a small inflow to Loweswater in 2004/2005.

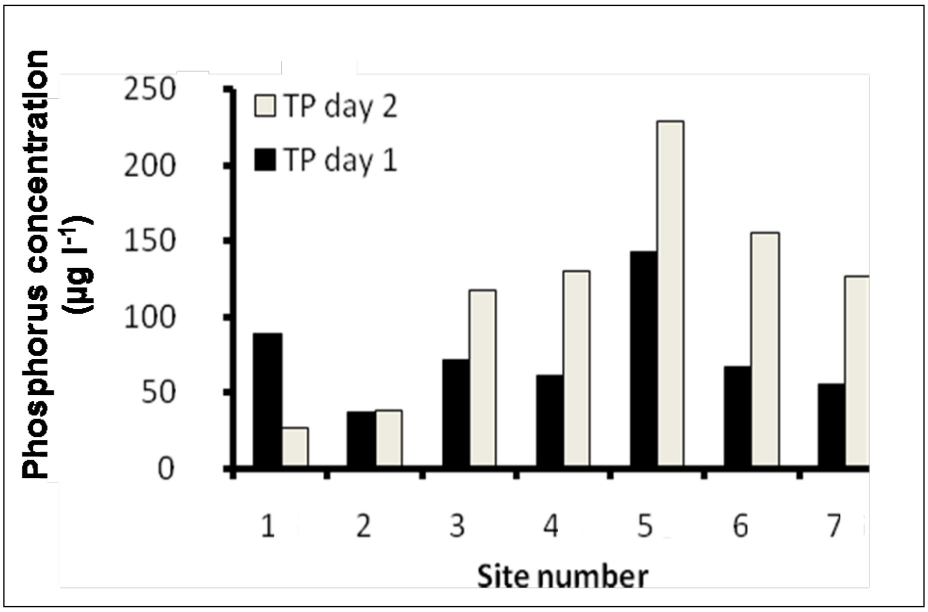

Figure 3.4 Total phosphorus concentration along a stretch of river in the Loch Leven catchment that passes a small group of properties between sites 4 (upstream) and 5 (downstream) *(after Brownlee, 2008).*

3.2.4 Example 4: River Wyre

The fourth example comes from a study of the River Wyre catchment in Lancashire (Nicholson, 2007). This study aimed to determine whether septic tanks contributed to SRP concentrations and loads in a stream that ran close to a small cluster of houses that were served by septic tanks. By measuring flows and concentrations along this stretch of the stream, the author was able to detect a marked increase in P concentrations and P loads downstream of the septic tank location, which was between sampling sites 4 and 5 (Figure 3.5). In-stream concentrations rose from about 50µg P l-1 to about 400µg P l-1 over a distance of about 100 m. As there were few other possible sources of P in this area, it was concluded that the sudden increase in P concentrations and loads in this stream were attributable to septic tank discharges. The discharge from these septic tanks to the river was estimated to be about 19.9 kg P y^{-1}.

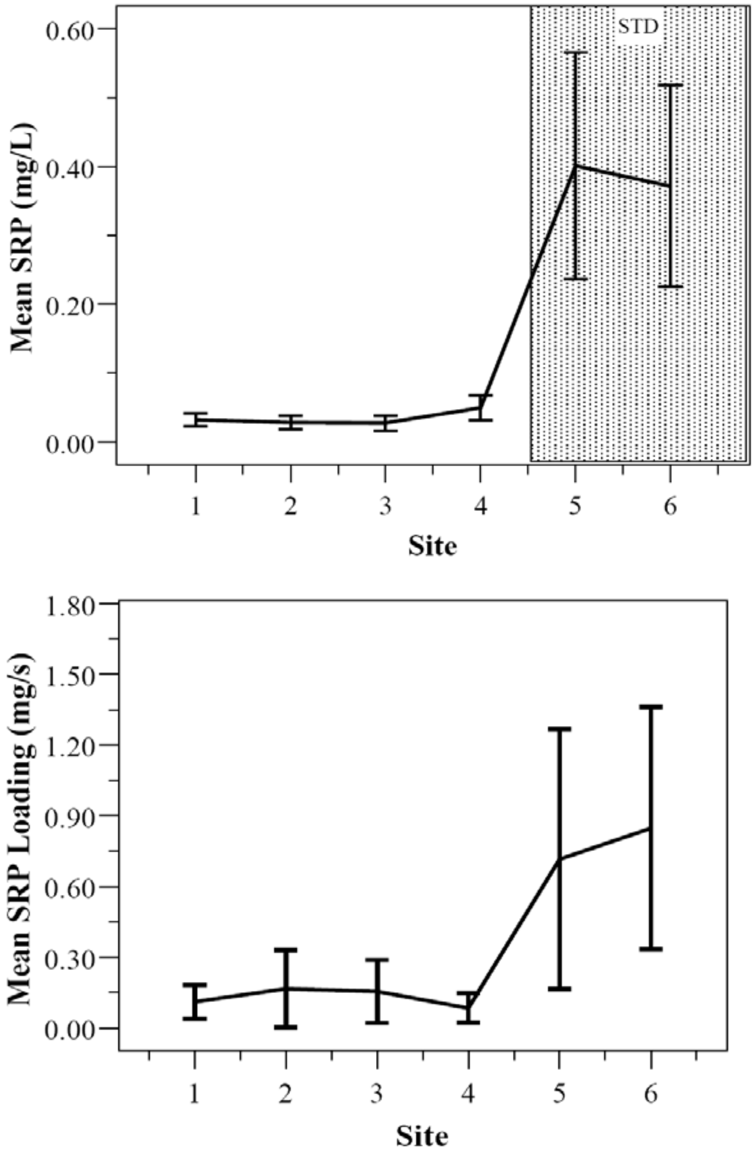

Figure 3.5 Average SRP concentration (upper panel) and load (lower panel) along a short stretch of river within the Wyre catchment showing a marked increase in both values between sites 4 and 5, close to a cluster of about 12 houses served by septic tanks *(after Nicholson, 2007).*

Aerial photography of the area suggests that there are about 12 houses in the immediate area, all within 50m of the stream. If so, the *per capita* P loss to water from this source, assuming the national average of 2.4 people per household, would be about 0.7 kg P y^{-1}. It is unclear whether these septic tanks discharged directly to the watercourse or *via* soakaway.

3.3 Impacts at the catchment scale

In addition to evidence at the site specific level, outlined above, Arnscheidt *et al*. (2007) have shown that the impact of septic tank discharges on water quality can also be detected at the catchment scale. Their study involved a survey of the septic tanks in three rural catchments in Northern Ireland together with high frequency (10 minute intervals) monitoring of TP concentrations at the outlet of each catchment.

Table 3.1 Semi-quantitative scoring sheet to assess the risk of septic tanks causing water quality problems (after Arnscheidt et al., 2007).

Criteria	Factors	Score
Septic system		
Criterion weighting of 2	New GRP biodisk/aeration tank	1–3
	New GRP bottle tank	1–4
Scoring for this criterion is based on structure, condition, type, suitability and age	Concrete tank with filter tank (age/size)	3–7
	Concrete tank with no filter tank (age/size)	4–7
	Engineering brick with filter tank (age/size)	3–7
	Engineering brick with no filter tank (age/size)	4–7
	Block and plaster tank with filter tank (age/size)	3–10
	Block and plaster tank with no filter tank (age/size)	4–10
Method of discharge		
Criterion weighting of 2	Herring bone or 'Y' trenches with no final discharge (area/age)	2–6
	Soak pit	2–7
Scoring for this criterion is based on type of system used, drainage area and age	Herring bone or 'Y' trenches with a final discharge (area/age)	2–8
	No drainage pipe straight to final discharge	4–10
	Drainage pipe straight to final discharge (area)	7–10
Maintenance and operation		
Criterion weighting of 2	Clear access hatches, annual with no odours	1–3
	Clear access hatches, cleaned in <3 years with no odours	2–4
Scoring for this criterion is based on regularity of cleaning, accessibility and odours	No access hatches, cleaned <5 years slight odours	2–5
	No access hatches, cleaned <10 years slight odours	3–7
	Tank not visible, not cleaned in >10 years slight odours	4–9
	Tank buried, not cleaned in >10 years	4–10
Risk of pollution to water course		
Criterion weighting of 4	Very High	9–10
	High	6–8
	Moderate	5–5
Scoring for this criterion is based on the above, accounting for	Low	3–4
distance to permanent water course	Very Low	1–2

Each criterion consists of a number of factors that are scored. This score is weighted and the summation of the four criteria has a potential maximum score of 100. Each score can, therefore be expressed as a percentage. "GRP" is glass-reinforced plastic.

Their results showed that a range of threshold TP concentrations were exceeded more frequently in catchments with higher densities of septic tanks than those with lower densities (Figure 3.6). They also found that more than 60% of these tanks were at high risk of causing water pollution because of their condition, management and location. The level of risk was based on a semi-quantitative method based on a number of expertly predefined criteria that were given weightings according to their importance. Those weightings (shown in Table 3.1) were summed to provide a final score that reflected the potential for each septic tank system to cause water quality problems.

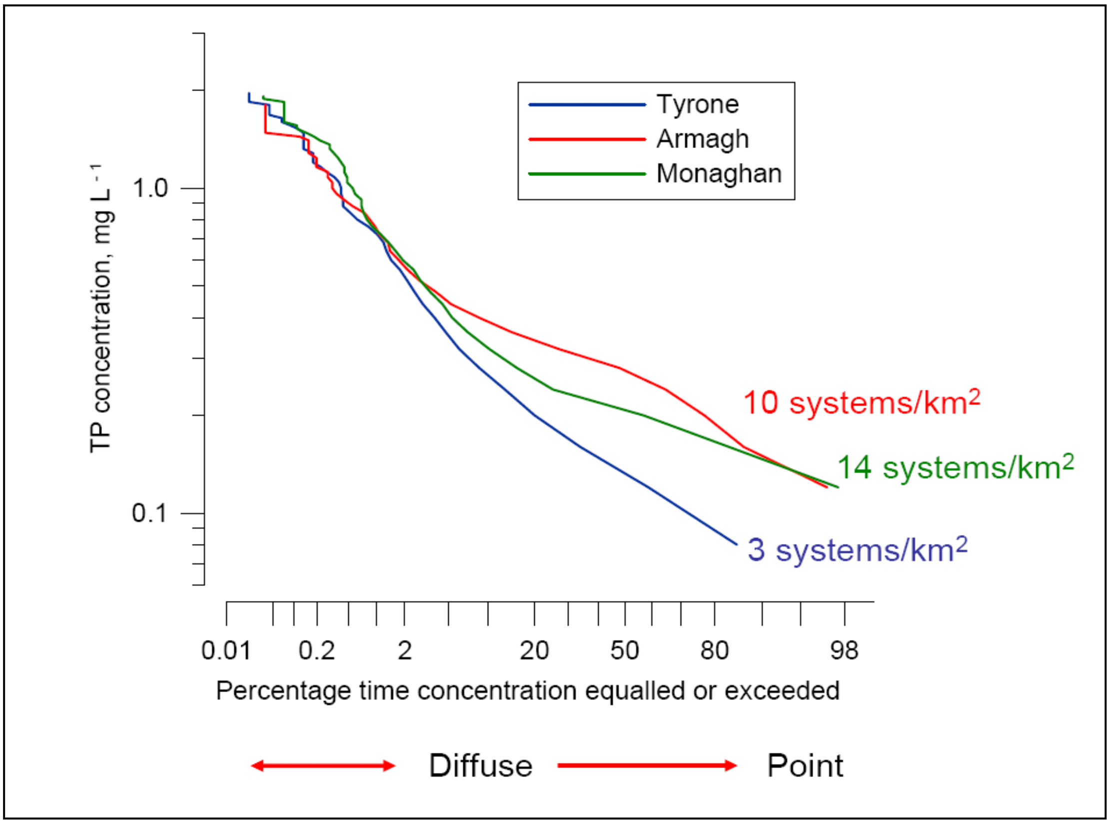

Figure 3.6 Percentage of time in which the TP concentration in the stream at the catchment outlet was greater than or equal to a given threshold concentration in relation to the upstream density of septic tank systems (after Arnscheidt et al., 2007).

The authors also found that the relationship between the median TP concentration in the drainage waters and the ratio of the pollution risk score to the upstream catchment area was similar across all of the catchments (Figure 3.7). These results suggest a link between in-river TP concentrations and the number, condition and management of septic tanks within each catchment.

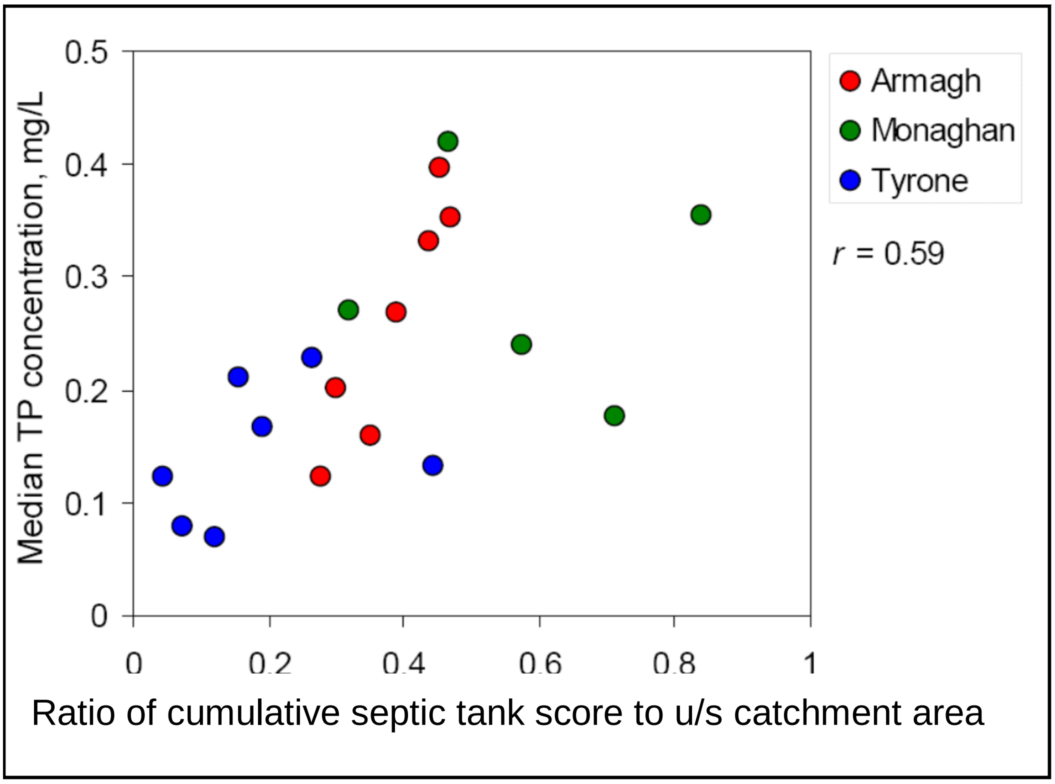

Figure 3.7 The relationship between in-river TP concentrations and the ratio of the pollution risk (score) to upstream catchment area in three catchments in Northen Ireland (after Arnscheidt et al., 2007).

3.4 Impacts on ecology

Increases in P concentrations caused by septic tank discharges will have impacts on the ecology of the receiving waters. Although these impacts are well known at the whole lake or whole river scale, the local impacts of such discharges have rarely been quantified. That said, with local increases in P concentration being as high as those outlined above, it is clear that there will be at least some immediate impact even if discharges become more diluted further downstream. This is a particular problem if the immediate area around the discharge has importance as a SSSI or SAC. Ecological damage can be a very local phenomenon. For example, damage to SSSIs within the River Dee catchment, in Scotland, has been attributed to small discharges of effluent from a small, nearby, WWTW even though there has been little impact on the River Dee, itself (DCMP, 2006). Similarly, the ecological impact of eutrophication on a stream flowing into an oligotrophic mire bog (Morden Bog) near Wareham, Dorset, has also been attributed to small, local, discharges of sewage effluent (see Section 3.4.2, below).

3.4.1 Overview of the impacts of eutrophication on aquatic ecology

The current state of knowledge on the relationship between P concentrations in rivers and impacts on ecology has recently been critically reviewed by Natural England (Mainstone, 2010), and this has been used to underpin a family of P targets for the protection of SSSI rivers. Some of the studies cited that are particularly relevant to the ecological risks posed by septic tanks are summarised below.

The recently completed PARIS project (Defra 2008), which spanned a wide spectrum of water column P concentrations, found a strong relationship between P availability and ecological response in periphyton and macroinvertebrate communities. As P concentrations increased, taxonomic diversity declined and biomass increased. The study also found that community respiration rates on stones, silted leaf litter and gravels increased with increasing P availability, whereas leaf litter decomposition declined. A corresponding shift in macroinvertebrate functional feeding groups, with shredders (mainly Gammaridae) declining largely in favour of fine particle collectors (such as Chironomidae), was also noted.

Defra (2008) also explored the mechanistic role of P in these relationships using laboratory dosing experiments, which generated a gradient of P availability while keeping all other variables constant, and a field analysis of phosphatase activity in the periphyton (phosphatase activity increases when algae have insufficient access to inorganic P and attempt to access P that is locked up in organic complexes, instead). The results indicated that levels of phosphatase activity fell as P concentrations increased up to a threshold of about 90 µg P l^{-1} SRP. Beyond this level, increasing P concentrations did not seem to affect either algal and macroinvertebrate communities. A similar upper boundary on P limitation in rivers was also found by Bowes *et al*., (2007) who developed an *in situ* technique for evaluating the algal response to P availability in streams. This technique comprised replicate in-stream floating channels that are fed with river water that has be subjected to P additions or removals (the latter using ferric sulphate), thus generating a gradient of P availability under otherwise identical field conditions. The authors reported a rapid increase in periphyton biomass as P concentrations increased between 0 and 90 µg P l^{-1} SRP, with little response above this upper value.

However, it is not P concentration alone that generates the effects outlined above. For example, Biggs (2000) found that the periphyton could exploit 'windows' of opportunity between scouring events more effectively at higher nutrient concentrations. This suggests that reducing nutrient supply can help control not only maximum algal biomass but also the frequency and duration of benthic algal proliferations. Similar observations have also been reported by Lohman *et al*. (1992).

Nutrient enrichment can also affect leaf litter decomposition rates in streams (Gessner 2009). This is especially important in pristine stream systems where the trophic structure of the biological community is strongly driven by externally derived (autochthonous) nutrients, especially leaf litter from riparian trees. Anthropogenic enrichment greatly increases leaf decomposition rates, with rates levelling off at low levels of enrichment of around 20-30ug P l^{-1} SRP, reducing the role of shredders and promoting a shift towards autotrophic processes that increase primary production by algae and macrophytes.

Both invertebrate and fish populations can also be affected by eutrophication in rivers through a range of mechanisms that include de-oxygenation and changes in habitat and food supply. McGarrigle (2009) found that rivers with the highest class of ecological status, as determined from the macroinvertebrate community, were those strongly associated with the lowest P concentrations. Graham *et al*. (2009) found that salmon dominated rivers with lower P levels, with trout becoming increasingly dominant in rivers with P concentrations above about 30 µg l^{-1} SRP (40 µg P l^{-1} TP). The effect seemed to be mediated through the food chain, with increased primary production leading to an increase in primary consumption through invertebrate grazers that, in turn, reduced energy expenditure by fish on foraging. The efficient foraging strategy of salmon, which give them a competitive advantage over trout in rivers under lower nutrient conditions, is less of an advantage as nutrient concentrations increase. As a result, the socially dominant brown trout out-compete salmon for territory as nutrient enrichment increases. For a full account of the evidence base on the relationship between ecological change and phosphorus concentrations in rivers see Mainstone (2010).

In addition to the impact on river ecosystems, eutrophication is perhaps the greatest threat to the ecology of shallow, lowland lakes. Several studies (cross-lake, historical and palaeolimnological) have described the lake and, principally, the aquatic vegetation responses to this process., Generally, reductions in species diversity have been documented, with prominent losses of isoetid and low growing taxa (Vestergaard & Sand-Jensen, 2000; Sand-Jensen *et al.,* 2008). In the advanced stages of eutrophication, however, a complete loss of aquatic plants has often been observed (Jeppesen *et al.,* 1998).

3.4.2 Evidence that discharges of P from small point source, such as septic tank, may cause ecological impacts

Ekholm and Krogereus (2003) investigated the potential for P-laden runoff and discharges in rural areas to cause ecological damage, using a series of algal bioassays on waste water from rural populations, urban populations, dairy houses, forest industry, aquaculture and field runoff. The authors found that about 89% of the P in waste water discharged by rural populations was potentially available for algal and plant growth, in contrast to only 16-30% of the P in runoff from land. This is an important issue because it suggests that, weight for weight, the P in septic tank effluent is potentially three to five times more likely to promote algal and plant growth in receiving waters than that associated with runoff from land.

Site specific evidence of the impact of nutrients from rural sewage treatment systems on a stream flowing into an oligotrophic mire bog (Morden Bog) near Wareham, Dorset, can be seen as a darker green plume of colour along the stream in the lower right of Figure 3.8. Inspection of the vegetation in the area revealed a marked difference between the area close to the route of the water that runs into the bog and those areas that were further away. The affected vegetation was found to be much taller than elsewhere with many species characteristic of eutrophication being found, here. The 'plume' was widest where the watercourse first met the bog, narrowing further downstream. An investigation into the causes of this 'plume' concluded that these changes were probably due to treated sewage effluent entering the stream from a nearby park and inn (Kite, pers. comm.).

Figure 3.8 Aerial photography of the area around Morden Bog, Dorset, showing the impact (bottom right) of treated sewage effluent on bankside vegetation along a stream that flows into the bog.

The results summarised in Section 3.2 support the case that effluent from septic tanks and other small discharges can have a marked impact on P concentrations in receiving waters. In the River Wyre study, for example, the average SRP concentration upstream of a cluster of houses served by septic tanks was 49 µg P l^{-1}, whereas that immediately downstream of these properties was 420 µg P l^{-1} (Nicholson, 2007). This represents an increase in SRP concentration of 371 µg P l^{-1} (>750%). Similarly, TP concentrations in the Greens Burn at Loch Leven increased from 61 µg P l^{-1} to 142 µg P l^{-1} on day 1, and 130 µg P l^{-1} to 222 µg P l^{-1} on day 2, when samples were taken upstream and downstream of a small cluster of houses (Brownlee, 2008). This represents an increase of 81-91 µg P l^{-1} (70% – 130%). In both cases, inputs of P from these small discharges appear to have raised the local in-stream P concentration to a value that is significantly above that which is believed to cause ecological impacts (Mainstone 2010).

It is interesting to note that mass input (load) of P from each of the sources outlined above is very similar, i.e. 4 - 5 g d^{-1}. The difference, in terms of impact on P concentration, depends on the rate of dilution which, itself, is determined by the ratio of effluent discharge to river discharge. River flow in the Greens Burn was about 6 l s^{-1} while that in the tributary of the River Wyre was only 2 l s^{-1}, so the discharge was more diluted when it entered the stream. As dilution capacity of the nearby water course is clearly a strong moderator of environmental impact, it is important to take this into account when assessing the likely impacts of new or existing systems at the local level. The possible impacts of climate change on dilution capacity should also be taken into account.

A study of Crystal Lake, Michigan, by Kerfoot and Skinner (1981) illustrates this quite clearly. The authors investigated the local influence of septic tank discharges on plant and algal biomass in receiving waters and found that, although the lake itself was oligotrophic, patches of algae and plants tended to accumulate along the shoreline. These were most prolific in areas around shoreline housing developments, i.e. close to areas with septic tanks. Interestingly, the authors found that this enhanced productivity was more closely related to the level of nutrients in the local groundwater than the total load to the lake. The authors suggested that, to estimate risk at this local level, potential impact should be assessed as P load per unit length of shoreline rather than P load per unit volume or area of lake. Overall, the study estimated that the average P load to the lake from each individual dwelling was 3.6 kg P y^{-1} (or 9 g d^{-1}).

