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Practical measures for reducing phosphorus and faecal microbial loads from onsite wastewater treatment system discharges to the environment

A review

Scotland's centre of expertise for waters

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Research Summary

Key Findings

Onsite wastewater treatment systems (OWTS), the majority of which are septic tanks, are a contributing factor to phosphorus and faecal microbial loads. OWTS contribute to waterbodies failing to meet Water Framework Directive (WFD) objectives and as such, measures to improve the quality of OWTS discharges are required. Literature has been reviewed for a range of measures designed to reduce phosphorus and pathogen concentrations in effluent from OWTS. A feasibility assessment focussed on their application, effectiveness, efficiency, cost and ease of adaptation. A wide range of measures have been identified that could potentially improve water quality.

Results show no one solution could be applied to reduce phosphorus and pathogen loadings to the water environment. The literature suggests that OWTS need to be designed to the local flow and load characteristics of the effluents streams, as well as site specific conditions. With that in mind, measures such as awareness raising, site planning, and maintenance are likely to contribute to reduction of impact of OWTS on the environment. The level of load reduction possible from measures such as awareness raising is difficult to quantify, but it is low-cost and relatively easy to implement. Those most effective for phosphorus and pathogen removal are post-tank measures that maximise physical removal, through adsorption and filtering, and maintain good conditions for biological breakdown of solids and predation of pathogens.

A full summary of the measures reviewed is presented in Section 7 of the report. The following table presents a selection of the most practical measures to reduce P or pathogen releases from OWTS.

Measure	Removal of P possible	Removal of pathogens possible	Practicality	Site requirements	Cost	Likely uptake
Using P-free detergents	Yes - up to 50%	No	Legislation will ensure this is implemented	None	Low	Guaranteed
Reducing food waste flushed to drains	Yes - quantity unknown	Unknown	Awareness raising could assist. May be more practical for hotels, restaurants	None	Low - awareness raising	Possible with awareness raising
Appropriate site and setback distances	Likely	Likely	May require change in building regulations (linked to risk based approach)	Increased distance to water body	Related to increased land take and pipe distances	Possible
Risk based measures	Unknown	Unknown	Targeting measures to most at risk sites	None	Cost of consultation, deregulation	Currently being applied in England
Awareness raising	Unknown	Unknown	Practical if providing advice on operations, inspection and maintenance	None	Low if electronic; costs associated with leaflets or public events	Likely
Replacing old tanks with new tanks: Tank design (baffles and shape)	Yes	Yes	Practical where current system is poorly functioning. Baffles may be more practical for pathogen reduction than P reduction	Access for machinery and adequate space for new system	High	Possible
Increased Hydraulic Retention Time (HRT) – correcting misconnections	Yes	Yes	Practical as an inspection measure for site owners/ occupiers to improve function	Access to pipe connections and knowledge of OWTS	Low	Likely
Increased HRT - desludging	Unknown	Yes	Practical as a maintenance measure; may have unintended impact on P releases	Access to desludging equipment; consideration of end use of sludge	Relatively low	Likely

Measure	Removal of P possible	Removal of pathogens possible	Practicality	Site requirements	Cost	Likely uptake
Introducing chemical additives	Yes	Yes	Depends on site, scale of improvement required and dosing mechanism. In tank chemical use may destabilise microbes	Access for dosing, may be more suited to multi-chamber system. May require electricity	Medium (depending on additive and dosing frequency)	Possibly as a polishing step
Soak away, drainfield, or mound system	Yes	Yes	Could provide additional treatment at sites with direct discharges to water body	Land requirement, suitable soil conditions and slope. Electricity need if pumps used	High, depending on level of site work required	Likely
Lagoons/WSP	Yes	Yes	Depends on site and polishing requirement. Can allow for UV treatment or chlorination	Land requirement, and protections against exposure to pathogens	Installation and maintenance costs may be high	Possible
Removing P from discharged effluent using ochre	Yes	Unknown	Depends on site and polishing requirement, could dose in WSP or use as filter medium	Land requirement for treatment area, or dosing mechanism	High	Possible for additional polishing
Constructed wetland	Yes	Yes	Practical where space available, allows for increased retention time, facilitates increased absorption of both P and pathogens	Land requirement, protection against exposure to pathogens, substrates and vegetation harvesting over time. Electricity need if pumps used	Installation and maintenance costs may be high depending on system	Likely
Sand filter	Yes	Yes	Practical where adequate space allows	None	Installation and maintenance costs may be high	Likely
Peat filter	Yes	Yes				
Alternative filter media	Yes	Yes	Practical where space available, and proven to be safe (no additional pollutant releases)	Land requirement, electricity need if pumps used; consideration of filter material disposal	Installation and maintenance costs may be high	Possible with further evidence
Combination systems	Yes	Yes	Practical where adequate space on site allows	Land requirement higher for site with mixed treatments; electricity need if pumps used	May be high depending on system	Possible for sites in sensitive areas
Package treatment plants	Yes	Possible	May allow for treatment where limited space available onsite	Similar to septic tanks, requires electricity	Range of costs, can be cheaper than septic tanks to install, but maintenance costs may be higher than septic tanks	Possible

Introduction

The 2013 WFD classification identified 220 WFD baseline rivers and 71 baseline lochs in Scotland as being impacted by phosphorus in their chemistry and/or ecology. Faecal microbial loads are also recognised as a contributing factor to downgraded protected areas. In particular, pathogen pollution can result in contamination of bathing waters and shellfish waters, increasing the risk of human exposure to pathogens and associated impacts on industries such as shellfish growing.