Although the Scottish Executive Environment Group (2005) suggest that P losses from agriculture dominate (52%) diffuse P discharges to water in Scotland, this appears to be in contrast to some catchments in England and Wales. Here, Jarvie *et al*. (2006) found that, even after P stripping at major WWTWs, SRP concentrations in rivers are still dominated by point source discharges. Jarvie *et al*. (2006) concluded that current controls on P are unlikely to yield ecological benefits until smaller point sources are tackled upstream of the point of impact, especially where these effluents discharge into ecologically sensitive tributaries in rural agricultural landscapes.

Mainstone and Parr (2002) investigated the risk to riverine ecosystems from artificially enhanced P loads, focusing on the impact of P enrichment on aquatic plant communities, which are the basis of a healthy and diverse river ecosystem because a wide range of animal species depend on them. The authors concluded that the risk of adverse effects on these ecosystems can be kept to a minimum by maintaining in-river P concentrations as close as possible to background levels. This is especially true in more rural areas where in-river P concentrations are below the upper limit of P limitation 90-100 µg l^{-1} defined by Bowes et al. (2007) and Defra (2008).

4 Estimating losses of P from septic tanks to water at the catchment scale

4.1 Introduction

It is relatively easy to estimate the amount of P that is discharged into septic tank systems from *per capita* values for human excreta (e.g. 1.6 - 1.7 g *per* day- Schouw *et al*., 2002), the usage of P-based detergents (e.g. 0.5 – 1.0 g *per* day – Comber *et al*., 2008), and the percentage contribution of P from various sources to raw domestic sewage (Table 4.1). However, it is much more difficult to determine the amount of P that is ultimately discharged into the environment after processing within the tank and retention within the drainage field. This is because there are few measured values and the level of discharge from these systems depends on site specific details such as:

- the extent to which the source discharges directly to a watercourse (Patrick, 1988)
- the efficiency of the drainage systems in adsorbing P; this depends on environmental factors such as soil type (Ptacek, 1998), level of P saturation (Robertson, 1995), distance to the nearest watercourse (Chen, 1988; Robertson, 1995; Woods, 1993) and level of water-logging of the surrounding soil (Patrick, 1988)
- the extent to which septic tanks are maintained, especially how regularly they are de-sludged
- the nature of the household sewage; this will reflect lifestyle factors such as the extent to which P-rich detergents are used (Alhajjar, 1989, 1990; Harper, 1992) and the level of usage of dishwasher detergent, in particular, which has a high P content (Comber *et al*., 2008)
- the timing of the production of sewage; many properties that use septic tanks are used as holiday accommodation, so septic tank discharges may vary seasonally (Harper, 1992)
- whether the septic tank receives roof runoff; older tanks may be flushed by roof runoff and overflow during heavy rainfall (e.g. in the Loweswater catchment, Norton, *pers. comm*.) whereas, in contrast, more recent building regulations do not permit these systems to be connected to rainwater drainage systems (The Building Regulations, 2000).

Table 4.1 Source apportionment of P in raw domestic waste (Defra, 2008).

Source	Contribution
Faeces	23%
Urine	41%
Food waste	5%
Mains supply (phosphate added to reduce lead in drinking water)	5%
Toothpaste	1%
Dishwasher detergent	7%
Laundry detergent	18%

In spite of these problems, and a general lack of reliable P discharge measurements, many studies have attempted to gain at least some estimate of P discharges to water from septic tanks on the basis of best available knowledge. Various methods have been used for this purpose; these are described below. Most are based on an export coefficient approach, which estimates P discharges from these systems on the basis of an average P loss rate *per* person or *per* septic tank, which is then applied at the catchment or sub-catchment scale. Others are based on more novel methods such as those described at the end of this Section. All of the methods discussed are summarised in terms of their relative strengths and limitations, and gaps and uncertainties in the available data and information are highlighted.

4.2 Export coefficient methods

Export coefficient methods are often used to estimate the P discharge from septic tanks to water. These methods assume a constant (or coefficient) loss of P from this source over time, which is expressed as an amount of P (kg) *per* unit of source (i.e. *per* person, *per* household or *per* septic tank), *per* unit of time (e.g. *per* day or *per* year). So, for example, the amount of P entering receiving waters from a septic tank serving a population of N people would be estimated as follows:

$$P_{septic} = P_{export} \times N$$

where:

P_{septic} = P loss to surface waters *per* septic tank (kg y^{-1})

P_{export} = P loss to surface waters *per* person (kg y^{-1})

N = number of people connected to the septic tanks system

This method is equivalent to that commonly used for estimating nutrient loads from land-based sources, such as agriculture, except that the coefficient is expressed in terms of people rather than area of land drained.

Although the concept of using export coefficients for estimating P losses from septic tanks is common and widespread, methods of calculating the individual coefficients vary considerably among studies. The main limitations on the way that these coefficients are estimated are, generally, the quality of the available data and the scale of the study. Very few studies have determined site-specific export coefficients by actually measuring P losses from septic tanks to water; most have simply used published values from other studies for their calculations (e.g. May & Gunn, 2000; Hall, 2001; May *et al*., 2001; SEPA, 2002; Carvalho *et al*, 2005; SNIFFER, 2006a). Although acceptable at the national or regional level, this approach is of limited use at the site specific level, because export coefficients often reflect the local conditions under which they are determined and are not readily applicable to other sites.

Ideally, a questionnaire-based survey needs to be carried out to determine local factors that affect P discharge and transport at every study site. These factors include frequency of tank de-sludging, distance from a watercourse and method of discharge (i.e. soakaway or directly to watercourse). Survey-based approaches have been used successfully to obtain this type of information in some studies (e.g. Patrick, 1988; Selyf Consultancy, 2002; Kirk *et al*., 2003; Arnscheidt *et al*., 2007).

Once compiled, these data can also be used to evaluate the risk of pollution occurring from each septic tank or group of tanks in the area, as demonstrated by Arnscheidt *et* al. (2007).

A relatively crude method of estimating export coefficients for septic tank discharges at the national scale was used in a study that aimed to estimate all inputs of P to standing waters across Northern Ireland (Smith *et al*., 2005). This coefficient was determined by linear regression of measured in-stream SRP concentration against upstream population density for each sampling site and represented the ratio of the *per capita* discharge of SRP from urban sewage treatment works (WWTWs) to that from septic tanks within the Lough Neagh catchment (Smith, 1977). This ratio, calculated to be 0.58, was then applied across the whole of Northern Ireland (NI). The method assumed that the only non-constant source of SRP in these catchments was the human population, and that all P from sewage related sources was exported as SRP. It also assumed, like all export coefficient methods, a constant export coefficient across the entire study area and over time. Using an export coefficient of 0.44 kg P $capita^{-1}$ yr^{-1}, the authors calculated a total annual load of about 118 tonnes of P from septic tanks to standing waters across Northern Ireland as a whole (Smith *et al*., 2005), which equated to about 5% of the estimated P inputs to standing waters in this country and about 12% of that attributable to effluent from WWTWs.

Although useful at the national level, this method would be of limited use at the site specific level because individual septic tanks and small WWTWs vary in their construction, location, level of maintenance, method of discharge and, consequently, their P loss to nearby waterbodies. The method also provides no information on the location of ‘hot-spots’ although, if populations densities could be quantified at the sub-catchment scale, it would then be possible to identify areas that contribute the most P from human sources.

4.2.1 Application to catchment scale studies

The export coefficient approach can only be used at the catchment scale if the number and location of septic tanks within the catchment are known. However, this information is rarely available and usually has to be derived from data that have been collected for other purposes. Many of these data will not correspond exactly to the area of interest, as spatial data sources are rarely compatible in terms of their scale or geographical coverage. For example, the geographical boundaries of hydrological catchments rarely coincide with those of the main sources of information about people and their properties, such as electoral wards, parishes, counties, regions or countries. So, compromises have to be made. Nevertheless, various methods have been used to derive the number and location of septic tanks within catchments using readily available spatial datasets and these are outlined below.

4.3 Methods of estimating the number of septic tanks

The number of tanks within a catchment is difficult to quantify as they are currently not systematically recorded in official datasets, at least in England. The possible size of the problem can be illustrated by an examination of discharge consent information for the Loch Leven catchment, Scotland, in 2007. Records showed that there were only 18 septic tanks and 6 private WWTWs in this area (Figure 4.1). However, a study by Scottish Natural Heritage (SNH) suggests that the actual number was closer to 650 (Dudley *et al*., 2007). If the Loch Leven catchment were typical of many rural areas of the UK, these results could suggest that perhaps as few as 3% of septic tanks are recorded and their discharges consented.

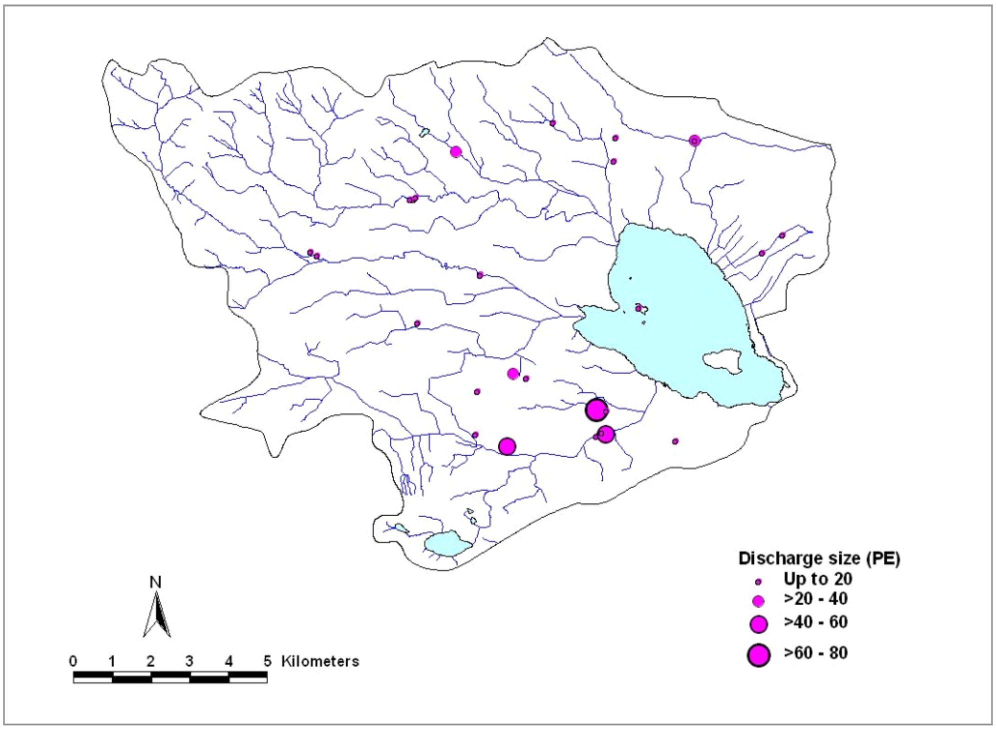

Figure 4.1 Location of consented discharges within the Loch Leven catchment, Scotland, showing size expressed as population equivalent (PE) *(after Dudley et al., 2007).*

4.3.1 The 'postcode' method

One method that has been used for locating septic tanks is the 'postcode' method, which was originally applied to the catchment of Bassenthwaite Lake (May *et al*., 1999; Figure 4.2). This method involves deleting the list of dwellings that pay sewerage connection charges from a master list of all dwellings in the area, making the assumption that the remainder are served by septic tanks. For confidentiality reasons, postcodes were used to approximately locate the dwellings that seemed to be served by septic systems as this ensured that individuals and their properties could not be identified.

This method has potential for widespread use over large geographical areas if appropriate data and information on sewer connections are available and has since been used to approximately locate septic tanks within the catchment of Loch Leven (Dudley *et al*., 2007) and across the whole of Scotland (SNIFFER, 2006a).

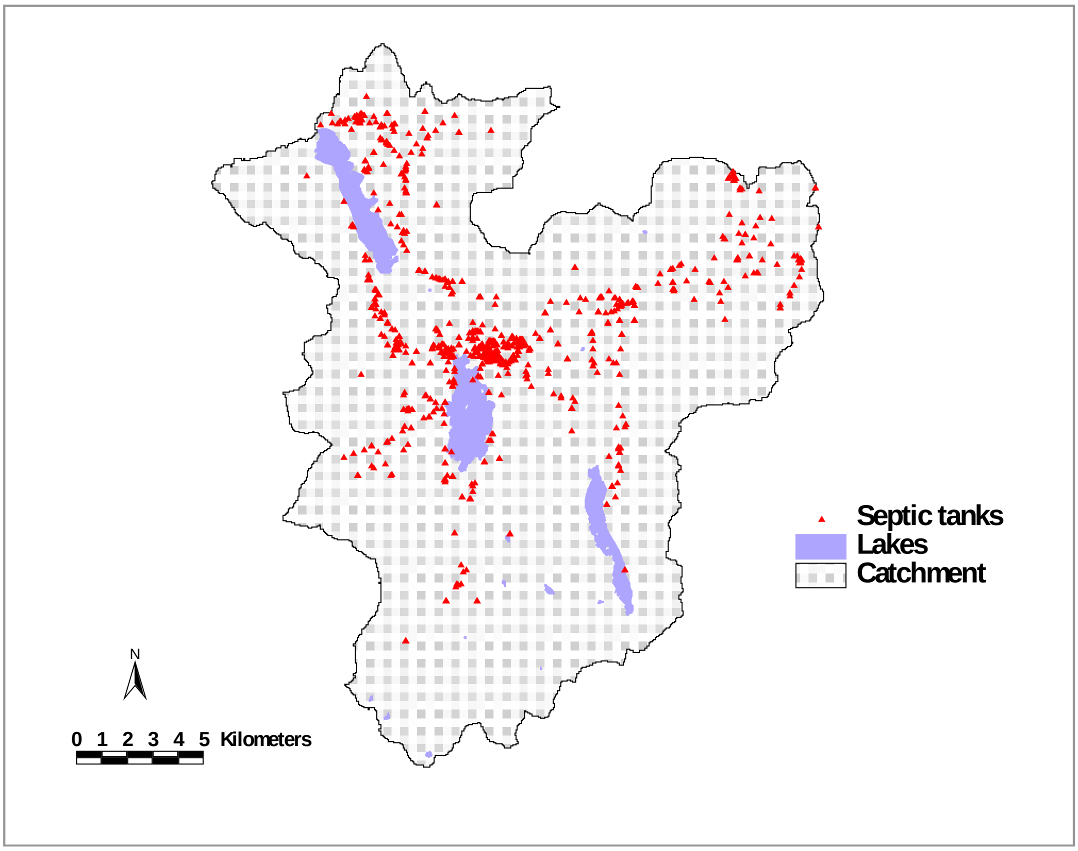

Figure 4.2 Estimated location of septic tanks within in the catchment of Bassenthwaite Lake *(after May et al., 1996).*

4.3.2 The 'sewerage network' method

Another method of estimating the number of septic tanks is described by Hilton *et al*., unpublished. This involves using sewer system network diagrams to derive the area of a catchment that is served by the public sewer system. This method assumes that premises that are outside sewered areas are connected to private sewage treatment systems, such as septic tanks. Although effective, this method may be difficult to use in practice because the utility companies, in some cases, may be unwilling to disclose the necessary information about their sewer networks because of its commercial value and security implications. Also, it cannot necessarily be assumed that all properties within an area served by a mains sewerage system are connected to that system. The authors of this report are aware of properties within such areas that are served by septic tanks because the sewer network is uphill of the property and connecting to it would require pumps to be put in place. Also some owners have chosen to retain their septic tanks because the cost of connecting to a recently installed sewer network, which would involve personal cost to the householder, has been deemed too high.

4.3.3 The 'local knowledge' method

Another example is a map-based method used by May and Gunn (2000), May *et al*. (2001) and Weller (2000) for the relatively small catchments of Lochs Ussie, Flemington and Earn, respectively. In these studies, individual dwellings were identified by eye from a 1:50 000 scale Ordnance Survey Landranger map. Although, this method would be far too time consuming for application to large areas, it is effective at the small or local scale. At this scale, septic tanks can also be located by local knowledge (Figure 4.3; Maberly *et al*., 2006).

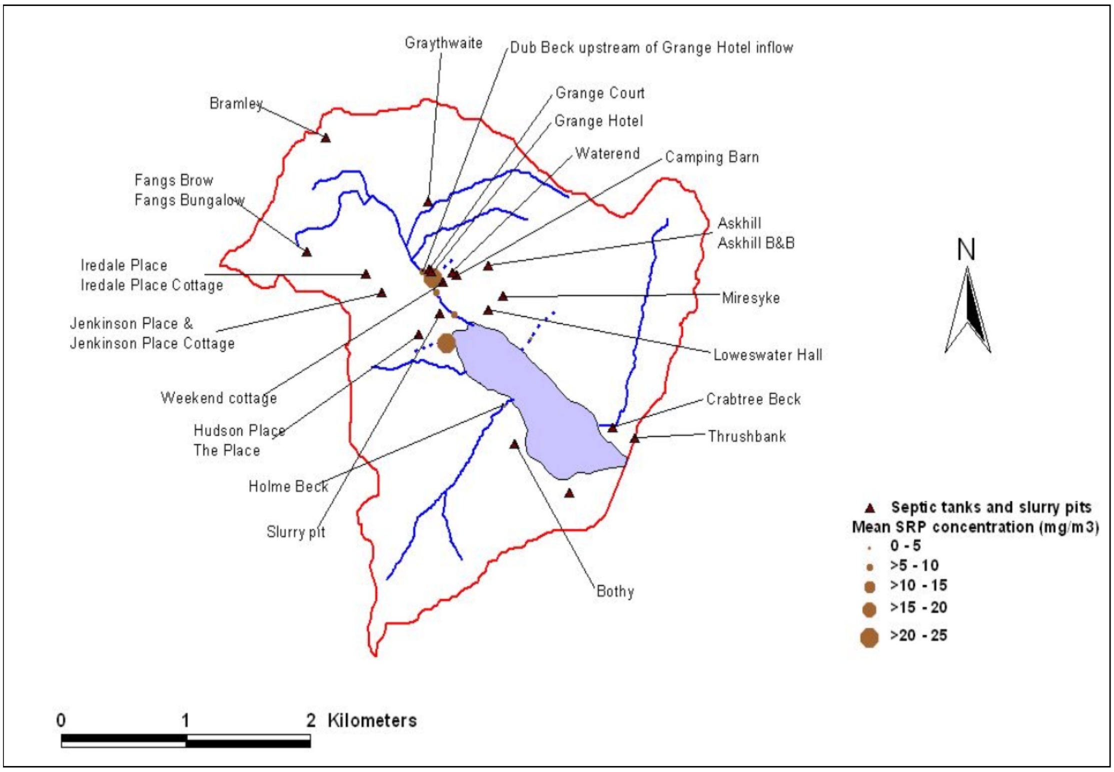

Figure 4.3 Location of septic tanks within in the Loweswater catchment based on local knowledge *(after Maberly et al., 2006).*

In many countries, the population census contains a question about the sewage treatment facilities used by each household. This provides 'local knowledge' on a national scale and is probably the best source of information on unconsented systems, where available. Although this information is available for Northern Ireland and the Republic of Ireland, there is currently no similar source of information on septic tank usage in the census data collected in England, Wales or Scotland. So, at present this method is not applicable to these areas.

4.3.4 The 'large area analysis' method

A recent re-analysis of data compiled by Faber Mausell (2003), Anthony *et al*. (2006) and Stapleton *et al*. (2006), has shown that, at the larger scale, the approximate number of septic tanks in any given area can also be derived from nationally available datasets (Anthony, *pers. comm*.). The data comprised:

1. For Northern Ireland: information on septic tanks usage from 1991 population census returns
2. For Scotland: properties located within Postcode Sectors across Scotland, as derived from an OS Address Point database, and outside of a sewered area
3. For North West England: information on properties known to be using septic tanks from local water company data

These analyses were performed at district council and postcode sector level and the relationship between property density and percentage connection has not been validated for application elsewhere.

Although there are large uncertainties within these data, a clear relationship was found between the percentage of properties that are not connected to mains sewerage systems and the density of properties (Figure 4.4). Although this method does not provide details of the exact locations of individual properties and is probably too coarse for application at a site or catchment specific scale, it does provide a way of estimating the number of properties that are served by on-site sewage treatment facilities at the regional or national scale. This can provide a valuable insight into the likely impacts of P discharges from these sources on water quality at these larger scales and allows areas that are most likely to be most affected to be highlighted.

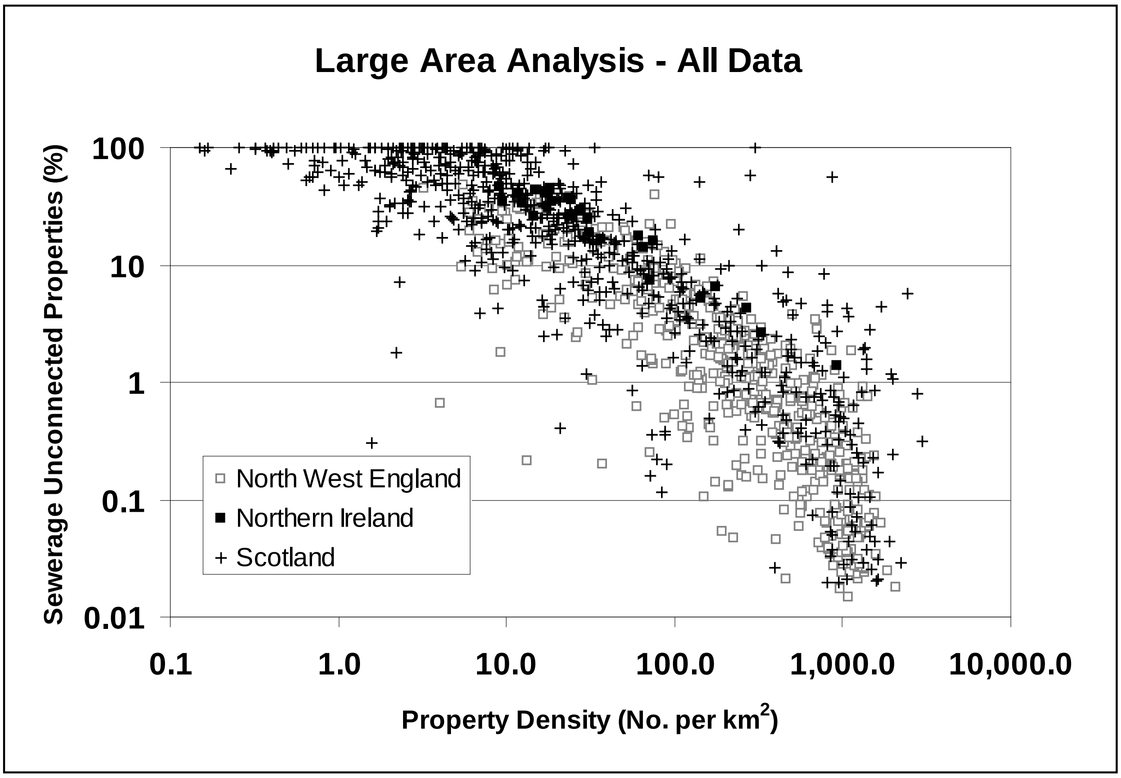

Figure 4.4 The relationship between the percentage of properties that are not connected to mains sewerage systems in parts of the UK and the density of properties in that area (after Anthony, pers comm, ADAS UK Ltd.).

Large area statistics on the the proportion of septic tanks across the UK can also be obtained by integrating water company service data with population census data. The data in Table 4.1, which were calculated by differencing total property counts against an OFWAT (2008) report of the number of properties serviced for sewage by each water company, suggest that about 1.2 million properties in England and Wales are served by septic tanks.

Table 4.1 The number of properties and percentage that are served by septic tanks in areas covered by different water services providers within England and Wales (Anthony, *pers comm*, ADAS UK Ltd.).

Service Company	Properties (Million)	Septic Tanks (%)
Anglian	2.4	0.5
Northumbrian	1.2	1
Severn Trent	3.7	4.9
South West	0.6	11.7
Southern	1.8	7.9
Thames	5.3	7.3
United Utilities	3.1	1.9
Welsh	1.3	8.4
Wessex	1.1	12
Yorkshire	2.1	0.2
England and Wales	**22.7**	**5.1**

4.3.5 Improved record keeping

Under the Control of Pollution Act 1974 (CoPA) all sewage discharges to surface waters in Scotland required consent from the Scottish Environment Protection Agency (SEPA). However, in most cases, there was no requirement for consent from SEPA for sewage discharges to soakaway. This led to incomplete records of septic tank locations. Since 1 April 2006, there have been significant changes to the control of sewage discharges following the introduction of the Controlled Activity Regulations (CAR) in 2005. Under these regulations, all new sewage discharges from domestic properties serving a population equivalent (PE) of ≤ 15 (one house of three or less bedrooms is taken to be 5 PE) will need to be authorised by registering with SEPA. This includes all sewage discharges to soakaway. For population equivalents of more than 15, a licence is required. SEPA discourages direct sewage discharges to rivers, lochs, estuaries or coasts. However, where ground conditions are not suitable for soakaways, SEPA will consider approving discharges to surface water if these are environmentally acceptable. SEPA requires such discharges to be registered, as with discharges to soakaway, so that the locations of all new discharges are known.

A recent change to legislation within Scotland aims to address the problem of unconsented discharges retrospectively at the national scale. Since April 2006, all septic tanks must be registered with the Scottish Environment Protection Agency (SEPA) when properties change ownership. This, over time, will create a record of the size, location and discharge of all septic tanks in Scotland (SEPA, 2006). To speed up registration, this process was supplemented by a registration fee waiver for a limited period (November 2008 to May 2009) to encourage earlier registration of existing discharges. This campaign proved very successful, with SEPA receiving more than 50,000 new registrations of septic tanks and small sewage discharges over this six month period. Most of these were from the rural north of Scotland.

In April 2010, England and Wales introduced a compulsory registration system for septic tanks. This was later suspended in England, pending a government review of the process, but was continued in Wales.

5 First time sewerage schemes

5.1 Background

The Water Industry Act of 1991 aimed to consolidate the supply of water and the provision of sewerage across England and Wales. Section 101a of the Act charges public sewerage undertakers to provide access to public sewers where environmentally and economically justified if:

- the drainage of premises not connected to a public sewer is giving rise, or is likely to give rise, to adverse effects on the environment or amenity;
- the actual or likely adverse effects are from more than one building;
- the drainage of those premises is for 'domestic sewerage purposes';
- the relevant premises are not currently connected to a public sewer; and
- provision of a public sewer is the most appropriate solution

These additional responsibilities came into force in 1996 and were designed to address environmental and amenity problems associated with rural on-site sewage storage/treatment systems (e.g. septic tanks).

5.2 Funding

Funding of improvements required under the Act is the responsibility of the sewerage undertakers and several have announced significant financial investment in this process. For example, Anglian Water has announced that it is investing £70M between 2010 and 2015 to connect 2,970 rural properties to a new sewerage system. (http://www.anglianwater.co.uk/news/planned-investments/7F0FDE1BB47149FE8BFD357E63DBA81D.aspx). However, funding to connect properties to new (first time) sewerage systems applies to domestics properties, only. The costs incurred in connecting utility buildings to the public sewer (estimated to be about £3k; McMahon *et al*., 2000) must be met by the applicant. Exact costs vary, depending on the sewerage undertaker concerned.

5.3 Applications procedure

Applications for the installation of first time sewerage schemes are invited from groups of local residents or relevant local authorities on behalf of groups of residents. Applications are submitted to the relevant sewerage undertaker. A guidance note on the application process has been published by Defra and the Welsh Assembly Government (DoE & Welsh Office, 1996) and most sewerage undertakers also provide their own specific application guidelines and time scales. The Environment Agency (EA) is responsible for resolving any disputes that arise during the application process. Only houses built before 20 June 1995 are considered for funding under this scheme and applications must involve more than one household or building.

Completed applications are considered in liaison with local environmental health officers and the EA and in relation to a range of criteria. These include an assessment of options, costs and benefits in accordance with government guidelines. An application will be successful if the assessment confirms that on-site sewage treatment facilities are causing environmental or amenity problems and that connection to a public sewer is the most cost effective way of providing suitable drainage. However, if it is concluded that the most cost effective solution to the problem is the improvement and better maintenance of the on-site system, this remains the responsibility of the owner. Even if the need is agreed, it can be several years before an agreed new sewerage scheme is implemented. At the present time, it is unclear how many such improvement schemes have been approved and how many have been implemented.

5.4 Evidence of effectiveness

There are few robust data available with which comparisons can be made between the impacts of on-site sewage treatment facilities and those of first time sewerage schemes on water quality in receiving waters. Nevertheless, there is a general assumption that water quality will be improved by these schemes. This assumption appears to be supported by Barden (2007), who documents changes in water quality in the River Chew following the implementation of a first time sewerage system in December 2002. This system replaced on-site sewage treatment systems that served a local population of about 500 people, two public houses and a primary school, and transferred the effluent from the Litton and Chewton Mendip catchment to the Midsomer catchment where it is now treated by a sewage treatment works at Radstock. Water quality monitoring data from the River Chew at Litton, before and after this system was introduced, show a marked and sustained improvement in quality with OP concentrations decreasing from an average of 251 µg P l^{-1} to an average of 86 µg P l^{-1} since the new system was installed (Figure 5.1). The exception to this was two high OP events that were recorded in late 2005 and mid 2006. The explanation for these is unclear, but they may have been caused by rainfall driven runoff events relating to the decommissioned septic tanks. Overall the decrease in annual mean OP concentration achieved was about 200%.

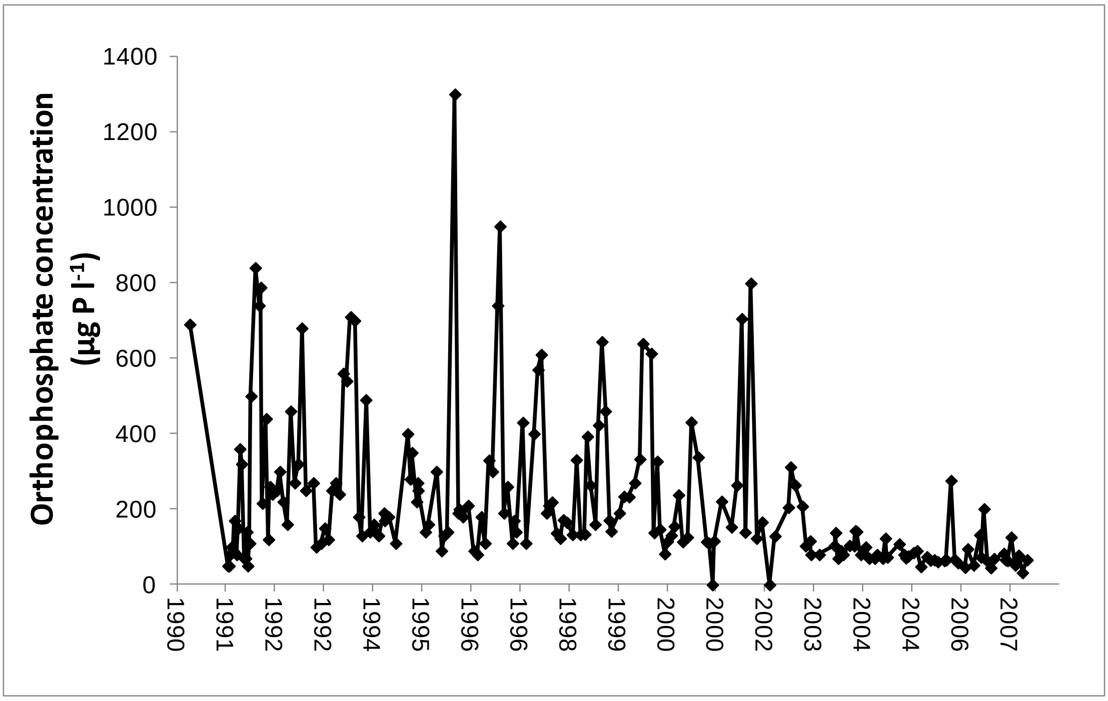

Figure 5.1 The impact of a first time sewerage scheme on orthophosphate concentrations in the River Chew at Litton (after Barden, 2007).

The results presented by Barden (2007) strongly suggest that replacing septic tanks with first time sewerage schemes in rural areas is very effective at reducing P impacts on receiving waters. However, it should be noted that this scheme removed all of the sewage-related P from the properties affected into another catchment, so there was no longer any impact on the River Chew at Litton. The overall impact of such schemes can only be properly assessed if the impact on the receiving waters where the effluent is diverted to is also taken into account.

There are also some wider issues to be addressed in relation to the replacement of on-site sewage treatment systems with first time mains sewerage schemes. These are focused on the high volume of water that is required to move or treat water in conventional WWTWs in comparison to that required by on-site systems. It has been suggested that, with increasing demand for water, the water usage for sewage treatment must be reduced (Bakir, 2001). For this reason, septic tanks could provide a better option for sewage treatment in areas where there are likely to be water shortages in future, especially due to climate change, and it might be better to focus more attention on improving their functioning than replacing them with WWTWs in the longer term.

6 Case studies: estimating the number and influence of septic tanks in the Hampshire Avon and the Broads catchments

6.1 Introduction

Now that the P discharges from larger point sources are being reduced at many locations, attention is being focused on P inputs to waterbodies from other sources in rural areas. In the past, such inputs have been attributed mainly to agricultural sources. However, it has now been recognised that part of this ‘diffuse’ P load may be associated with sewage effluent from the many properties across rural catchments that are not served by mains sewerage systems and are, therefore, connected to septic tanks[1]. It is difficult to estimate the size of this contribution because the number and location of these tanks is generally unknown. Also, the level of discharge from these sources to water depends on many site specific issues such as mode of discharge (whether the effluent discharges to water directly or indirectly, e.g. *via* a soakaway), distance to the nearest watercourse and the hydrological connectivity of the catchment.

The impact of these factors on P transport is not well understood. Many studies have shown that, if septic tanks discharge to soakaway, a high proportion of the P in the effluent is removed in the first 30 - 100 centimetres of soil that it passes through (Jones & Lee, 1979; Harman *et al*., 1996; Robertson & Harman, 1999; Sawney & Starr, 1977; Zanini *et al*., 1998). However, because initial concentrations are so high, the remaining P in the effluent plume 50 – 100 m from the original source can still be high enough to pollute receiving waterbodies. Wieskel & Howes (1992) estimated that only about 0.3% of the original P content of the effluent would reach a waterbody 100 m from a septic source. A similar result was obtained by Chen (1988), who measured P concentrations at distances of 40 m and 100 m from a septic tank and found P concentrations of 0.1 mg l^{-1} and 0.04 mg l^{-1} in the effluent plume, respectively, at these distances. These values, however, refer to properly sited and maintained septic systems. It has been found that, within parts of the UK, septic systems are often improperly sited and rarely emptied. In Northern Ireland, for example, Gill *et al*. (2007) found that about 95% of the septic tanks examined were located too close to the water table, while Jordan (*pers. comm*.) found that 50% of those inspected elsewhere in Northern Ireland failed percolation tests. In southern Ireland, another study found that up to 90% of the septic tanks surveyed were full of sludge and had rarely, if ever, been emptied (Kirk *et al*., 2003). Accumulation of sludge reduces the storage capacity of the tank, causing it to overflow. Although there are few documented studies of the siting and/or management of septic tanks in other parts of the UK, the survey conducted by Selyf Consultants (2002) in North Wales and other anecdotal evidence suggests that many may not be properly sited or maintained, and that this can cause local water quality problems.

[1] Septic tanks and package treatment plants were not distinguished in this report because detailed, site specific data were not available to make this determination.

This part of the project assesses the scale and nature of the input of P to sensitive waterbodies from septic tanks within two designated freshwater sites in England:

1. The upper Hampshire Avon

2. The Broads

For these case studies, the P output from consented and unconsented small discharges has been calculated and their combined impact on the P concentrations of the receiving waters estimated. Case study 1 focuses on two rural subcatchments of the Hampshire Avon, i.e. those of the rivers Nadder and Wylye. Case study 2 focuses on two rural subcatchments of The Broads, i.e. those of the rivers Ant and Bure. The estimated P inputs to these waterbodies from septic systems have been compared with those believed to come from sewage treatment works and agricultural sources.

Table 6.1 Key assumptions made in estimating worst case scenarios for P output from consented and unconsented small discharges.

1.	All of the P exported from septic tanks to the environment eventually reaches a watercourse.
2.	Households outside of sewered areas us on-site waste water treatment facilities, such as septic tanks, to treat domestic waste.
3.	The majority of tanks are likely to be poorly maintained (Kirk *et al*., 2003; Arnscheidt *et al*., 2007) and may not be working effectively (Gill *et al*., 2007).
4.	None of the tanks receive roof runoff, which can cause tanks to overflow during heavy rainfall.

It is important to stress, however, that the calculations and conclusions presented in this chapter are based on worst case scenarios. Further information on the assumptions made is provided in Table 6.1, and more specific information is provided within each case study. To refine these calculations, further research is needed on the factors that affect the loss of P from small domestic discharges and the extent to which P is then trans-located to water bodies. Some limited work in this area has been conducted in North America, but it is inappropriate to assume that this can be applied directly to the UK situation where soil types and climate are different. A follow on project, also funded by Natural England, is now looking at the capacity of soil soakaways to remove P from discharged effluent before it reaches waterbodies within England.

Recognising the need to conduct more realistic risk assessments in the future, and prioritise action across sites, a preliminary screening technique has been developed to help identify those septic tanks that are at greatest risk of contaminating nearby watercourses. This uses a combination of catchment level and site specific data and information (Appendix 1). However, this approach requires further development and validation before it can be applied usefully in a real world context. This should include the collection of field data.

6.2 Case study 1: The upper Hampshire Avon

6.2.1 Site description

This part of the study focuses on two well documented sub-catchments of the upper Hampshire Avon, i.e. those of the River Nadder upstream of Wilton (flow gauging station at NGR 409800, 130800) and of the River Wylye upstream of South Newton (flow gauging at NGR 408600, 134300; P concentrations: NGR 408821, 133844). Within the Wylye catchment, the Chitterne Brook subcatchment, upstream of Codford St Mary (NGR 397400, 139600), which constitutes about 10% of the Wylye catchment, receives particular attention. This is because detailed flow and water chemistry monitoring data are available for this stream near its confluence with the River Wylye and because the stream itself receives no direct inputs of treated effluent from upstream WWTWs. The only WWTW within this subcatchment is very small, serving only 0.02% (0.012 km^2) of the subcatchment by area, and discharges to a floodplain soakaway (Jarvie *et al*., 2006, 2008). Households in the remainder of the subcatchment use on-site waste water treatment facilities, mainly septic tanks, to treat their domestic waste water.

The Nadder catchment is 216 km^2 in area, while that of the Wylye is 448km^2. Both are very rural in character (Table 6.2), with only 17% and 5.5% of their area, respectively, being served by mains sewage treatment facilities (Figure 6.1). For the purposes of this study, it has been assumed that households outside of these areas have on-site waste water treatment systems, such as septic tanks.

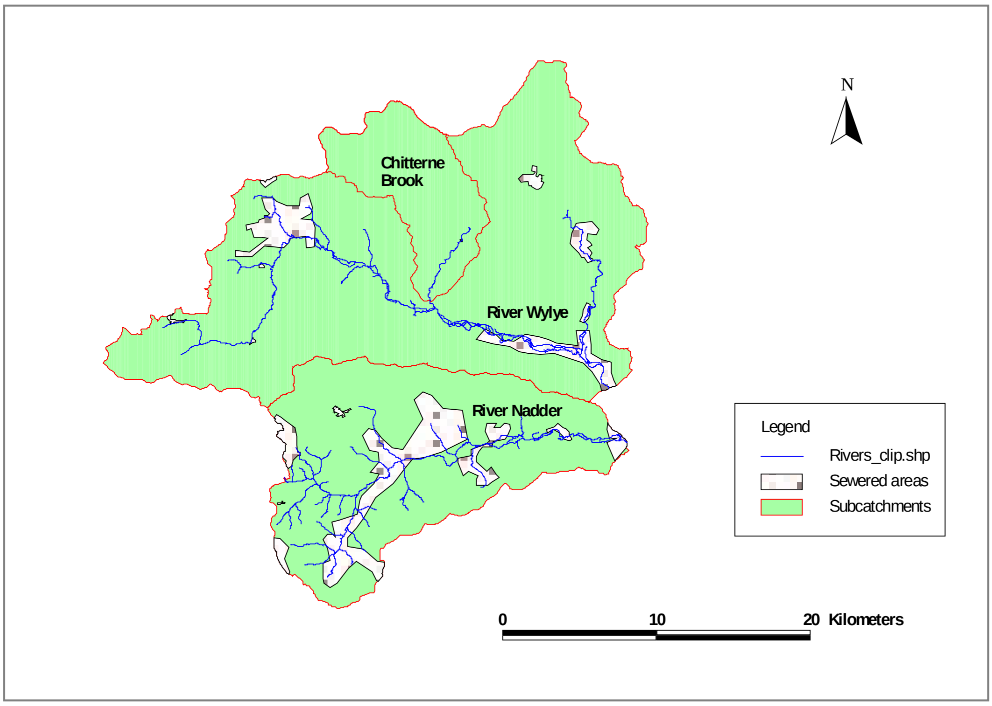

Figure 6.1 The Nadder and Wylye subcatchments of the upper Hampshire Avon showing areas served by mains sewerage systems. The Chitterne Brook subcatchment, which is part of the Wylye catchment, is also shown.

Table 6.2 Characteristics of the Wylye and Nadder catchments, and the Chitterne Brook subcatchment.

Catchment	Area (km^2)	Sewered area (km^2)	Landcover (%)				Reference
			Arable /crops	Improved grassland	Rough grazing	Other	
Nadder	216	37.5	39	28	6	27	Jarvie *et al.*, 2008
Wylye	448	25	76	24	0	0	German & Sear, 2003
Chitterne	68	0.012	33	47	15	5	Jarvie *et al.*, 2008

6.2.2 Methods

As there was no information available on the number and position of septic tanks within these catchments, this was derived from the location of residential properties ('houses') that were within these catchments but outside of the areas served by mains sewerage systems. Each unsewered 'house' was identified by eye from a 1 m resolution aerial photograph of the area that had been derived from 25 cm resolution images supplied by Natural England.

For this exercise, 'houses' were distinguished from farm and other non-residential buildings mainly by the presence of chimneys on their roofs and/or areas of garden to front and back. Where these 'houses' formed part of a terrace, the number of properties in each terrace was estimated from the number of garden plots separated by walls, fences or hedges and/or the number of garden paths leading to the front or rear of the building. The same criteria were applied to semi-detached properties but, in addition, these could often be distinguished by the number of garages or driveways associated with each 'house'.

The location of each 'house' was digitised from the aerial photography using Erdas Imagine® software and it was assumed that each 'house' outside of a sewered area was connected to a septic tank. Septic tanks and package treatment plants were not distinguished in this analysis, although based on wider evidence, it is likely that the majority of these dwellings were served by septic tanks. The location of these 'houses' is shown in Figure 6.2. For comparison, the location of all discharges from on-site sewage treatment facilities (serving single or multiple 'houses') that have been consented by the Environment Agency is shown in Figure 6.3.

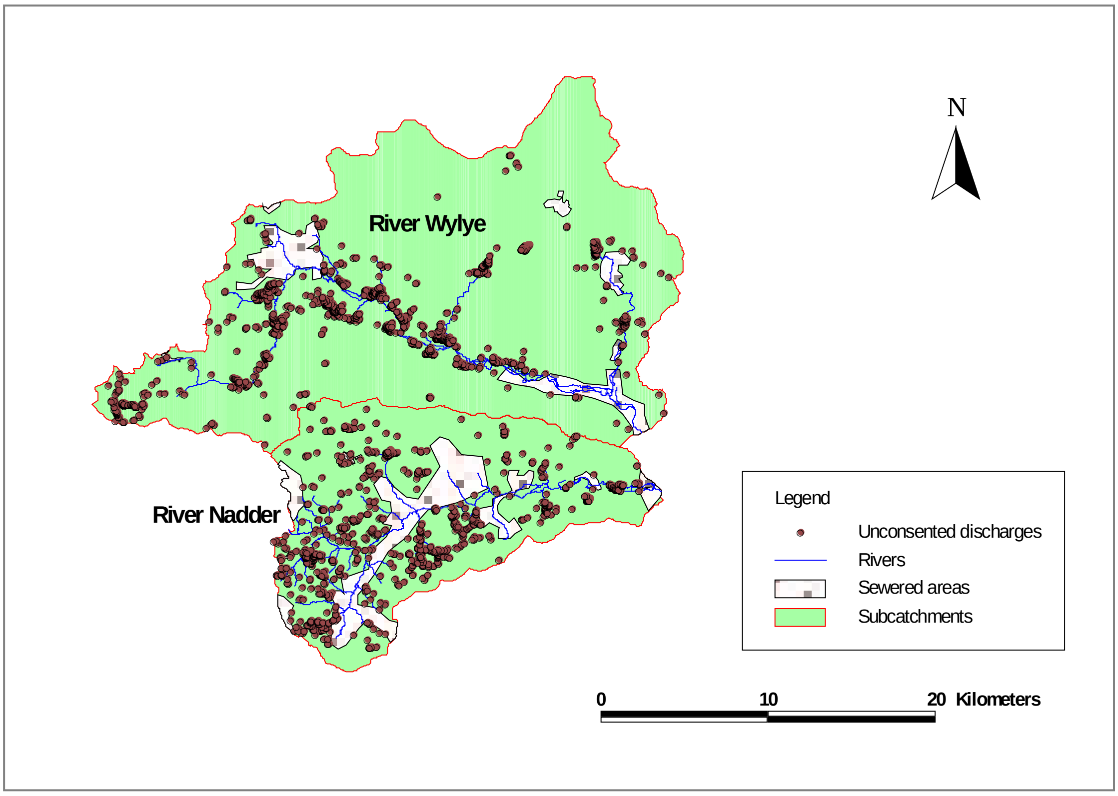

Figure 6.2 Map of the Nadder and Wylye subcatchments of the upper Hampshire Avon, showing the location of unconsented septic tank discharges.

The potential worst case P input to the rivers draining these catchments from consented and unconsented septic tank discharges was estimated based on the assumptions in Table 6.1. These include the assumption that the P load from all of these tanks was translocated to the watercourse. Whilst this is probably over-estimating the actual risk posed, it is noteworthy that Montgomery *et al*. (1984) found no reduction in the OP concentrations in septic tank effluent over at least the first 2m depth of soil drainage fields in the chalk areas of Kent, which he suggested was possibly due to rapid travel time due to percolation through fissures in these types of soils.

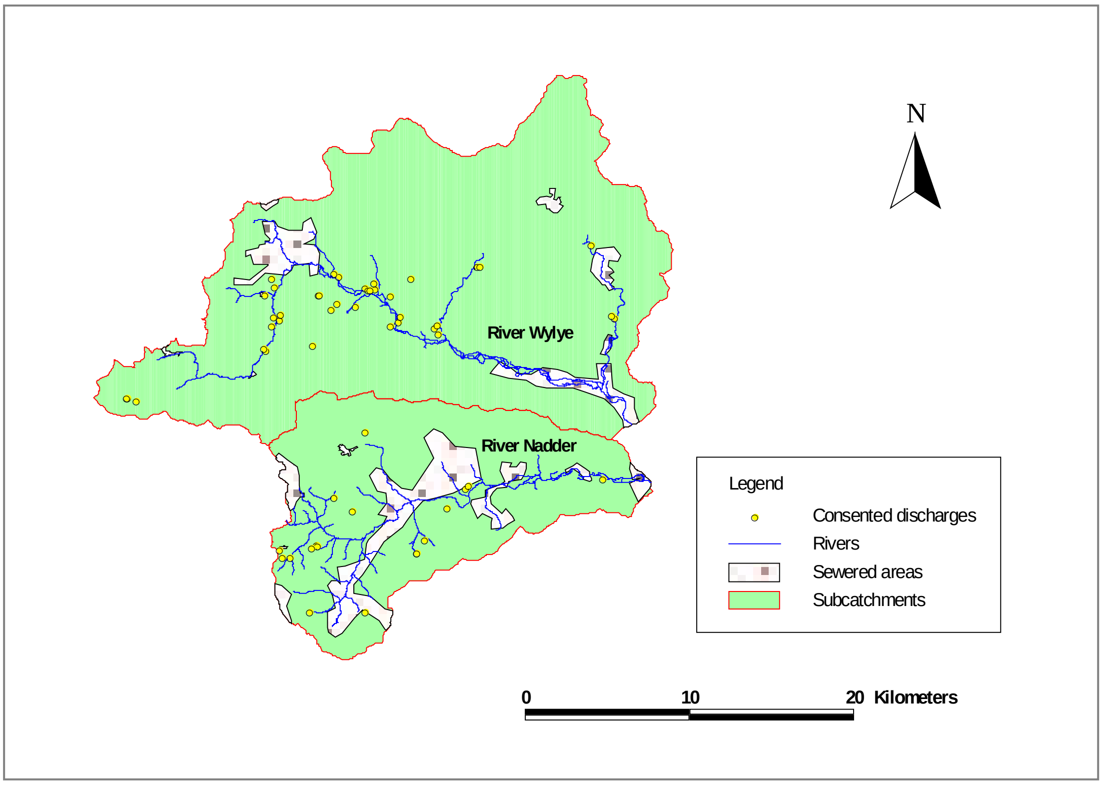

Figure 6.3 Map of the Nadder and Wylye subcatchments of the upper Hampshire Avon, showing the location of consented septic tank discharges.

To estimate the P losses from septic tanks associated with each household, the average number of people *per* household within the local authority area of East Hampshire (i.e. 2.4 - www.neighbourhood.statistics.gov.uk) was multiplied by the average *per capita* volume of water used each day in this area in 2007-2008 (i.e. 148 litres – www.defra.gov.uk) and by the most commonly reported concentration of P in septic tank effluent (i.e. about 10 mg l^{-1} – see Section 2.2.2). This equated to an annual *per capita* P export from each tank of about 0.54 kg y^{-1} (1.3 kg $household^{-1}$ y^{-1}), which falls within the ranges of values given by other authors for P discharges from septic tanks to water (Table 6.3). In doing these calculations, it was assumed that none of these tanks received roof runoff, which can cause systems to overflow during heavy rainfall, although it is likely that some of the older systems still receive water from this source. To estimate P losses at the catchment scale, those losses from each household were multiplied by the estimated number of households in each catchment or subcatchment. Where appropriate, these values were compared to the in-river P load at the catchment outflow point to determine the relative importance of this source in comparison with discharges from upstream WWTWs and runoff from agricultural land.

Table 6.3 Estimated *per capita* discharge of P from septic tanks to water from published sources.

***Per capita* P discharge**	**P discharge per household**	**Reference**
0.24 – 0.4 kg P y^{-1}	0.6 – 1.0 kg P y^{-1}	Foy & Lennox (2000)
0.3 kg P y^{-1}	0.7 kg P y^{-1}	SNIFFER (2006a)
0.3 kg P y^{-1}	0.7 kg P y^{-1}	Carvalho *et al*. (2005)
0.63 – 0.72 kg P y^{-1}	1.5 – 1.7 kg P y^{-1}	Pieterse *et al*. (2003) – direct discharge to water
0.69 kg P y^{-1}	1.7 kg P y^{-1}	SEPA (2002)

Water quality data and rates of discharge measured at the outflow points of the Nadder and Wylye catchments were supplied by the Environment Agency (EA). Those for the Chitterne Brook were provided by CEH/ADAS. The data were examined to determine annual discharges and TP loads at these sites. Particular attention was paid to estimating the importance of TP discharges from septic tanks in determining downstream water quality in the Chitterne Brook, because this is the only part of the study area where all of the sewage effluent is discharged to soakaway. Elsewhere, there are also large WWTWs that discharge directly to the watercourses.

As no P discharge monitoring data were available for small sewage works across the catchment, the level of P discharge from these works was estimated from the relationship between the dry weather flow (DWF) discharge consent and estimated P load for other small WWTWs across the catchments, i.e. those at Bardford St Martin, Tisbury, Fovant and Great Wishford. It was assumed that the level of treatment, and consequently the level of P removal, was the same for all small WWTWs. The relationship between DWF and P discharge for these small works (Figure 6.4) was estimated to be:

$$P\ load\ (kg\ P\ y^{-1}) = 2.24 \times DWF + 211$$

Where:

P load = P export from sewage works (kg P y^{-1})

DWF = dry weather flow ($m^3\ d^{-1}$)

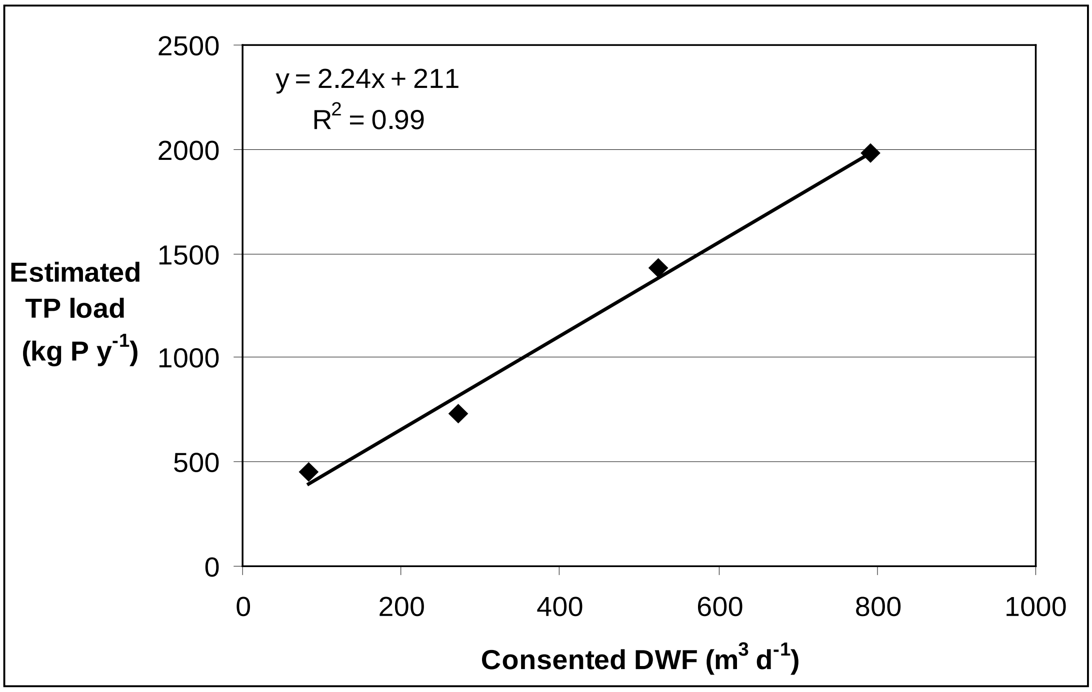

Figure 6.4 Relationship between estimated TP discharge and consented dry weather flows for small sewage work within the Nadder and Wylye catchments.

6.2.3 Results

The number of individual dwellings that were digitised from the aerial photography of the Nadder and Wylye catchments, and of the Chitterne Brook subcatchment, is shown in Table 6.4. In total, an estimated 3063 households were found to be using on-site sewage treatment facilities without a discharge consent. This compares to 243 discharges that do have consents. Overall, the density of unsewered households in these catchments was about 5.8 km^{-2} in the Nadder catchment and 4.6 km^{-2} in the Wylye catchment.

Table 6.4 Estimated number of unsewered households within the Nadder and Wylye catchments and the number that have discharge consents for their on-site sewage treatment facilities.

Catchment	Unsewered households	
	Total	Consented
Chitterne Brook	174	22 (12.6%)
Wylye (excluding Chitterne Brook)	1875	160 (8.5%)
Nadder	1257	63 (5%)
Total	**3306**	**243** (7.4%)

6.2.3.1 Chitterne Brook

The catchment that drains to the Chitterne Brook at Codford St Mary covers an area of about 68 km^2. Within this, a total of 174 houses were identified as being outside of the sewered area, with 37% of these being situated within 100 m of the stream and

75% being situated within 300 m of the stream (Figure 6.4). Of these, it is estimated that only 22 (13 %) are connected to septic tanks that have discharge consents.

The geology of this area is mainly cretaceous chalk, which may not be ideal for the siting of septic tank systems. This is because this type of soil has been found to provide little retention of P due to rapid travel times associated with percolation through fissures according to Montgomery *et al.* (1984). These authors found that there was no reduction in OP concentrations in septic tank effluent over at least the first 2m depth of soil in soakaways in chalk areas of Kent.

In addition to the problems associated with soil type, the fact that most of the tanks are within 100 m of the stream (Figure 6.5), mainly along the floodplain of the river, suggests that many of these tanks are probably sited too close to the water table to function correctly (see Section 6.1). Anecdotal evidence from at least two parish newsletters on the Chitterne village website (http://www.chitterne.com) suggests that several of these tanks become inundated during high flow events, causing the contents to spill into the local drainage channel, which connects to the river. It is interesting to note that local residents, having recognised the risk to water quality of their tanks overflowing under flood conditions, had their tanks emptied to reduce the level of pollution and purchased chemical toilets for use until the flood waters receded. However, having taken this initiative, they were unable to obtain help in finding a safe place to empty these chemical toilets while the problem lasted.

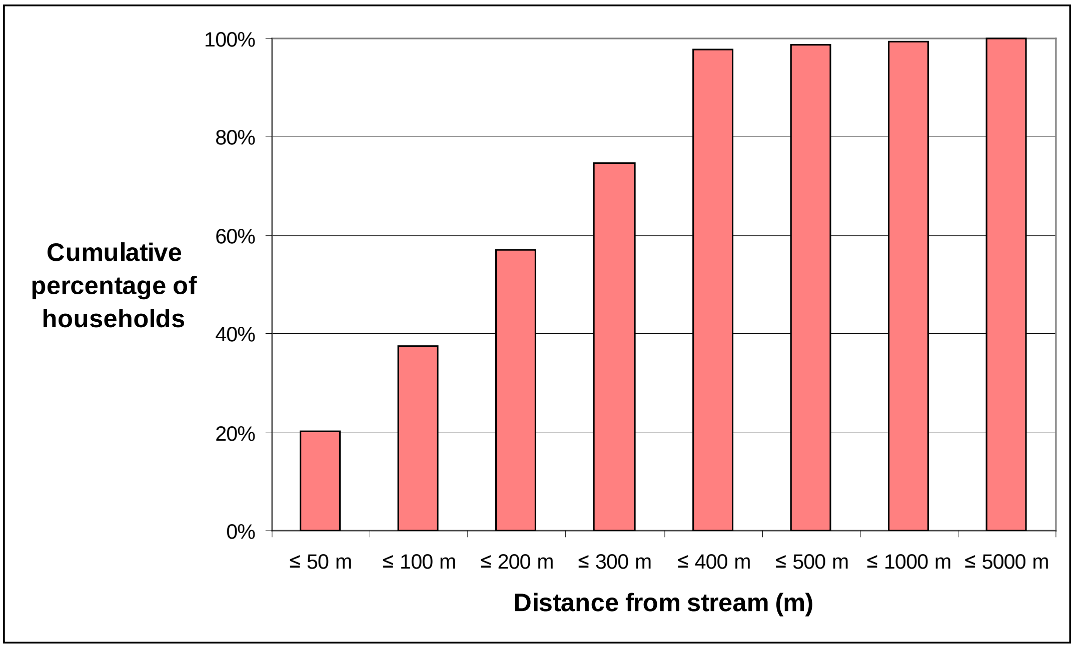

Figure 6.5 Cumulative percentage of unsewered households that are within the distance shown from the Chitterne Brook.

The fact that all sewage treatment facilities within this catchment discharge to groundwater, that most septic tanks are on the floodplain within 400 m of the stream, and that many may be inundated by floodwater during high flow events, suggests that this site will be significantly affected by discharges from these systems. This hypothesis was explored using monitoring data that had been collected at the outflow of this subcatchment for another study between August 2002 and November 2003 (Jarvie *et al.*, 2008). These data comprised weekly records of streamflow and corresponding measurements of P and boron (B) concentrations collected using sampling and analytical methods described in detail by Jarvie *et al*. (2008). Boron measurements are important in this context, because this chemical can be used as a conservative tracer of sewage effluent (Neal *et al*., 1998). This is because, until recently, boron entered waste water treatment systems as a component of domestic laundry detergents.

The data clearly show that in-stream loads, calculated as instantaneous flow multiplied by concentration, of both P and B increased dramatically, and in parallel, during periods of high winter flows (1 - 6 m^3 s^{-1}, November to March – see Figure 6.6), but remained relatively low during periods of low flow (0.05 - 0.07 m^3 s^{-1}, July to October – see Figure 6.6) . In general, it was found that the average TP load under baseflow conditions was 0.07 kg P d^{-1}, while that under high flow conditions was about 4.65 kg P d^{-1}, more than 66 times higher than that at baseflow. The corresponding average annual in-stream TP load at this site was estimated to be 0.58 t P y^{-1}.

If it is assumed that most of the load during baseflow conditions comes from groundwater and remains at a similar level throughout the year, then the increase in load between baseflow and high flow, i.e. 4.6 kg P d^{-1}, represents the P load to the river that comes from other sources under wet conditions. As this increase is strongly associated with an increase in B load, and there is no obvious dilution effect in the relationship between flow, and P and B concentrations, it is seems likely that this additional P comes from 'diffuse' sources rather than effluent from sewage treatment works (Jarvie *et al*., 2008). These are probably small sewage treatment facilities, such as septic tanks, although it should be noted that there is also a small sewage works in this area that discharges to groundwater which may also be contributing (Jarvie *et al*., 2008).

Calculations based on the number of households within this subcatchment, as identified from aerial photography, suggest that there are about 174 unsewered households here that contribute an estimated 0.23 t P y^{-1} to the river. This represents about 39% of the total annual P load, and is 8 times greater than the value that would have been estimated if the calculation had been based on the 22 consented discharges, alone.

By dividing the annual TP load associated with septic tank discharges (i.e. 0.23 t P y^{-1}) by the annual stream discharge at the sampling point (i.e. 15.1 x 10^6 m^3 y^{-1}), it is possible to estimate the average annual elevation in in-stream TP concentration that is attributable to these discharges. The data suggest that the associated increase in in-stream TP concentration is probably about 15 µg l^{-1}. As the corresponding measured average annual TP concentration is about 21 µg l^{-1}, this suggests that more than 70% of the TP in this small rural stream may potentially be attributable to septic tank discharges.

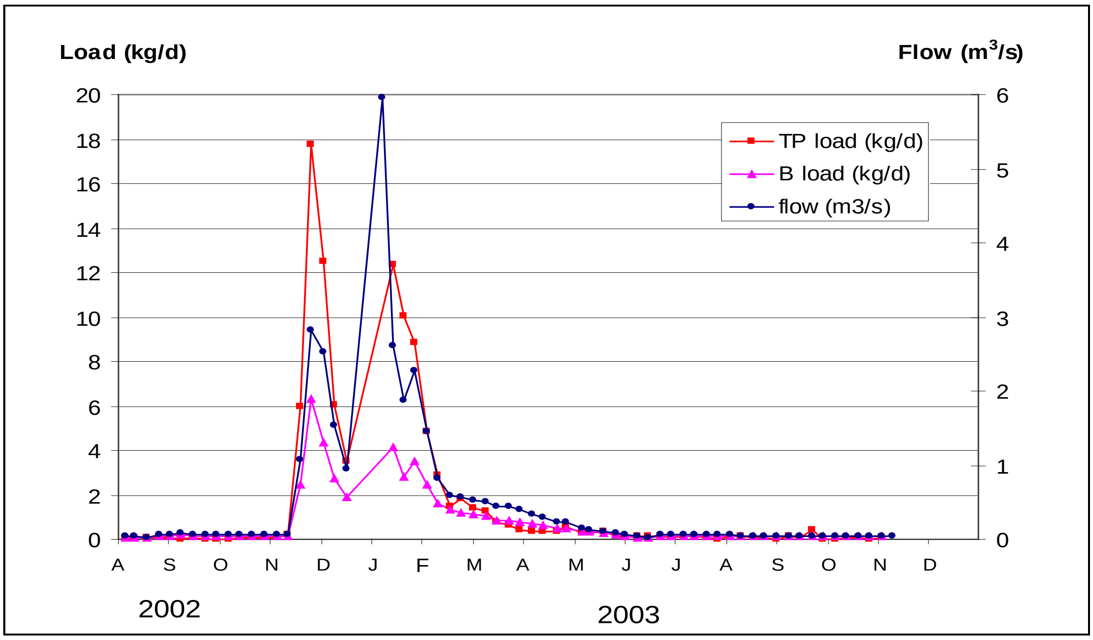

Figure 6.6 Temporal variation in streamflow and loads of boron (B) and total phosphorus (TP) in the Chitterne Brook at Codford St Mary, between August 2002 and November 2003.

In summary, the annual TP load in the Chitterne Brook at Codford St Mary is about 0.58 t P y^{-1}, with potentially about 0.23 t P y^{-1} (40%) entering the stream from septic tank discharges, while the remainder (60%) comes from agricultural sources and a small sewage works that discharges to soakaway within this subcatchment,. Figure 6.7 summarises the results outlined above and compares them to the results that these calculations would have given if unconsented septic tank discharges had been excluded from the study. It is clear that a much greater proportion of the TP load from the catchment would have appeared to be coming from agriculture if these inputs had been excluded.

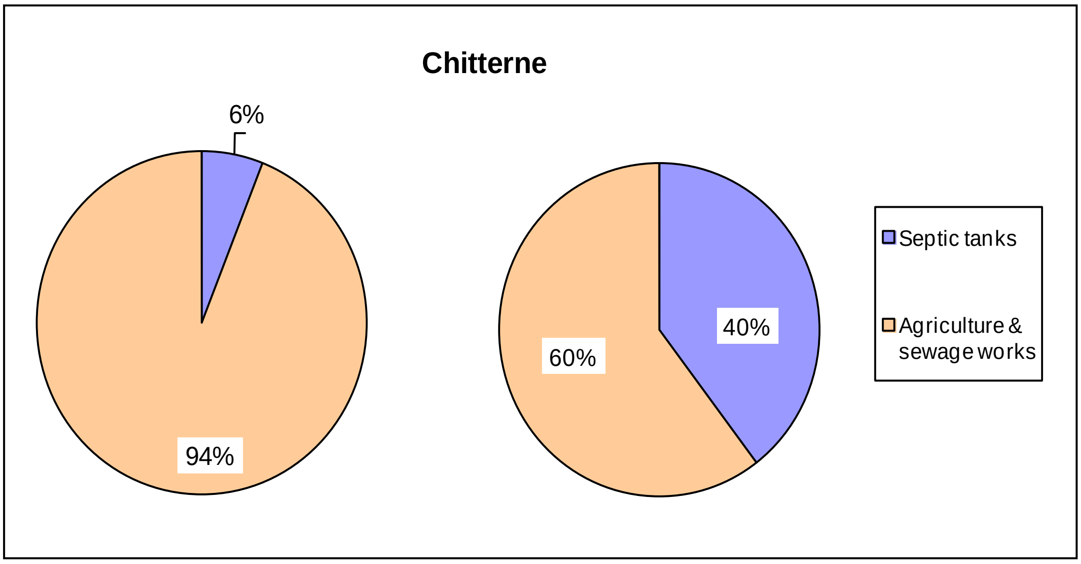

Figure 6.7 Estimated source apportionment of the TP load from the catchment to the Chitterne Brook at Codford St Mary, excluding (left) and including (right) estimated discharges from unconsented septic tanks.

6.2.3.2 River Wylye

The catchment of the River Wylye as a whole is 448 km^2 in area and consists of a predominantly rural catchment with three relatively large WWTWs that discharge directly into the river. These are at Warminster Garrison, Great Wishford and Warminster (Ash *et al*., 2006). The annual P discharge from these WWTWs given by Ash *et al*. (2006) is 0.99 t y^{-1}, 1.98 t y^{-1} and 7.55 t y^{-1}, respectively. However, the effluent monitoring data provided by the EA for this catchment show that the P output from Warminster fell by 74% following an upgrade in November 2001. Prior to this date, the average P concentration in the effluent had been 4.2 mg l^{-1} whereas it was only 1.1 mg l^{-1} between 8/3/02 and 1/10/09. So, for the purposes of this project, the annual P discharge from this WWTW was estimated to be about 26% of 7.55 t y^{-1}, i.e. 1.96 t y^{-1}. In addition to these larger works, there are small WWTWs at Monkton Deverill and Shrewton. The P outputs from these were estimated from the DWF discharge consent, as described above, and found to be 0.24 t y^{-1} and 1.3 t y^{-1}, respectively. The overall annual TP load to the river from WWTWs within this catchment was, therefore, estimated to be 6.5 t y^{-1}.

From the aerial photography, it was estimated that there were approximately 2049 ‘houses’ outside the sewered area. Of these, 31% were found to be within 100m of the River Wylye or one of its tributaries, while 68% were less than 500m away (Figure 6.8). In general, the unsewered households were further away from the main watercourses across the Wylye catchment as a whole than they were in the Chitterne subcatchment.

The amount of P entering the streams from all septic tanks within the Wylye catchment was determined as outlined above. It was found that this amounted to approximately 2.7 t y^{-1}, in comparison to the 0.24 t y^{-1} that would have been estimated on the basis of the 182 consented discharges, alone.

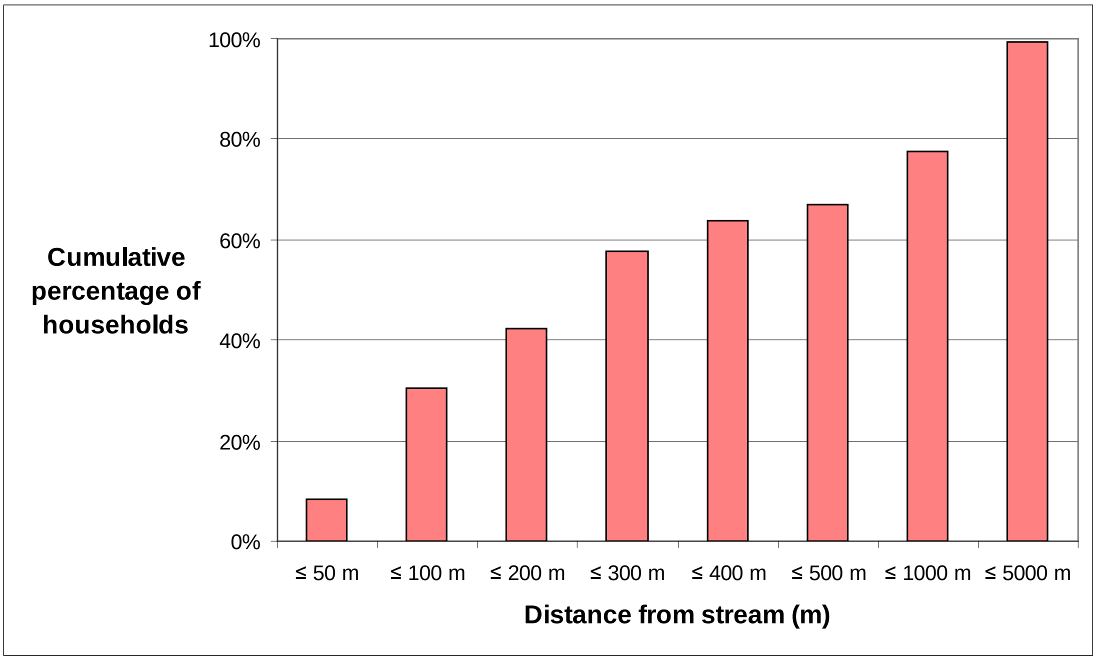

Figure 6.8 Cumulative percentage of unsewered households that are within the distance shown of the River Wylye or one of its tributaries.

The in-stream TP load at South Newton, the outlet of this catchment, was estimated from data supplied by the EA. This was based on values collected from 2007 onwards (i.e. 11/1/07 to 15/12/08), because TP values were not collected prior to that date. The annual TP load in the river at this site was estimated to be 11.6 t y^{-1}.

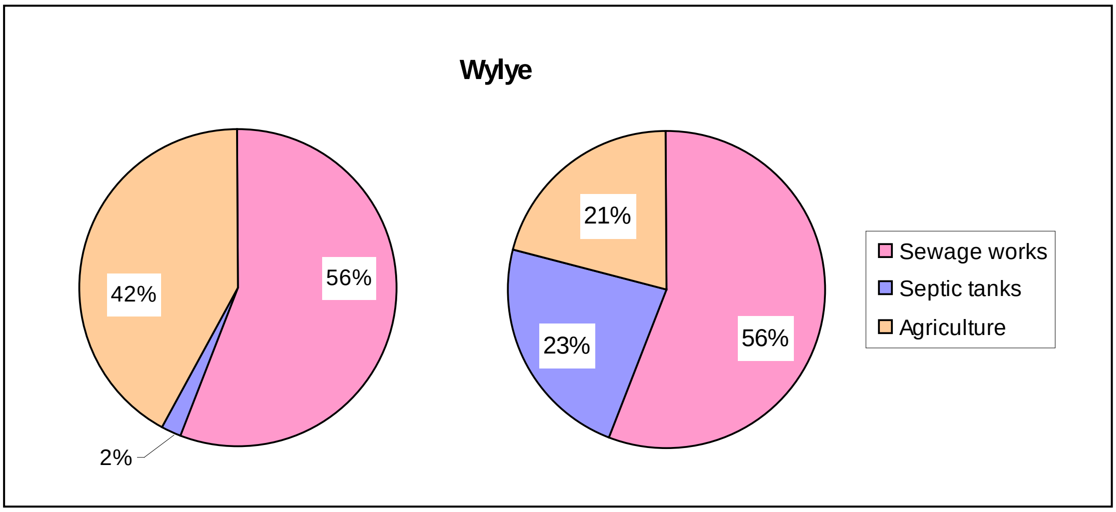

Figure 6.9 Estimated source apportionment of the TP load for the Wylye catchment above South Newton, excluding (left) and including (right) estimated discharges from unconsented septic tanks.

By dividing the annual TP load associated with septic tank discharges (i.e. 2.7 t P y^{-1}) by the annual stream discharge at the sampling point (i.e. 145 x 10^6 m^3 y^{-1}), it was possible to estimate the average annual elevation in in-stream TP concentration that could be attributed to septic tank discharges. The data suggest that TP discharges from this source probably increased in-stream TP concentrations by an average of about 19 µg l^{-1}. The corresponding measured average annual TP concentration is about 146 µg l^{-1}, which suggests that about 13% of the TP in terms of average concentration in this river may come from septic tank discharges on average over the year.

In summary, the study has shown that the annual TP load in the River Wylye at South Newton is about 11.7 t P y^{-1}. The results suggest that up to 2.7 t P y^{-1} (23%) may enter the stream from septic tank discharges, while a further 6.5 t P y^{-1} (56%) is probably contributed by treated effluent from WWTWs. The remainder, 2.5 t P y^{-1} (21%), probably comes from agricultural sources. Although this value is much smaller than the figure given by Ash *et al*. (2006) for P losses from agriculture in this catchment, i.e. 23.5 t P y^{-1}, the reason for this is unclear. Certainly the methods used were very different. Ash *et al.* (2006) estimated their figure from the number of animals and types of crops in the catchment by applying an export coefficient approach, while this study was based on in-stream measurements of P concentration and flow. However, the amount of P estimated to come from agricultural sources by Ash et al. (2006) is approximately twice as high as the annual P load suggested by the measured data for all P sources.

Figure 6.9 summarises the results outlined above and compares them to the results that these calculations would have given if unconsented septic tank discharges and small WWTWs had been excluded from the study. It is clear that a much greater proportion of the TP load from the catchment would have seemed to be coming from agricultural sources if these unconsented inputs had been excluded.

6.2.3.3 River Nadder

The Nadder catchment covers an area of 216 km^2. Land use in this area is dominated by cereal crops in the north and south of the area, with rough grazing and woodland along the river corridor (Johnes & Butterfield, 2005). Livestock production in this area is dominated by cattle rearing.

The results of the aerial photography interpretation suggested that there are approximately 1257 unsewered households within the Nadder catchment. Of these, only 63 (<5%) have consented discharges for their on-site sewage treatment facilities, while the remaining 1194 do not. About 203 (16%) of these unsewered houses are situated within 100 m of the river network, while 729 (58%) are within 500 m (Figure 6.10). So, the potential to discharge of P from these systems to the river in this very porous catchment is high. When these figures are converted to an annual P discharge potential to the river, they equate to a total of 1.6 t P y^{-1} from all unsewered households within the catchment, with only 0.1 t P y^{-1} (< 5%) of this attributable to consented discharges.

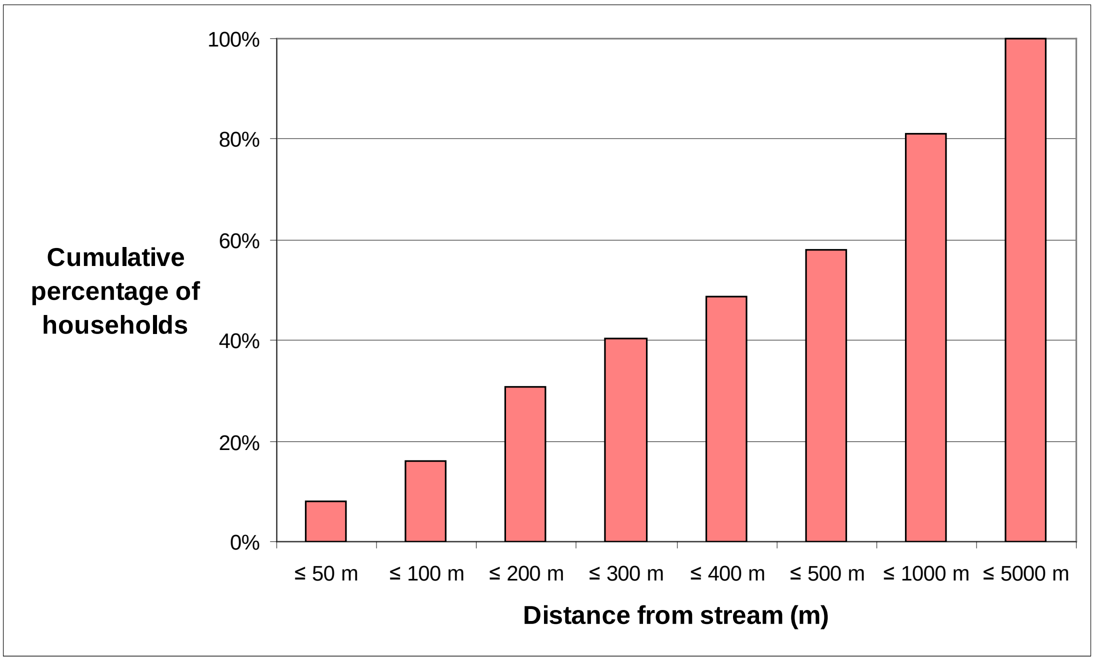

Figure 6.10 Cumulative percentage of unsewered households that are within the distance shown of the River Nadder or one of its tributaries.

The input to the river Nadder from large WWTWs was derived from summary flow and concentration data given by Ash *et al*. (2006) for the works at Bardford St Martins, Tisbury and Fovant. These values are shown in Table 6.5 and, together, account for a P input to the river of about 2.6 t y^{-1}. Although there are two other small WWTWs in the western part of the catchment (i.e. at Hindon and Semley), these have not been included in the calculations due to lack of data.

Table 6.5 Total phosphorus input to the River Nadder from sewage treatment works effluent.

Sewage works	Mean flow ($l\ s^{-1}$)	Mean TP concentration ($mg\ P\ l^{-1}$)	TP input to river ($t\ P\ y^{-1}$)
Bardford St Mary	1.4	10.15	0.5
Tisbury	9.1	5.00	1.4
Fovant	4.7	5.00	0.7
Total			**2.6**

It has been suggested that the P load in the river at Wilton that is attributable to agricultural sources ranges from 12 t P y^{-1} (Johnes & Butterfield, 2005) to 14.3 t P y^{-1} (Ash *et al*., 2006). However, the average annual TP load at this site from all sources, as estimated during this study from EA flow and concentration monitoring data spanning the period 24/1/06 – 15/12/08, was only about 12.4 t P y^{-1}. Of this, 8 t (64%) was found to be transported between November and March and 2.9 t (23%) between July and October.

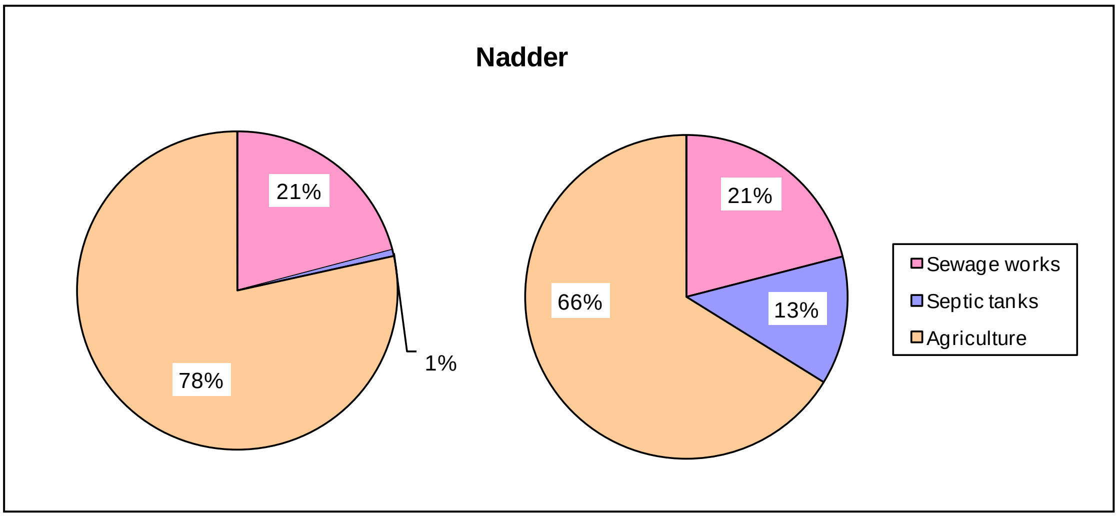

Figure 6.10 Estimated source apportionment of the TP load from the Nadder catchment upstream of Wilton, excluding (left) and including (right) estimated discharges from unconsented septic tanks.

By dividing the annual TP load associated with septic tank discharges (i.e. 1.6 t P y^{-1}) by the annual stream discharge at the sampling point (i.e. 91.8 x 10^6 m^3 y^{-1}), it was possible to estimate the average annual elevation in in-stream TP concentration that may be attributable to septic tank discharges. The data suggest that TP discharges from this source could potentially increase in-stream TP concentrations by an average of about 17 µg l^{-1}. The corresponding measured average annual TP concentration is about 86 µg l^{-1}. This suggests that about 20% of the TP in this river may be attributable to septic tank discharges, on an annual scale.

In summary, the study has shown that the annual TP load in the River Nadder at Wilton is about 12.4 t P y^{-1}. In terms of source apportionment, the results suggest that up to 1.6 t P y^{-1} (13%) may enter the stream from septic tank discharges, while a further 2.6 t P y^{-1} (21%) may be contributed by treated effluent from WWTWs. The remaining 8.2 t P y^{-1} (66%) probably comes from agricultural sources. However, this value is much smaller than the figures given by Johnes and Butterfield (2005) and Ash *et al*. (2006) for this catchment, i.e. 12 t P y^{-1} and 14.3 t P y^{-1}, respectively. The reason for this is unclear but probably lies in the different methods used for the calculations. Both Johnes and Butterfield (2005) and Ash *et al*. (2006) based their results on farm animal counts and cropping areas by applying an export coefficient modelling approach. Our study estimated total P loads from in-stream, measured data and subtracted the estimated inputs from known point sources to estimate the agricultural P load. The measured in-stream figure was only 12.4 t P y^{-1} for P from all catchment sources.

Figure 6.10 summarises the results outlined above, and compares them to the results that these calculations would have given if unconsented septic tank discharges and small WWTWs had been excluded from the study. It is clear that a much greater proportion of the TP load from the catchment would have appeared to be coming from agriculture if these inputs had been excluded.

6.3 Case study 2: The Broads

6.3.1 Site description

This part of the study focuses on two sub-catchments of The Broads, the upper River Bure and the upper River Ant. The upper Bure catchment is defined as that above Horstead Mill (628800, 318700), while that of the upper Ant is defined as that above Honing Lock (633100, 327000). These subcatchments were chosen because they are upstream of EA flow gauging stations, correspond to in-river P sampling locations and are above the tidal limit. Below the tidal limit, P budgeting in the Broads becomes very complex because the physical environment and in-stream P recycling/retention processes differ below this point (Halcrow, 2004), making it difficult to construct a nutrient budget based on upstream sources.

The upper Ant catchment has an area of about 44 km^2, while that of the upper Bure covers an area of about 328 km^2. Both catchments are very rural in character (Table 6.6), with only 6.2% and 15.3% of their area, respectively, being served by mains sewage treatment facilities (Figure 6.11). For the purposes of this study, it has been assumed that households outside of these areas are served by on-site sewage treatment facilities such as septic tanks. Septic tanks and package treatment plants were not distinguished in this analysis although, based on wider evidence, it is likely that the majority of these dwellings were served by septic tanks.

Table 6.6 Characteristics of the upper Bure and Ant subcatchments

Catchment	**Area** (km^2)	**Sewered area** (km^2)	**Landcover** (%)				**Source**
			Arable/ crops	**Improved grassland**	**Rough grazing**	**Other**	
Upper Bure	328	18	77	12	0	11	Johnes (1996)
Upper Ant	44.4	6.8	68	11	2	19	LCM2000 data (Fuller et al., 2002)

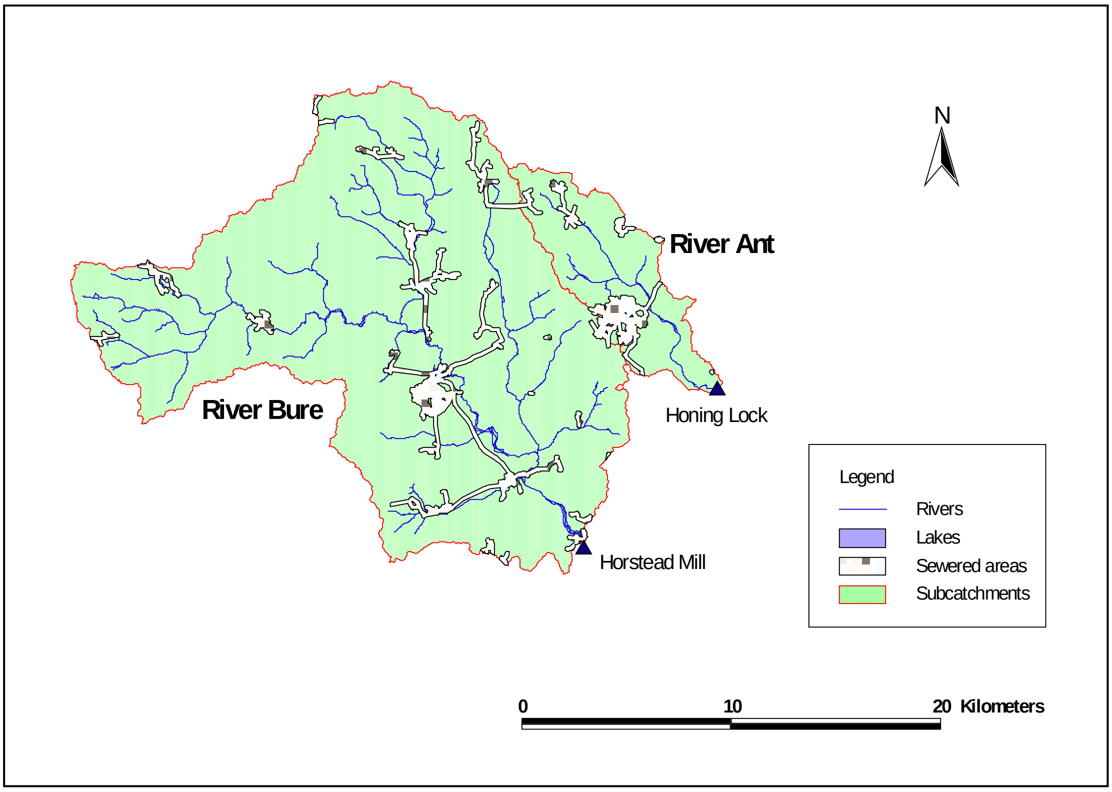

Figure 6.11 The upper Bure and Ant subcatchments of the Broads showing areas served by mains sewerage systems.

6.3.2 Methods

The number and location of all septic tanks within the Ant and Bure catchments was derived from the number of residential properties ('houses') outside the sewered area that were visible on a 1 m resolution aerial photograph of the area, using the method described in Section 6.2.2. The sewered area was defined as being within 100m of the sewer network as shown on maps provided by Anglian Water Plc. The number and locations of these septic tanks is shown in Figure 6.12. For comparison, the number and location of all septic tanks systems in this area that have discharge consents is shown in Figure 6.13.

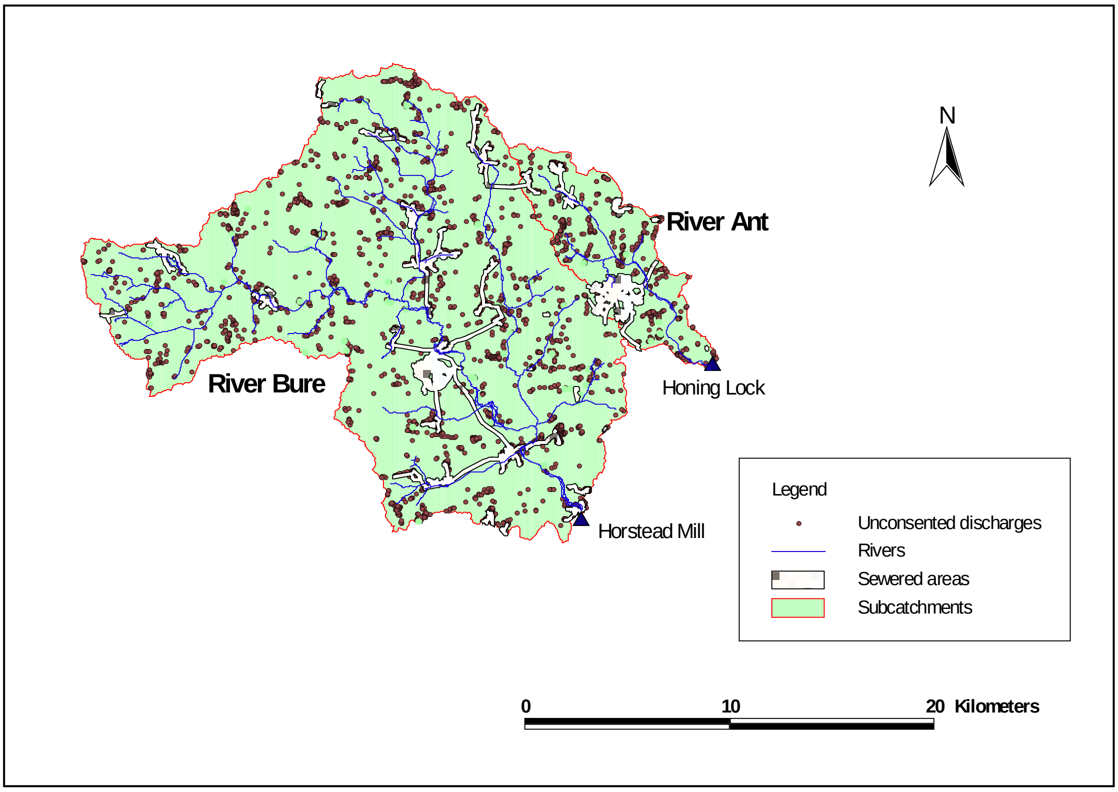

Figure 6.12 The upper Bure and Ant subcatchments of the Broads showing location of unconsented septic tank discharges.

The likely amount of P entering the drainage channels within these catchments from both consented and unconsented septic tank discharges was estimated making the assumption that all of these tanks were badly sited and not properly maintained. The assumption about location was considered to be reasonable within the context of this study, which aimed to examine the worst case scenario. The land draining into these two catchments is lowland and so the topography is flat. This may result in connection of the surface and ground water more often than in upland areas. In addition, in the Ant catchment, 40% of properties are known to be within 100m of a watercourse (Kelly, *pers. comm*.). However, it should also be noted that East Anglia has a semi-arid climate and that a significant proportion of the catchments in this study are actually in well drained areas. The assumption that many tanks are likely to be poorly maintained is supported by evidence from Ireland which indicates that 80-90% of septic tanks may not working effectively (Kirk *et al*., 2003; Arnscheidt *et al*., 2007).

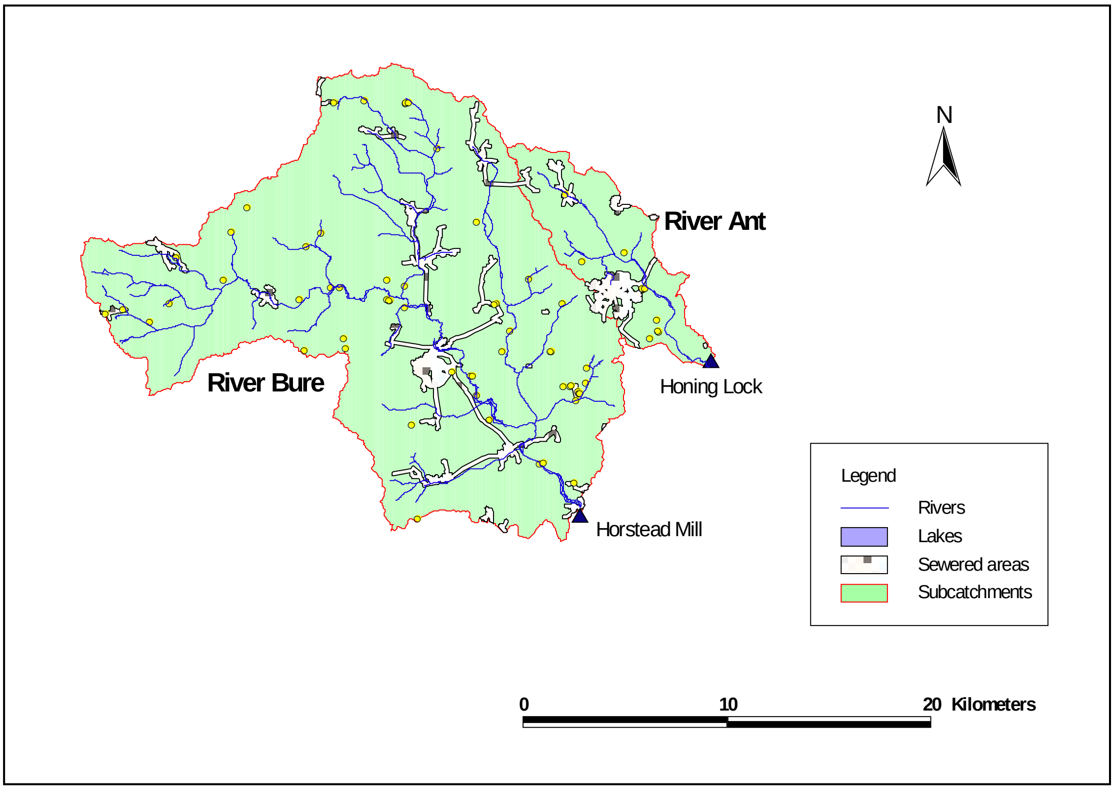

Figure 6.13 The upper Bure and Ant subcatchments of the Broads showing location of consented septic tank discharges.

The P loss from each tank was estimated from the average size of a household in Broadland using summary data from the 2001 census, i.e. 118,513 people ÷ 50,009 households = 2.4, the average *per capita* daily water usage for this area, i.e. 148 litres (www.defra.gov.uk), and the most commonly reported concentration of P in septic tank effluent (i.e. about 10 mg l^{-1}, see Section 2.2.2.). This equated to an annual P export per tank of about 0.54 kg *per capita* y^{-1} or 1.3 kg household^{-1} y^{-1}. It was assumed that none of these tanks received roof runoff, which can cause systems to overflow during heavy rainfall. Phosphorus losses from these systems at the subcatchment scale were estimated by multiplying the annual export *per* household by the corresponding number of households. These values were compared to the overall P load in the rivers at the subcatchment outflow and to the proportions of that load that could be attributed to upstream WWTWs or agricultural runoff.

The annual average TP load in the River Bure at Horstead Mill was calculated from instantaneous loading data for 1995/1996 given in Appendix A-18 of Johnes (1996). These data comprised 18 values determined over a 12 month period between 8/3/95 and 4/3/96. Total phosphorus concentrations in the River Bure over that period ranged from 5 µg l^{-1} on 23/8/95 to 181 µg l^{-1} on 8/3/95, with an average in-stream concentration of 60 µg l^{-1}. The corresponding value for the upper River Ant at Honing Lock was calculated by multiplying the annual average discharge at that point between 2007 and 2009, i.e. about 0.46 m s^{-1} (EA flow monitoring data), by the corresponding annual average TP concentration, i.e. about 100 µg l^{-1} (Johnes *et al*., 2003).

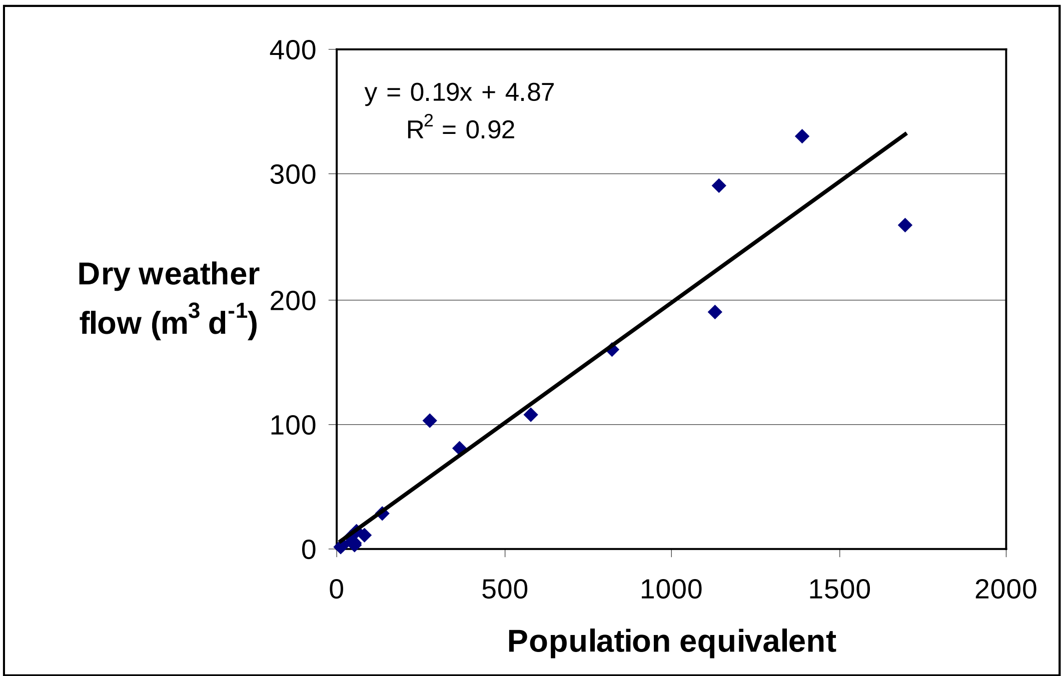

Figure 6.14 Relationship between dry weather flow consents and population equivalent values for small sewage works (PE = ≤ 2000) in Bure and Ant subcatchments of the Broads.

Annual discharge of P from sewage treatment works within each subcatchment was estimated on the basis of the population equivalent value (PE) of each works and a *per capita* P export coefficient 0.053 kg P y^{-1}. This coefficient was calculated from values based on effluent monitoring at the South Repps works given by Johnes et al. (2003), i.e. a P discharge of 0.12 kg P d^{-1} (43.8 kg P y^{-1}) and a PE value of 821. Where PE values were unavailable, i.e. Swanton Abbott works, these were estimated from the dry weather flow (dwf) values for these sites using the relationship shown in Figure 6.14. This relationship was derived from data provided by Anglian Water, which gave both a dwf and a PE value for other small sewage treatment works in the area.

6.3.3 Results

The number of individual houses that were digitised from the aerial photography of the upper Ant and Bure catchments is shown in Table 6.7. Overall, it was estimated that there were a total of 3478 unsewered households in this area, a number that is far greater than the 66 unsewered households that have discharge consents. The density of unsewered households in each of these catchments is about 7.7 km^{-2} in the upper Bure and 21.7 km^{-2} in the upper Ant.

Table 6.7 Total number of unsewered households within the upper Bure and Ant subcatchments and the number that have discharge consents for their on-site sewage treatment facilities.

Catchment	Unsewered households	
	Total	Consented
Bure	2515	56 (2.2%)
Ant	963	10 (1%)
Total	**3478**	**66 (1.9%)**

6.3.3.1 River Bure

The catchment of the River Bure above Horstead Mill is predominantly rural in character with 11 WWTWs that serve about 14,950 people. Most of these discharge directly into the watercourse, with the exception of Brisley and Felmingham, which discharge to soakaway. Using a *per capita* TP export coefficient of about 0.053 kg y^{-1} for works without P stripping (i.e. Aylsham, Briston and Roughton) and assuming a value of 20 % of this for those with P removal (White & Hammond, 2002), This equates to a total annual discharge of about 0.27 t P y^{-1} to the river from these sources (Table 6.8).

In addition to discharges from WWTWs, about 2515 septic tanks within the catchment are also a potential source of P to the river. Of these, about 10% were found to be located within 100m of a watercourse, while about 50% were less than 500m away (Figure 6.14). The potential amount of P entering the drainage system from these septic tanks was estimated to be about 3.3 t y^{-1}. This value is 37 times greater than the value that would have been estimated if consented septic systems, only, had been taken into account (i.e. 0.09 t y^{-1}).

Table 6.8 Waste water treatment works (WWTWs) within the upper Bure catchment, the number of people that they serve (PE) and their associated TP export *per* year[2,3]; data were provided by the Environment Agency (Data enquiry ref. CCE/2009/39585).

WWTW	**PE**	**TP export** (kg P y^{-1})
Aldborough	1,144	60.6
Aylsham	8,672	91.9
Brisley	15	0.8
Briston	2,472	26.2
Corpusty	579	30.7
Felmingham	84	4.5
Gresham	365	19.3
Hindolveston	281	14.9
Roughton	1,133	12.0
Skeyton	10	0.5
Swanton Abbott	192	10.2
Total	**14,947**	**271.6**

The annual TP load in the river at Horstead Mill was estimated to be about 6.6 t y^{-1} (Johnes, 1996). Of this, it was estimated that about half may be coming from septic tanks and a further 0.27 t y^{-1} form WWTWs. By difference, this suggests that the remainder, i.e. about 3.03 t y^{-1}, is probably attributable to agricultural runoff (Figure 6.15).

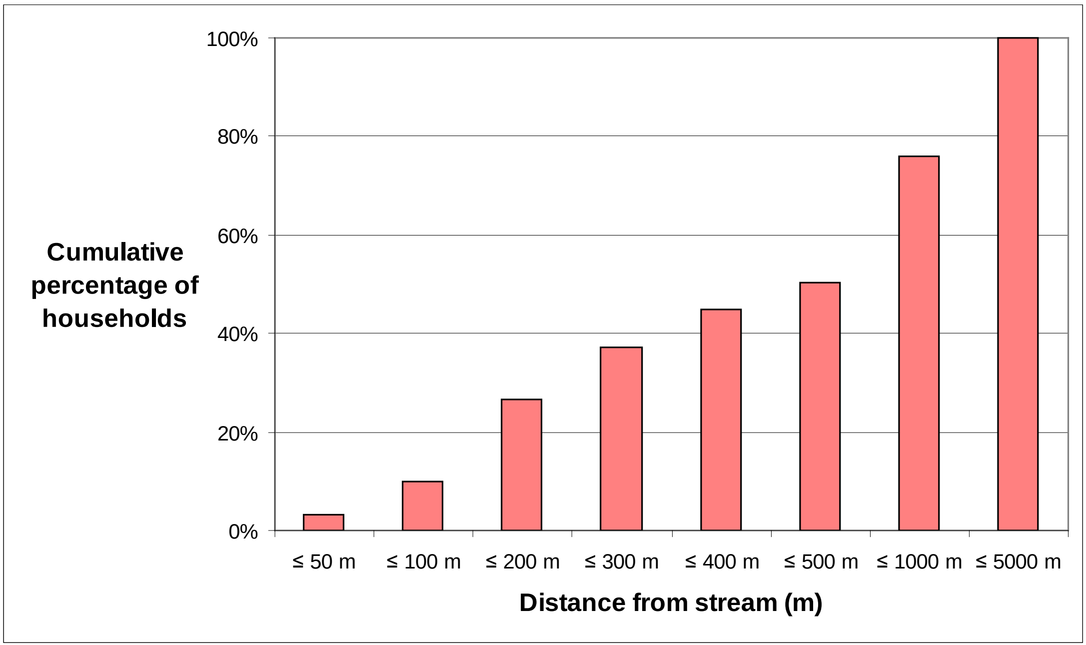

Figure 6.14 Cumulative percentage of unsewered households that are within the distance shown of the upper River Bure or one of its tributaries

[2] Investigations to assess the potential significance of Brisley and Felmingham WWTWs with respect to WFD water body failures were proposed as part of the AMP5 National Environment Programme.

[3] Since undertaking this analysis, the Environment Agency has informed us that a large potato processing plant may also be contributing a significant P load to the system. In any future risk assessments, this needs to be taken into account.

In summary, the study has shown that the annual TP load in the River Bure at Horstead Mill is about 6.6 t P y^{-1}. The results suggest that up to 3.3 t P y^{-1} (50%) may enter the stream from septic tank discharges, while a further 0.27 t P y^{-1} (4%) is probably contributed by treated effluent from WWTWs. The remainder, 3.03 t P y^{-1} (46%), probably comes from agricultural sources. Figure 6.15 summarises these results and compares them with the results that these calculations would have given if unconsented septic tank discharges and small WWTWs had been excluded from the study. It is clear that a much greater proportion of the TP load from the catchment would have seemed to be coming from agricultural sources if these inputs had been excluded.

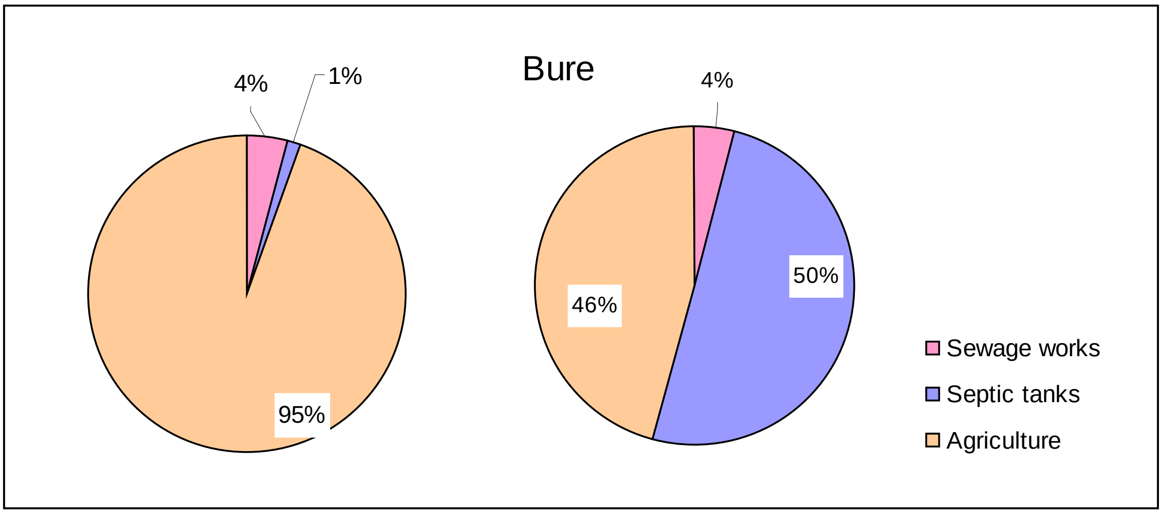

Figure 6.15 Estimated source apportionment of the estimated TP load from the Bure catchment upstream of Horstead Mill, excluding (left) and including (right) estimated discharges from unconsented septic tanks.

By dividing the annual TP load associated with septic tank discharges (i.e. 3.3 t P y^{-1}) by the annual average stream discharge at the sampling point (i.e. 7.4 x 10^6 m^3 y^{-1}), it is possible to estimate the average annual elevation in in-stream TP concentration that may be attributable to septic tank discharges in this river. The data suggest that TP discharges from this source could potentially increase in-stream TP concentrations by an average of about 44 µg l^{-1}. The corresponding measured average annual TP concentration is about 60 µg l^{-1} (Johnes, 1996). So, it is estimated that about 73% of the average TP concentration in this river may be attributable to septic tank discharges at an annual scale.

6.3.3.2 River Ant

The catchment of the River Ant above the tidal limit at Honing Lock is also predominantly rural in nature, with three WWTWs that serve a total of about 1,340 people. Of these, Honing and Southrepp discharge directly into the watercourse, while Trunch discharges to soakaway.Using a *per capita* TP export coefficient of about 0.053 kg y^{-1}, this equates to a total annual discharge of about 0.5 t P y^{-1} from WWTW sources (Table 6.9).

Table 6.9 Waste water treatment works (WWTWs) within the upper Ant catchment, the number of people that they serve (PE) and their associated TP export per year; data were provided by the Environment Agency (Data enquiry ref. CCE/2009/39585).

WWTW	PE	TP export (kg P y^{-1})
Honing	29	1.5
Trunch	134	7.1
Southrepps	821	43.8
Total	**984**	**52.4**

In addition to discharges from WWTWs, there are also an estimated 963 septic tanks within the catchment (Table 6.7). Of these, just over 40% lie within 100m of a watercourse and about 75% are less than 500m away (Figure 6.16). The amount of P entering the drainage system from these septic tanks was estimated to be about 1.25 t y^{-1}. This value is almost 100 times greater than would have been estimated if only those septic tanks with discharge consents had been taken into account (i.e. 0.013 t y^{-1}).

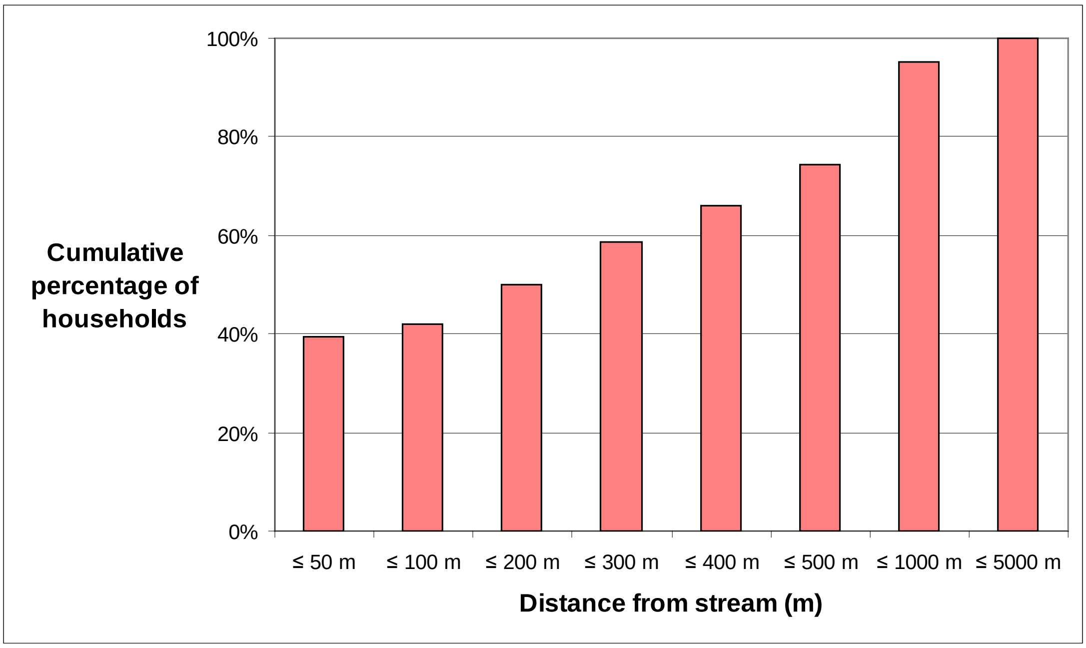

Figure 6.16 Cumulative percentage of unsewered households that are within the distance shown of the upper River Ant or one of its tributaries.

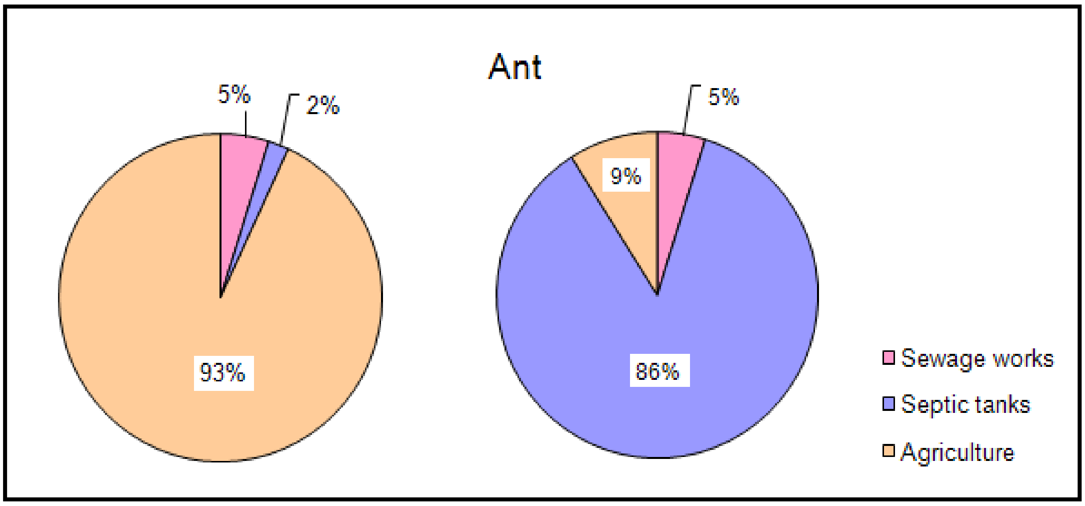

Figure 6.17 Estimated source apportionment of the estimated TP load from the Ant catchment upstream of Honing Lock, excluding (left) and including (right) estimated discharges from unconsented septic tanks.

In summary, the study has shown that the annual TP load in the River Ant at Honing Lock is about 1.45 t P y^{-1}. In terms of source apportionment, the results suggest that up to 1.25 t P y^{-1} (86%) may enter the stream from septic tank discharges, while a further 0.07 t P y^{-1} (5%) may be contributed by treated effluent from WWTWs. The remaining 0.13 t P y^{-1} (9%) probably comes from agricultural sources. Figure 6.17 summarises the results outlined above, and compares them to the results that these calculations would have given if unconsented septic tank discharges had been excluded from the study. It is clear that a much greater proportion of the TP load from the catchment would have appeared to be coming from agriculture if these inputs had been excluded.

By dividing the annual TP load associated with septic tank discharges (i.e. 1.25 t P y^{-1}) by the annual stream discharge at the sampling point (i.e. 1.45 x 10^9 m^3 y^{-1}), it is possible to estimate the average annual elevation in in-stream TP concentration that may be attributable to septic tank discharges. The data suggest that TP discharges from this source probably increase in-stream TP concentrations by an average of about 86 µg l^{-1}. The corresponding measured average annual TP concentration is about 100 µg l^{-1} (Johnes et al., 2003). This suggests that about 86% of the TP in this river at Horstead Mill may be attributable to septic tank discharges when considered at an annual scale.

7 Discussion and Recommendations

Eutrophication occurs when excessive amounts of nutrients enter freshwater systems as a result of human activity within the catchment. This results in a general deterioration in chemical and ecological water quality that poses a threat to the conservation status of many protected waters. Serious eutrophication problems are usually caused by an excess of phosphorus (P), because this nutrient usually limits productivity in freshwater systems.

A major source of P in many catchments is domestic waste. In urban areas, discharges of P from this source are controlled by the use of large waste water treatment works (WWTWs) with P stripping capabilities to process domestic waste. However, in rural areas sewage treatment facilities are often limited to small, on-site, systems that are much less efficient at retaining P. These include septic tanks systems, which can often be sited close to relatively clean and environmentally sensitive freshwater ecosystems.

In theory, properly maintained and correctly sited septic tank systems should pose little threat to the environment because much of the P in effluent discharged from the holding tank is removed as it percolates through the soil in the drainage field (Bouma, 1979; Wood *et al*., 2005; Gold & Sims, 2006). However, in practice, many septic tank systems do not function properly because they are not properly maintained, wrongly sited or incorrectly installed. They are often too close to the water table or surface water drainage system, situated in areas with inappropriate soil types, and some are rarely, if ever, emptied (Aitken *et al*., 2001; Kirk *et al*., 2003; Arnscheidt *et al*., 2007). Research suggests that 60 - 80% of septic systems are probably affected by at least one or more of these problems (Kirk *et al*., 2003; Arnscheidt *et al*., 2007). In some areas of the country, for example, as many as 82% of septic tanks have been found to discharge directly to water rather than *via* a drainage field (Aitken *et al*., 2001). Age of the system also affects the effectiveness of the drainage field in removing P from the effluent produced. This is because the soils in this area can become P saturated over time and lose their adsorption capacity.

Septic tank discharges can cause very noticeable local increases in in-stream P concentrations. In a tributary of the River Wyre, Lancashire, in-stream concentrations were shown to increase from 50 µg P l^{-1} to 400 µg P l^{-1} as the stream passed a cluster of houses served by septic tanks (Nicholson, 2007). In a similar study in Leicestershire, Stoate (*pers comm*.) noted that the P concentrations in one stream increased by 350% as it passed a small group of unsewered houses, while that in another stream rose by 240% under similar circumstances, reaching in-stream concentrations of up to 400 µg l^{-1}. However, it is possible to reduce these impacts. Barden (2007) found that the annual mean OP concentration in part of the River Chew fell from about 250 µg P l^{-1} to about 85 µg P l^{-1} when a first sewerage scheme was installed at Litton, replacing on site sewage treatment systems serving more than 500 people. All of these studies show marked impacts of discharges from these systems on chemical water quality.

In terms of seasonality, it is plausible that septic tank effluents increase the P concentrations of receiving waters all year round. This is because the P-laden discharges from most tanks, apart from those that discharge directly to a watercourse, will enter drainage channels *via* two different routes. In spring/summer, and under low flow/dilution conditions, P seeps through the surrounding soil or soakaway to the receiving water at a low, but relatively constant, rate. In autumn/winter, and under high flow/dilution conditions, higher levels of P can be flushed from the tanks themselves and the surrounding soils during heavy rainfall. So, although P loads to receiving waters will be low in spring/summer and high in autumn/winter, in terms of impact on receiving waters, the combination of these two mechanisms has the potential to elevate in stream concentrations throughout the year. In the case of P discharges from WWTWs, roughly the same load and concentration of P is continuously discharged generally leading to higher P concentrations in receiving waters under low flow/dilution conditions in spring/summer, when ecological sensitivity is greatest rather than during the autumn/winter (Jarvie *et al*, 2006).

Although dilution and in-stream processing further downstream mean that P concentrations decrease with increasing distance from septic tank sources, the P concentrations outlined above are very high locally and are likely to cause considerable ecological damage. It has been shown that macroinvertebrate and diatom biodiversity declines when P levels increase up to 100 µg l^{-1} with an associated increase in diatom biomass (Mainstone, 2010; Stoate, 2008). The values recorded downstream of septic systems are often 3 – 4 times greater than these in-stream concentrations and may, therefore, threaten internationally and nationally important nature conservation sites when sited in their catchments.

When integrated at the catchment scale, the impact of septic tank discharges on P concentrations is less marked. However, at some sites, it could still affect the likelihood of meeting water quality targets (Arnscheidt *et al*., 2007). Estimates from the present study suggest that discharges from these systems might potentially increase in-stream P concentrations by up to 15 – 19 µg P l^{-1} in rural areas of the Hampshire Avon in southern England, i.e. the Chitterne Brook and Rivers Wylye and Nadder. As average in-stream P concentrations in these waterbodies are about 20 µg P l^{-1}, 90 µg P l^{-1} and 150 µg P l^{-1}, respectively, such an elevation in P concentration due to septic tank discharges could significantly contribute to a failure to meet water quality targets at such sites. However, it is important to stress that these calculations are based on extreme worst case assumptions, and so further research is required to refine them.

Although the EA routine monitoring data are not ideal for exploring the impact of septic tank discharges on downstream river water quality because they are focused, primarily, on sites downstream of large point source discharges, it has been possible to estimate in a broad way the likely contribution of P from these systems to drainage waters from the catchment. The amount of P estimated to be entering the rivers Wylye and Nadder from septic tank discharges was compared to that estimated to be coming from agriculture and WWTWs in these areas. Within the Nadder river system, the P discharged by septic tanks was estimated to be equivalent to about 20% of that from agricultural sources and 57% of that from WWTWs. Within the Wylye river system, the corresponding figures were 100% and 42%. This suggests that P discharges from septic tanks may not be ‘negligible’ at the catchment scale, as has sometimes been suggested.

It should be noted, however, that the values outlined above refer to TP concentrations. Of this, it is generally accepted that soluble phosphorus (OP or SRP) is the component of TP that stimulates the growth of nuisance algal blooms and aquatic plants and causes impacts on other biota. Ekholm and Krogerus (2003) showed that (weight for weight) P levels in septic tank discharges are three to five times more likely to promote algal growth in receiving waters than runoff from land. If these factors are incorporated into the source apportionment values as a measure of their potential to cause ecological degradation of water quality, then the WWTW and septic tank discharges that have been identified in this study become an even more important source of in-stream P.

One of the main problems in estimating P losses from septic tanks at the catchment scale is estimating the number and location of the tanks, themselves. This is because, although new discharges now require planning consent within Great Britain, many older systems were installed before this was a statutory requirement. A range of methods for estimating the number and, in some cases, location of unconsented septic tanks are described. These include methods based on postcode data, large area statistics, local knowledge, aerial photography and population census returns. Most of these methods are too labour intensive to apply at the national scale and a system for recording the locations of septic tanks is needed if their discharges and, consequently, their impact on the environment are to be managed and reduced.

The apparent size of the unknown discharges problem was assessed within the present study using the Wylye and Nadder catchments as examples and using a newly developed method based on interpretation of aerial photography. The results suggested that less than 10% of septic tank discharges are consented within these catchments. It is unclear how many septic tank systems there are across the UK but in southern Ireland, where good records are kept, data from the 2006 census suggest that about 420,000 dwellings are served by septic tanks, which serve about 30% of the population (Central Statistics Office Ireland, 2006). In England and Wales, a corresponding figure of about 1.2 million households has been suggested (Anthony, *pers. comm*.), which equates to about 5% of the population.

The likelihood of any particular septic tank causing pollution problems depends partly on its location and partly on its condition and the way that it is managed. In terms of location, it is suggested that national datasets could be used to provides a useful estimate of the risk of septic tanks causing pollution problems in any particular area based on soil hydrological characteristics, topography and proximity to a watercourse. Being based on national datasets, this approach can be implemented at the catchment, regional or national scale, to provide an initial screening to identify areas at high risk of pollution that can be targeted for improvement first.

Within these areas, the potential for individual tanks to cause water pollution problems depends on their size, design and the way that they are managed (Morgensten, 2005). Assessment of these criteria requires site specific information on the age, size, condition and management of individual tanks that is not generally available at the wider scale. So, assessment of individual tanks must be implemented at a site specific scale. A framework for exploring these site-specific factors in local assessments is presented in Appendix 1, but further research on risk factors in the UK situation is required to develop this into a more reliable risk assessment tool.

A variety of options are available for reducing the potential of septic tank systems to pollute the environment. By far the cheapest, and probably the most effective, is encouraging owners to de-sludge their systems more frequently to prevent the overflow of untreated sewage; it has been estimated that up to 80% of septic tanks are rarely, if ever, emptied. This would otherwise block the soakaway, reducing its capacity to remove P from the effluent. Some tanks may also need to inspected and repaired, upgraded or replaced, where necessary, to ensure that they are fit for purpose. Within the Lly Tegid catchment in Wales, for example, an inspection of private sewage treatment systems found that 30% - 50% were in poor condition and either polluting or showing potential to pollute (Selyf Consultancy, 2002). Many tanks may be badly sited. Gill *et al*. (2007), for example, found that only 4 out of 74 sites examined were actually suitable for septic tank installation. Most were rejected on the basis of too high a water table, making it unlikely that many of the problems could be solved by simply moving the tank to a more suitable location, locally. In areas where there are clusters of septic tank systems in unsuitable areas, it may be possible to replace them with a first time sewerage scheme. These can be very effective at reducing P concentrations in receiving waters (Barden, 2007), but care should be taken to ensure that the level of P removal within the new sewage treatment system is sufficient to ensure that the P load from the new source is less than that from the septic tanks that it is replacing.

A key issue in trying to quantify septic tank discharges of P and assess their impacts on chemical and ecological water quality is that very few data are available for analysis. For this reason, the results in the present study are based on a modelling approach that assumes that all of the P discharged by properly functioning septic tank systems eventually ends up in a nearby watercourse. This assumption has undoubtedly led to overestimates of the significance of septic tanks in the catchments studied, but the results do indicate that further work on the pollution risk posed by septic tanks is justified. Although, in a properly sited and maintained septic system, some of this P would be retained by soil adsorption and biological uptake in the drainage field, in reality, many septic tank systems are unlikely to working properly and, as a result, are probably discharging far more P than they should. On balance, given the available information, it was assumed that discharges to water from an 'average' septic tank system were probably about 0.54 kg P $capita^{-1}$ y^{-1}, a value that is similar to many reported in the literature. It is, however, recommended that further work is required to investigate the concentrations of P in septic tank effluents, and the factors affecting this, together with work to better characterise soil P concentrations across discharge plumes. This is essential if more realistic estimates of the risk posed by septic tanks around freshwater SSSIs are to be produced. A follow on project, also funded by Natural England, is now looking at the capacity of soil soakaways to remove P from discharged effluent before it reaches a waterbody.

8 Recommendations for identifying SSSIs at greatest risk of contamination from septic tank discharges

Although there is considerable anecdotal evidence that discharges of P from septic tank systems can cause local and catchment level elevations in P concentrations that are sufficiently high to cause ecological damage to receiving waters, there are few data available with which the extent of this problem can be assessed accurately. In an ideal world, all SSSIs would be assessed at a site specific level for potential contamination from septic tank discharges. However, realistically, resource constraints mean that this can only be achieved at a small number of locations. So, as a first step, it is important to be to identify, and focus on, sites that have been identified as being at greatest risk of nutrient enrichment from these discharges using a high level risk assessment procedure based on existing, national level, datasets. Such a procedure has been proposed in Appendix 1, but it should be noted that some of the boundary values proposed for assessing the level of increased risk associated with different distances from a waterbody, slopes, soil types, septic tank densities, *etc*. are derived from values determined in other countries. As such, they have not been validated for application within the UK where key variables, such as soil hydrological characteristics, are different. Also, the relative importance of these factors, an important issue when combining them to create an overall, single, risk assessment value, has not been quantified.

Recommended actions for risk assessment at the catchment/regional scale:

1. Further develop and validate the catchment/regional scale risk assessment procedure outlined in Appendix 1 for application within the UK.
2. Determine the number and location of all septic tanks.
3. Apply the resultant risk screening model across potentially vulnerable sites taking into account slope, soil type, density of septic tanks and distance from a waterbody.
4. Compare outputs from the risk screening model with site condition assessments at SSSIs and any other relevant information, e.g. P source apportionment, outputs from SIMCAT modelling, *etc*.
5. Produce a prioritised list of SSSIs for further assessment.

One of the biggest challenges in implementing the recommendations outlined above is locating all of the relevant septic tank systems because, in most parts of GB, less than 10% currently have discharge consents. Several suggestions on ways of achieving this at the site specific/small catchment scale are outlined in Section 4.3. Understanding the locations of small domestic discharges is critical to help ensure that risk screening, as described above, is reliable.

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10 List of abbreviations

AMP	Asset Management Plan
CAR	Controlled Activities Regulations
CEH	Centre for Ecology and Hydrology
CoPA	Control of Pollution Act
DoE	Department of the Environment
EA	Environment Agency
EU	European Union
GIS	Geographical Information System
HOST	Hydrology of Soil Types
P	Phosphorus
PE	Population equivalent
RoC	Review of Consents
SAC	Special Area of Conservation
SEPA	Scottish Environment Protection Agency
SPA	Special Protection Area
SPR	Standard percentage runoff
SRP	Soluble reactive phosphorus
TP	Total phosphorus
UK	United Kingdom
US	United States
UWWTD	Urban Waste Water Treatment Directive
WWTW	Waste Water Treatment Works

Appendix 1: Developing a risk assessment framework

A1.1 Background

In recent years, it has been widely recognised that poorly performing septic tanks can degrade the quality of nearby waterbodies (Harris, 1995; Scandura & Sobsey, 1997; Geary & Whitehead, 2001; Lipp *et al*., 2001; Carroll *et al*., 2006). In the US, it is believed that 60% of such systems do not function properly and they have been identified as the second most important cause of contamination of water sources across the country (Canter & Knox, 1985; US EPA 1997). Similar problems have been noted in other parts of the world. This includes the UK and Ireland, where surveys have shown that about 80% of these systems are not working effectively (Selyf Consultancy, 2002; Kirk *et al*., 2003).

There are many factors that affect the functioning of septic tank systems. Because they discharge to soakaway, these systems rely on soils in the drainage field to clean up the effluent that they discharge. As such, these soils form the 'last line of defence' between the point of discharge and the receiving waters (Dawes & Goonetilleke, 2003), so it is very important that these systems are located on appropriate soil types. However, it should also be noted that many other attributes such as size, location, local hydrology and level of maintenance, also have an impact on the effectiveness of these systems at removing pollutants (see Section 2.2.1).

Many attempts have been made to quantify the level of risk that discharges from these systems pose to the environment and human health, mostly in response to high profile outbreaks of disease associated with contaminated water resources. For example, following an outbreak of Hepatitis-A caused by eating shellfish from a lake that was contaminated with septic tank effluent, the State of New South Wales, Australia, implemented a programme of measures known as the "Septic✓Safe" programme. This included the development of a risk based model known as the 'Onsite Sewage Risk Assessment System' (OSRAS) (Brown & Root Services, 2001; Kenway *et al*., 2001), to estimate the hazards associated with failure of onsite sewage treatment systems. This followed the general approach outlined by Ganoulis (1994), which can be summarised as follows:

$$\textit{Risk} = \textit{probability of failure} = P\,(L > R)$$

where:

L = pollutant load to the system
R = the system's resistance to that load

In relation to water quality issues, R is a function of the water quality standards or threshold values that are in danger of being exceeded and L is the load of pollutant from the septic tank systems to the waterbody. Although straightforward in its concept, this approach can be very difficult to apply in practice, because the variables required are not easy to quantify from readily available information.

An alternative approach is that described by CMHC (2006), which is broadly based on the 'DRASTIC' model proposed by Aller *et al*. (1987) and tested by Kinsley *et al*. (2004) and Kinsley & Joy (2008). In outline, this identified a range of factors that affect the likelihood of septic tank effluent contaminating waterbodies, including system age, soil type, lot size (or septic tank density), depth to the water table, aquifer conductivity and proximity to surface water. Each risk factor was subdivided into five levels of risk rating (i.e. 0 = no risk to 5 = very high risk) and then given a weighting that reflected its relative importance, as a percentage, within the overall risk assessment (Table A1).

Table A1 On site wastewater risk assessment model factor weighting *(after CMHC, 2006)*.

Risk factor	Description	Weighting (% of total)
R_1	System age	30
R_2	Soil type	15
R_3	Lot size	15
R_4	Depth to high ground water table	15
R_5	Aquifer conductivity	5
R_6	Proximity to surface water	20

This approach enabled the overall risk model value *(RISK)* for each septic tank system, or group of systems, to be calculated by summing the products of risk rating (*RISK_RATING*) and associated weighting (*WEIGHTING*) for each risk factor, as follows:

$$RISK = \sum_{1}^{n} (RISK\_RATING \times WEIGHTING)$$

In relation to the current project, it has been shown that discharges from septic tanks in rural areas can cause elevated P concentrations in receiving waters and could potentially contribute to failure to meet water quality targets. Reducing the risks of contamination from these sources within the framework of limited resources, requires those systems that are most at risk of causing pollution problems to be identified and targeted first for remedial action. This Appendix begins the process of developing an hierarchical risk assessment procedure that will help identify areas where septic tanks are most at risk of causing water pollution problems at both catchment/regional/national and at site specific scales, using the ‘DRASTIC’ approach outlined above. The risk rating and relative weightings assigned to each variable are based on best available data, mainly from the US, and need to be tested, updated and validated prior to widespread application within the UK.

A1.2 Factors affecting pollution risk

The risk of water pollution by septic systems depends on a wide range of factors that affect their success or failure. In a study by Morgenstern (2005), these were summarised as:

- improper location
- poor design
- incorrect management

The relative contributions of these factors to system failures and consequent pollution problems are discussed below in relation to P discharges.

A1.2.1 Improper location

Slope, soil type and hydrological characteristics

Septic systems need to be located on suitable soil types, ideally well-drained sandy loam with acceptable year-round percolation rates and a minimum of 90 cm depth of soil above the highest level of the water table (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).

Slope also affects the way that the drainage field functions. It has been suggested that septic systems should only be sited on ground with a slope of 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 noted that tanks sited on steeper slopes probably needed larger drainage fields to work effectively than those on shallower slopes, estimating that the minimum size of drainage field required for a system to operate effectively on a slope of 5-10% is about 5 ha, while the corresponding values for slopes of 10-20% and >20% are 6 ha and 8 ha, respectively. They also recommended that systems should be placed on a concave rather than convex slope, wherever possible, and that they should not be sited at the base of a slope. They also noted that septic systems should not be sited in areas that are subject to seasonal flooding, because inundation could wash the contents of tanks into nearby watercourses (Canter & Knox, 1985).

Proximity to surface waterbody

It is also important that septic systems and their drainage fields are situated at a suitable distance from ditches, streams, lakes and other drainage channels to reduce the impact on water quality. Canter and Knox (1985) suggested a minimum ‘setback’ distance of about 30 m from a watercourse, but more recent research has suggested that that this distance needs to be much greater. McGarrigle & Champ (1999), for example, have suggested that septic tanks should not be installed within 400 m of a waterbody and other studies have suggested that even this value is too low. This is because, as septic tank systems age, the plume of discharge extends and they are more likely to contaminate waterbodies at greater distances. Robertson (2003) tracked the effluent plume from a septic tank that corresponded to a soil water concentration of 0.9 mg P l^{-1} and found that it moved towards a nearby waterbody at a rate of about 1 m per year over a 16 year period. This evidence also suggests that older installations are more likely to cause contamination of waterbodies at greater distances than newer installations.

Density of septic systems

The impact of septic tank systems on downstream water quality is also affected by the dilution capacity of the receiving water (Canter & Knox, 1985). Although this is rarely taken into account when new systems are consented within the UK, elsewhere it is recommended that the maximum density of these systems should be no more than one for every 4 ha of land and even less if the slope of the land is greater than 5% (Canter & Knox, 1985).

A1.2.2 Poor design

Age

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), although many have been in constant use for much longer than this. In contrast, most newer systems have been constructed from stronger and more watertight material that should last for up to 50 yrs (Canter & Knox, 1985).

The age of the system can also affect 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 it less able to remove P from effluent. Older installations are more likely to be affected by this problem than more recently installed systems.

Size

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, it is at high risk of polluting the environment due to hydraulic overloading. This situation is common and often results from of an increase in the size of, or change in the use of, the property served by a septic system without corresponding improvements to the system itself. It can also result from increased levels of water usage within a property as a result of changes in lifestyle. Many older systems that still receive roof runoff are also prone to hydraulic overloading during periods of heavy rainfall.

A1.2.3 Incorrect management

The operation and maintenance of septic tanks within the UK is largely unregulated and left to the individual householder. Many householders are unaware of the need to manage their systems correctly in order to reduce their impact on the environment. As a result, few use appropriate (i.e. ‘septic tank safe’) household cleaning products and many are unaware of the need to de-sludge their tanks regularly. There is also a widely held misconception that septic tanks do not need to be emptied if they are working properly; so, many such systems are full to overflowing. To make matters worse, many tanks are in a poor state of repair with holes in their main structure and/or leaking seals that allow untreated effluent to escape. It is estimated that more than 80% of septic tank systems within the UK are causing pollution problems due to improper management and lack of maintenance (Selyf Consultancy, 2002; Kirk *et al*., 2003).

A1.3 Assessing the risk of pollution

The original risk assessment model of CMHC (2006), on which the approach used in this study is based, was developed within the US. It used spatial datasets that, although readily available within the US, are not available within the UK. So, while the original approach can be followed in general, in detail it requires a certain amount of modification to meet the objectives of this project. In addition, for the purposes of the current project, the approach has been subdivided into two levels of assessment enabling catchment/regional/national scale identification of high risk areas using national level datasets (Section A1.3.1) and local level assessment of risks based on site specific information (Section A1.3.2).

A1.3.1 Assessment at the catchment scale

Assessment of the risk of pollution from septic tanks at the catchment scale needs to be based on spatial datasets that are available at the national, regional or catchment scale, because site specific data are not available over wide geographical areas. In terms of the risk factors that affect septic tank discharges within the UK, appropriate national datasets comprise:

- UK Hydrology of soil types (HOST) classification data at 1km resolution (Boorman et. al., 1995).
- Digital terrain model at 50m resolution (Morris & Flavin, 1990; 1994).
- Watercourses data at 1:50,000 scale (Moore et al., 1994).
- Lake shoreline data (Ordnance survey data © Crown copyright).

The risk assessment procedure outlined below is based on these datasets.

Soil type and hydrological characteristics

CMHC (2006) based their classifications of soil type on estimated soil hydraulic conductivity, with higher risk being attributed to soils that have relatively low permeability. Within GB, the most similar spatial dataset of soil hydrological characteristics is the Hydrology of Soil Types (HOST) classification, which was developed by Boorman et al. (1995). This dataset combines soil type, depth to high water table and aquifer conductivity into a single value that can be taken to reflect the ability of soils in a particular area to remove P from septic tank effluent and reduce the risk of water pollution problems.

In outline, the HOST classification divides UK soil type into 29 classes based on 11 conceptual hydrological response models that describe the dominant pathways of water movement through the soil and substrate. The soils are classified according to their physical properties and the hydrogeology of the underlying substrate. For the purposes of this project, the 29 original HOST classes have been grouped into high (5), medium (3) and low (1) risk of waterbody contamination on the basis of the depth of the impermeable or gleyed layer (where ≤ 100cm from the surface represents high risk and > 100 cm from the surface represents low risk) and whether by-pass flows are common (high risk) or uncommon (low risk). HOST classes with high risk ratings in both of these categories were rated as high risk (5); those with one low and one high risk rating were given an overall rating of medium risk (3); those with two low risk ratings were given an overall rating of low risk (1). The resultant allocation of HOST classes to overall risk rating groups is shown in Table A2.

Table A2 Risk ratings for impact of soil type and hydrological characteristics (HOST class) on likelihood of contamination of nearby waterbodies by septic tank effluent.

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,

The spatial distribution of these risks across the Ant and Bure catchments is shown in Figure A1. The risk of septic tanks contaminating watercourses within these catchments is low to medium in most areas because they mainly comprise HOST class 5, which has an unconsolidated, microporous substrate with little by-pass flow and water table at ≥ 2m depth. However, the risk is much higher in some areas because these mainly comprise HOST class 9, which has an unconsolidated, microporous substrate with a much higher by-pass flow and water table at ≤ 2m depth.

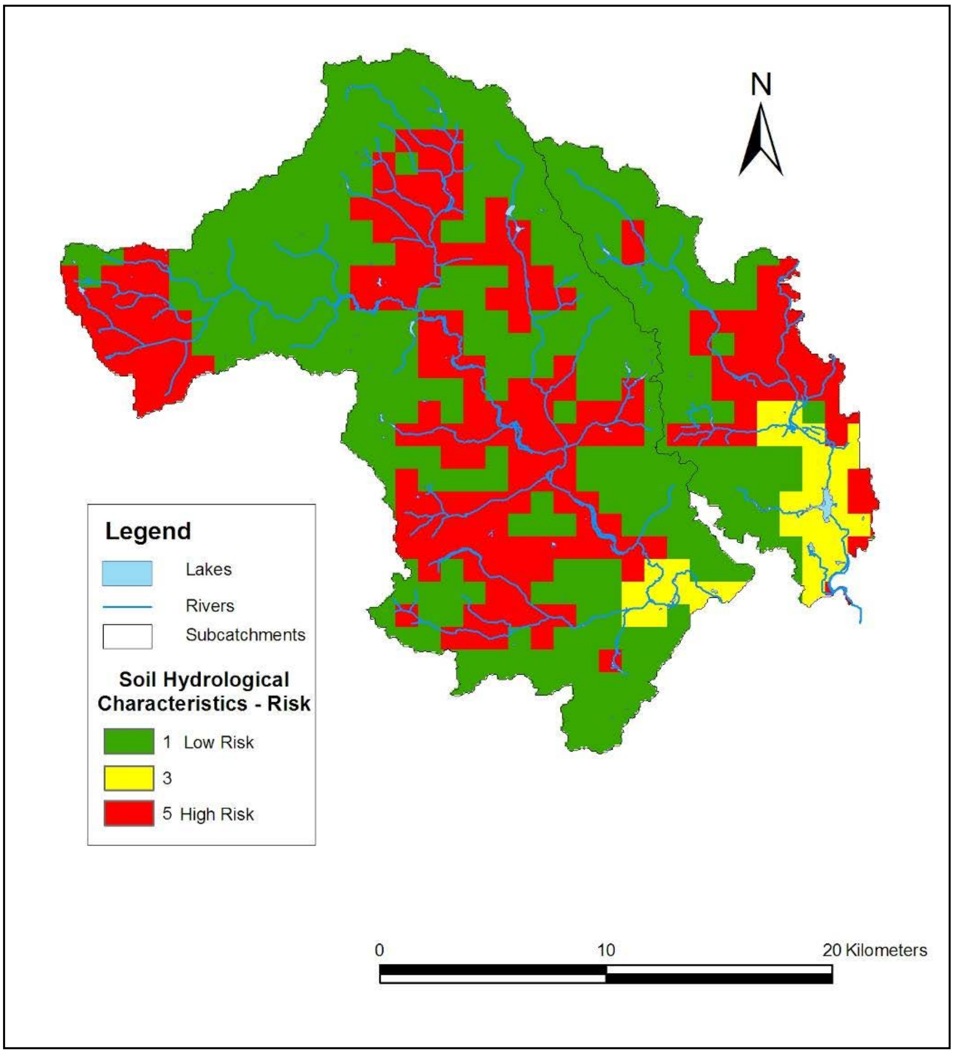

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Figure A1 Map showing level of risk of septic tanks contaminating watercourses based on soil type and hydrological characteristics within the catchments of the rivers Ant and Bure.

Proximity to surface water

Septic systems that are closer to waterbodies are more likely to cause pollution problems than those that are situated further away. CMHC (2006) proposed two levels of risk rating for proximity to a waterbody, i.e. either inside or outside of the

100 year flood plain boundary. This recognised the fact that any septic system that is either within or partially within this boundary is at high risk of causing pollution due to inundation when water levels are high. The authors suggest that septic systems outside this boundary will pose little to no risk of causing pollution problems.

The current study proposes a slightly different approach to estimating risk in relation to proximity to a waterbody. It assumes that systems that are within 25 m of a watercourse are at much higher risk of causing pollution problems than those that are more than 500 m away, with progressively increasing levels of risk between these values. Suggested risk ratings corresponding to different distances from a waterbody are shown in Table A3. The spatial distribution of these risks across the Ant and Bure catchments are shown in Figure A2. However, it should be noted that artificially enhanced hydrological connectivity in this area, such as that created by man-made drainage networks, has not been included in this assessment. The complex system of artificial drains in this area, if included, would probably increase the likelihood of contamination of a water course by septic tank effluent if incorporated into the risk assessment procedure.

Table A3 Risk rating for impact of proximity to a watercourse on contamination of waterbodies by septic tank effluent

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

Slope

Although CMHC (2006) have not included slope as a factor in determining the risk of contamination of surface waters by septic tank effluent, slope has been identified as a very important issue in relation to this by Canter and Knox (1985). These authors argued that, on steeper slopes, septic tank systems are more likely to produce P laden runoff than on shallower slopes and suggest that risks at slopes of less than 5% are relatively low, with risk progressively increasing as slope increases. The values used to create a risk rating table for the effect of slope on the likelihood of contamination in this project is given in Table A5.

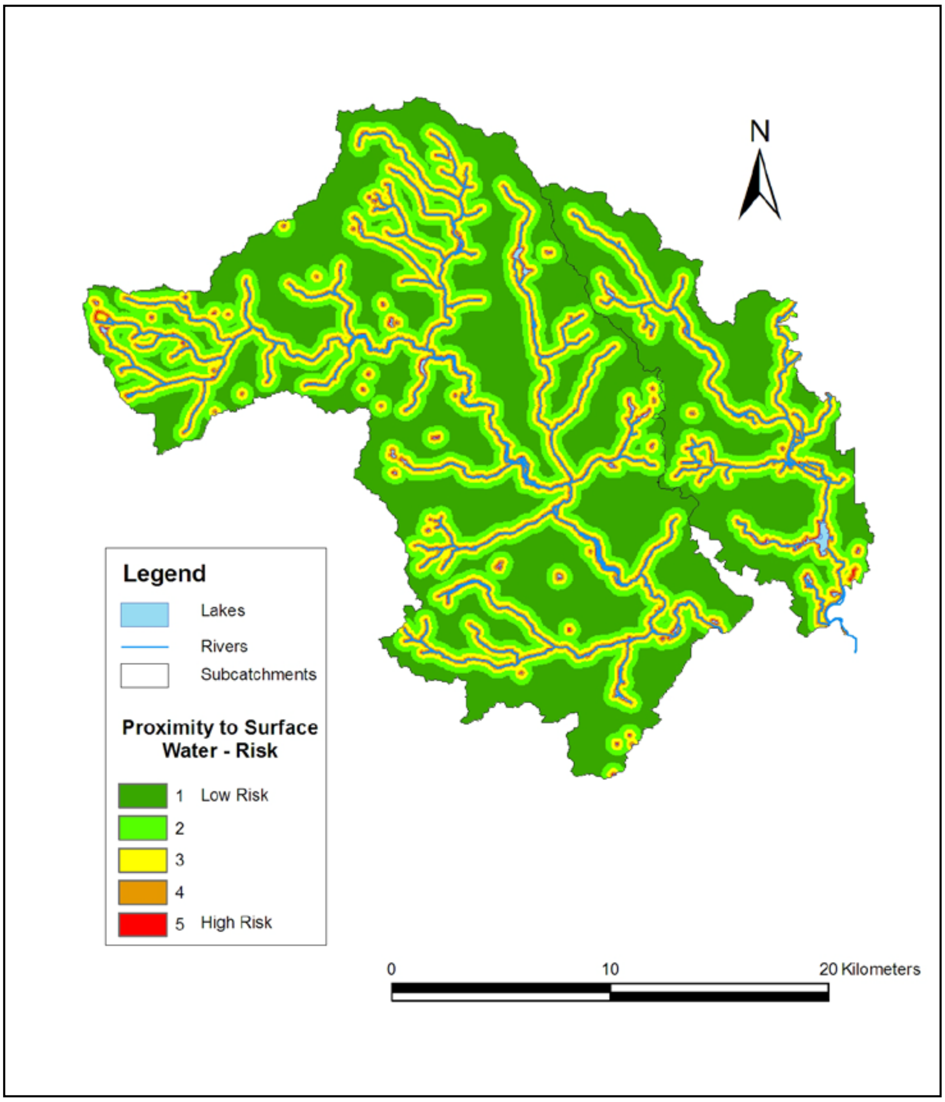

Figure A2 Map showing level of risk of septic tanks contaminating watercourses based on proximity to surface water body in the catchments of the rivers Ant and Bure.

Table A5 Risk rating for impact of slope on contamination of waterbodies by septic tank effluent

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

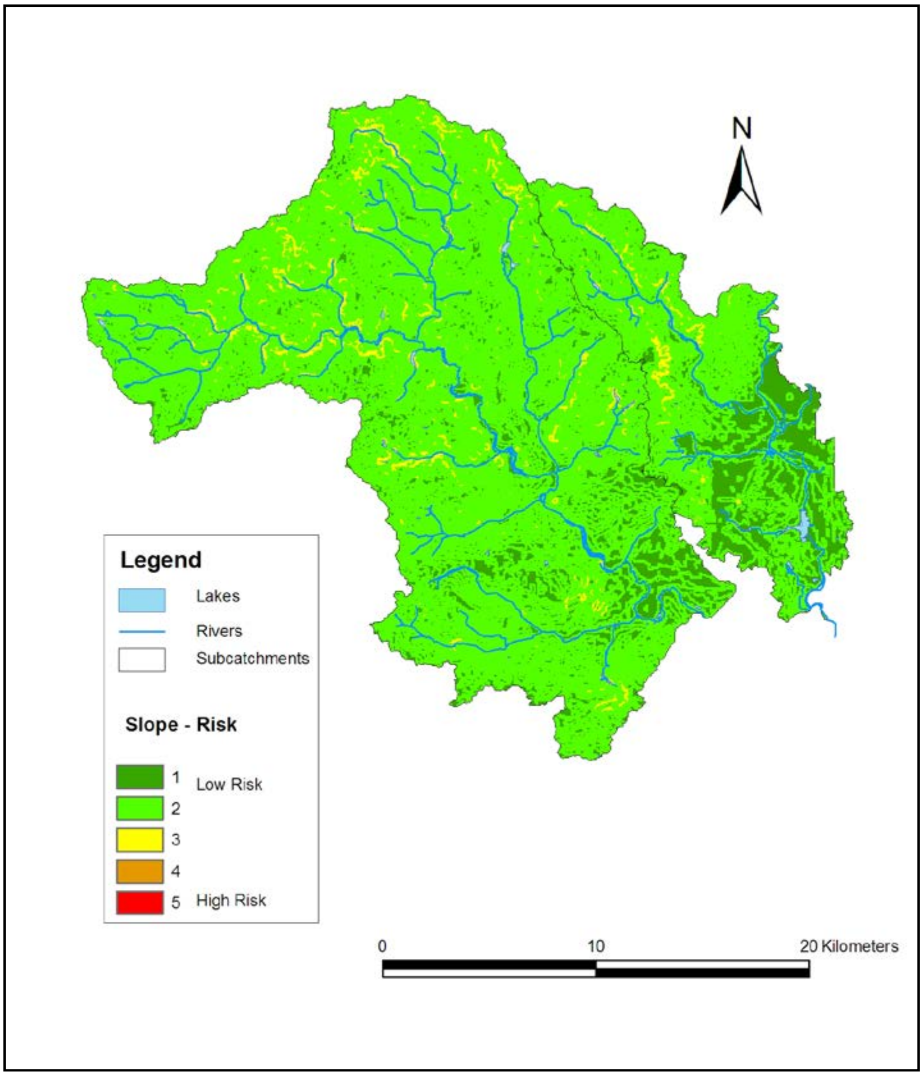

Figure A3 Map showing level of risk of septic tanks contaminating watercourses based on slope in the catchments of the rivers Ant and Bure.

A map showing the spatial distribution of risks across the Ant and Bure catchments is shown in Figure A3. Because The Broads are relatively flat, most of the area is at low risk of waterbody contamination by septic tank effluent being exacerbated by the slope of the terrain. In addition, some smaller areas with higher slopes have a medium risk of causing this type of pollution.

Density

The density of septic systems within a catchment also affects the risk of septic tank effluent polluting local watercourses (CMHC, 2006). In general, lower densities of tanks tend to cause less contamination of downstream waterbodies than higher densities of tanks (Arnscheidt *et al*., 2007), because areas with lower densities provide greater potential for P adsorption to soil particles generate more runoff which provides greater dilution potential for these discharges once they reach a watercourse. A suggested risk rating for different densities of septic tanks in rural areas, based on values given by CMHC (2006) and used in the current study, is shown in Table A6. A map showing the spatial distribution of these risk values across the Ant and Bure catchments is shown in Figure A4. Across most of the area the density of septic tanks, and therefore the risk of them causing pollution due to this factor alone, is relatively low. However, in some isolated areas the density of tanks is much higher, putting them at much higher risk of developing water quality problems due to discharges from this source.

Table A6 Risk rating in relation to tank density

Density of tanks (ha^{-1})	Risk rating
> 25	5
8 - 15	4
4 - 7	3
2 - 3	2
< 2	1

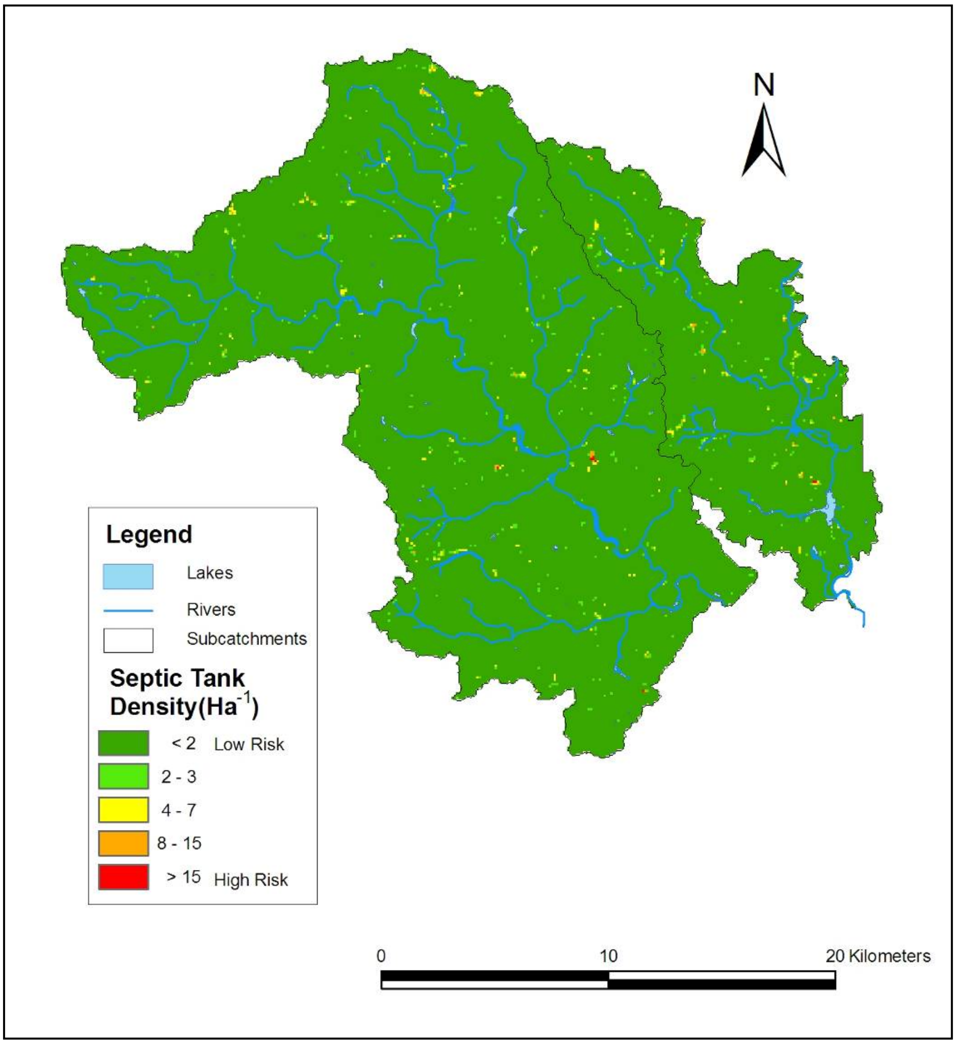

Figure A4 Map showing level of risk of septic tanks contaminating watercourses based on density per hectare in the catchments of the rivers Ant and Bure.

Weighting factors

Risk assessment weighting values for the factors outlined above were derived from those suggested by CMHC (2006) and from information on the relative importance of slope given by Canter & Knox (1985). These values, which are shown in Table A7, allow the different risk factors to be combined into an overall risk assessment value for a given area.

Table A7 Weightings for the main risk factors that affect septic tank contamination of waterbodies.

Risk factor	Weighting (% of total)
Soil hydrology	25%
Proximity to surface water	25%
Slope	25%
Density	25%

Overall risk assessment

By combining the risk factors and weightings detailed above, and using the approach of CMHC (2006) that is outlined in Section A1.1, a risk map was created showing areas within each catchment where septic tank systems are at highest risk of polluting watercourses due to the physical and hydrological characteristics of the area in which they are located and the density of septic tanks (Figure A5). This map provides information from which high risk areas could be identified for priority attention as part of a remediation strategy if resources to deal with the problem are limited. However, it should be noted that, as this approach has only been validated within the US, the modified approach proposed here requires further validation prior for use within UK. This is because it uses different datasets from those used in the US model and because key environmental variables, such as soil type, hydrological characteristics and rainfall patterns, differ between the UK and the US.

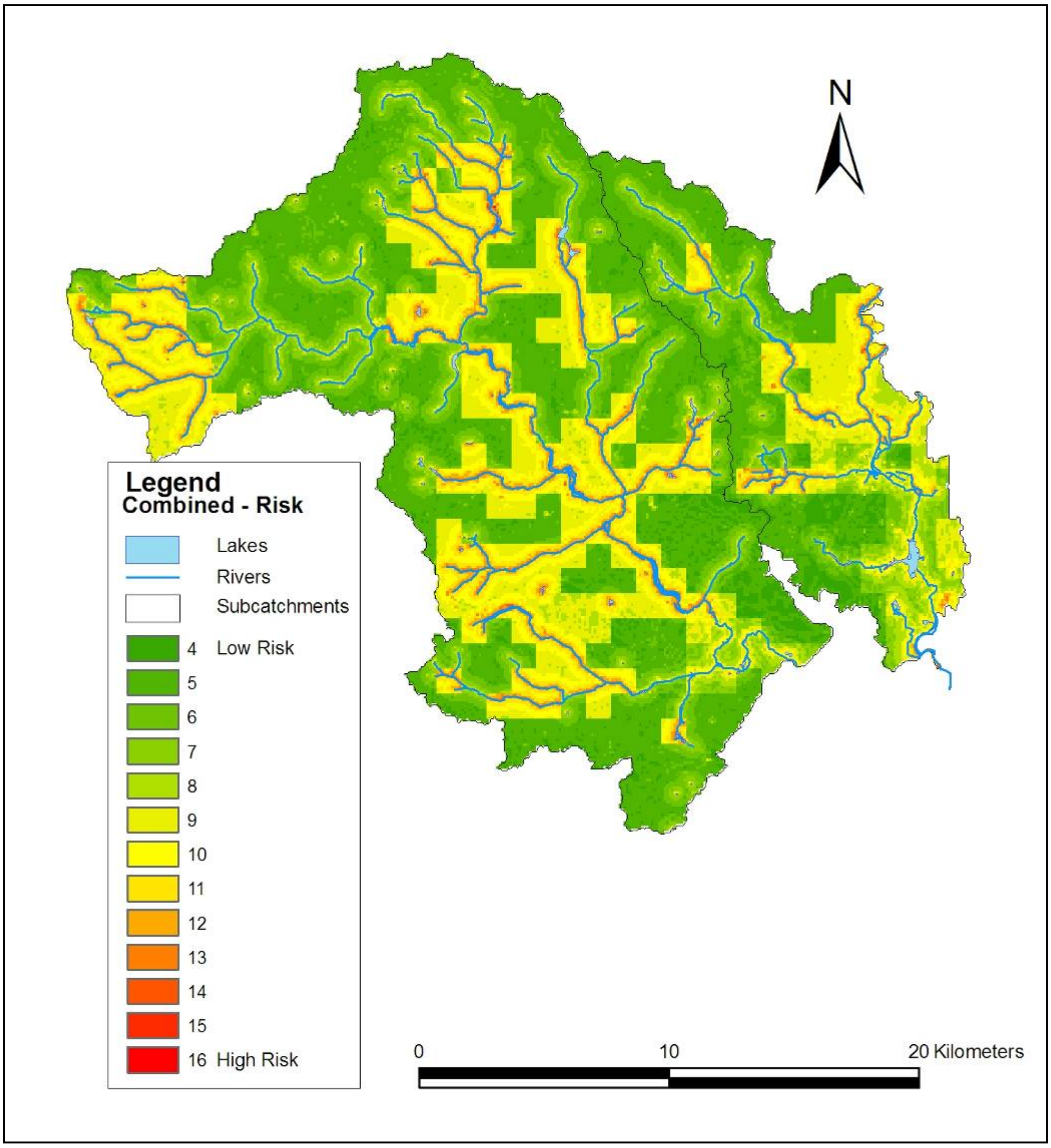

Figure A5 Map showing level of risk of septic tanks contaminating watercourses based on soil hydrology, slope, tanks density and distance from a watercourse.

A1.3.2 Assessment at the site-specific scale

Assessing risk at the site specific scale requires additional information about individual septic tank systems to be available, e.g. whether it is fit for purpose in relation to its age, size, condition; whether it is properly maintained in terms of drainage field management; how frequently it is de-sludged; whether appropriate household cleaning products are used. However, if these data are available, then it is possible to assess the risks associated with these factors, too, following the method proposed by CMHC (2006).

A proposed subdivision of these risk factors into different levels of risk rating is outlined below. There is insufficient data available to give each of these risk factors a weighting that reflects its relative importance within the overall risk assessment as applied at the catchment scale (above), so each risk factor has been subdivided across 3 risk ratings that reflecting high (5), medium (3) and low (1) risk of contamination. However, in view of the difficulty of combining these risk factors into a single value, the results are presented as a summary ‘traffic lights’ of individual risk factors in Table A16.

System age

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 that are less than 10 years old. The different risk ratings assigned to this factor by CMHC (2006) are summarised in Table A8. The authors also noted that, where the age of the on-site system is unknown, that age of the building that they serve can often be used as a good indicator of the age of the system itself.

Table A8 Risk ratings associated with system age.

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

Condition

Septic tank systems that are not kept in a good state of repair can leak untreated waste into the environment. Leaks can occur as a result of serious structural damage, such as large holes or cracks that result in significant discharge, or as a result of more minor problems, such as small cracks or broken seals. The only systems that are at very low risk of causing pollution problems are those that are watertight and in a good state of repair. The risk ratings shown in Table A9 reflect the likely impact of three levels of problem arising from the condition of individual septic tanks.

Table A9 Risk ratings associated with condition of tank.

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

Receives roof runoff

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. Systems that receive roof runoff will almost certainly overflow during heavy rainfall, in contrast to those that do not receive roof runoff. The risk rating for this factor has been subdivided into two categories, to reflect these two different scenarios, i.e. a high risk rating for ‘yes’ and zero risk rating for ‘no’ (Table A10).

Table A10 Risk ratings associated with receiving roof runoff.

Receives roof runoff?	Risk rating
Yes	5
No	0

Size for household

The size of a septic tank for the population that it serves is critical in determining whether or not it is likely to cause pollution problems. 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 – Approved Document H). This value is based on the expectation that the tank will be emptied on an annual basis. A tank that is too small ('small') will overflow unless emptied frequently, while one that is adequate ('medium') would be expected to cause occasional pollution problems. The only tanks that would be expected to have a low risk of causing pollution problems are those that have spare capacity to deal with all situations ('large'). For the purposes of this study it has been assumed that a tank that is less than 75% of the recommended volume is 'small', one at the recommended volume is 'medium' and one that is 125% of the recommended size is 'large' (Table A11).

Table A11 Risk ratings associated with tank size *per* household.

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

Maintenance of drainage field

To remain in good working order, drainage fields require on-going maintenance to manage vegetation growth, especially that of trees whose roots can damage underground pipes. In addition, pollution retention within these areas can be improved by alternately resting/dosing different areas of land. For the purposes of this project, the risk associated with lack of maintenance of drainage fields has been subdivided into 3 risk ratings according to the frequency of maintenance, i.e. never, occasional, frequent (Table A13).

Table A13 Risk rating associated with level of maintenance of drainage field.

Drainage field maintenance	Risk rating
Never	5
Occasionally	3
Frequently	1

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, causes the tank to overflow and discharge untreated waste. A proposed 3-level risk rating associated with the likelihood of pollution problems being caused by different frequencies of de-sludging are summarised in Table A14.

Table A14 Risk associated with different de-sludging intervals.

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

Use of appropriate household cleaning products

Although 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. It is, therefore, important that households use ‘septic tank safe’ products to reduce the risk of environmental contamination by their systems. The relative risk associated with different levels of usage of ‘septic tank safe’ cleaning products is summarised in Table A15.

Table A15 Risk rating for frequency of use of ‘septic tank safe’ cleaning products.

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

Overall risk assessment

The risks of waterbody contamination associated with the site specific factors outlined above are summarised in Table A16. The table is presented as a ‘traffic light’ system to enable problems that need to be addressed at each site to be quickly and easily identified. Those tanks that are at high risk of causing pollution (i.e. those with most red boxes) should be given highest priority for upgrade/repair in any planned remediation strategy.

In the future, it may be possible to give an overall risk rating for each septic tank by combining the scores for each of the individual risk factors outlined above. However, this requires each factor to be assigned a weighting factor in terms of its overall contribution to the problem as a whole. At present, there is insufficient information available for these weightings to be calculated.

Table A16 Potential risk of septic tanks contaminating waterbodies with P laden effluent at the site specific scale.

	Level of risk			
Attribute	**High**	**Medium**	**Low**	**References**
Design				
Age of tank	≥ 30 years	10 - 29 years	< 10 years	CMHC (2006)
Condition	Cracked, broken, leaking	Slight leak due to small crack or broken seal	Watertight and in good repair	Canter & Knox (1985)
Receives roof runoff	Yes	n/a	No	Canter & Knox (1985)
Size for household	Small (≤ 2 m^3)	Medium (> 2 m^3 - ≤ 3.4 m^3)	Large (>3.4 m^3)	Canter & Knox (1985)
Management				
Frequency of drainage field maintenance	Never	Occasionally	Frequently	Canter & Knox (1985)
Frequency of de-sludging	Rarely/never (more than 5 years intervals)	Occasionally (2-5 year intervals)	Frequently (< 2 years intervals)	Canter & Knox (1985)
Use of appropriate household cleaning products	Never	Occasionally	Always	Canter & Knox (1985)