Large numbers of properties in rural Scotland (estimated to be circa 160,000) are not connected to mains sewerage systems and instead rely on OWTS to process their domestic wastewater. These systems, mainly septic tanks, private sewage treatment works, and package treatment plants, are thought to contribute to the phosphorus and faecal microbial loads that impact on the status of WFD waterbodies and protected areas.

Research Undertaken

The project, in seeking to identify measures to improve OWTS discharges, considered:

1. The available measures for reducing phosphorus and faecal microbial loads from septic tanks and other OWTS.
2. An assessment of the feasibility of applying such measures to domestic households or larger private/ communal septic tanks, and the practicality of retrofitting any additional treatment.
3. Measures to deliver sustainable waste management solutions including energy generation and/or nutrient recovery that may reduce pressures on waterbodies and, at the same time, deliver value.
4. The load reductions which could potentially and realistically be achieved through each measure, individually and collectively.

Contents

Abbreviations

AD	Anaerobic Digestion
BOD	Biological Oxygen Demand
cfu	Colony forming unit
COD	Chemical Oxygen Demand
CW	Constructed wetland
DO	Dissolved oxygen
EC	European Community
EU	European Union
FC	Faecal coliform
FIO	Faecal indicator organism
FSA	Food Standards Agency
HF	Horizontal flow
HRT	Hydraulic retention time
LWA	Light weight aggregate
MEC	Microbiological electrolysis cell
MPN	Most probable number
N	Nitrogen
OP	Orthophosphate-P
OWTS	Onsite wastewater treatment system
P	Phosphorus
pe	Population equivalent
PP	Particulate-P
PTP	Package treatment plant
RBC	Rotating biological contactor
SBR	Sequential batch reactor
SEPA	Scottish Environment Protection Agency
SWPA	Shellfish Water Protected Area
SRP	Soluble reactive phosphorus
SS	Suspended solids
SSF	Sub-surface flow
ST	Septic tank
STE	Septic tank effluent
STS	Septic tank system (tank plus soakaway)
TC	Total coliforms
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UV	Ultraviolet
VF	Vertical flow
WFD	Water framework Directive
WSP	Waste stabilisation pond

1 Introduction

Many properties in rural areas of Scotland are not connected to mains sewerage systems and therefore rely on onsite wastewater treatment systems (OWTS) to process their domestic wastewater. These systems include septic tanks (STs) and private sewage treatment works or package treatment plants (PTPs). OWTS can be a potential source of environmental contaminants. In Scotland, phosphorus (P) and faecal pathogen pollution are of particular concern. Phosphorus pollution can result in loss of environmental quality and amenity. Pathogen pollution can result in contamination of bathing waters and shellfish waters, increasing the risk of human exposure to pathogens and associated impacts on industries such as shellfish growing.

Phosphorus is the main pollutant responsible for the downgrading of surface freshwater quality under the Water Framework Directive (WFD). It is SEPA's responsibility to identify pressures and appropriate mitigation measures to allow downgraded waterbodies to reach Good Ecological Status. The 2013 WFD classification identified a total of 220 out of 2406 baseline rivers and 71 out of 334 baseline lochs in Scotland as being impacted by P for their chemistry and/or ecology (B. McCreadie 2014, pers. comm. 17 Nov) (Figure 1).

There is increasing evidence that, in addition to agricultural runoff, P from small point sources such as septic tanks, may be an important input to waters in rural catchments. P discharges from septic tanks are likely to be high in soluble (bioavailable) P compared to agricultural runoff, giving them greater potential to degrade ecological water quality (Stutter et al. 2014). At the national scale, the contribution from STs is approximately 1% of loads discharged to the environment, but this figure can rise to up to 48% for individual waterbodies. Thirty six waterbodies have been identified as having P inputs from STs that are greater than 10% of the local catchment P load. Overall, it is estimated that P discharged (mainly as soluble reactive phosphorus (SRP)) from STs contributes about 4% of the total diffuse P load to surface waters in Scotland (SNIFFER 2010). This is mainly in rural catchments in north east Scotland where there is a relatively high population density and numbers of OWTS, in addition to intensive agriculture.

In addition to P, sewage can contain large numbers of faecal pathogens. Contamination of drinking water, shellfish, bathing waters and aquatic amenity sites with septic tank effluent increases the likelihood of waterborne illnesses being transmitted to human populations. There are also potential

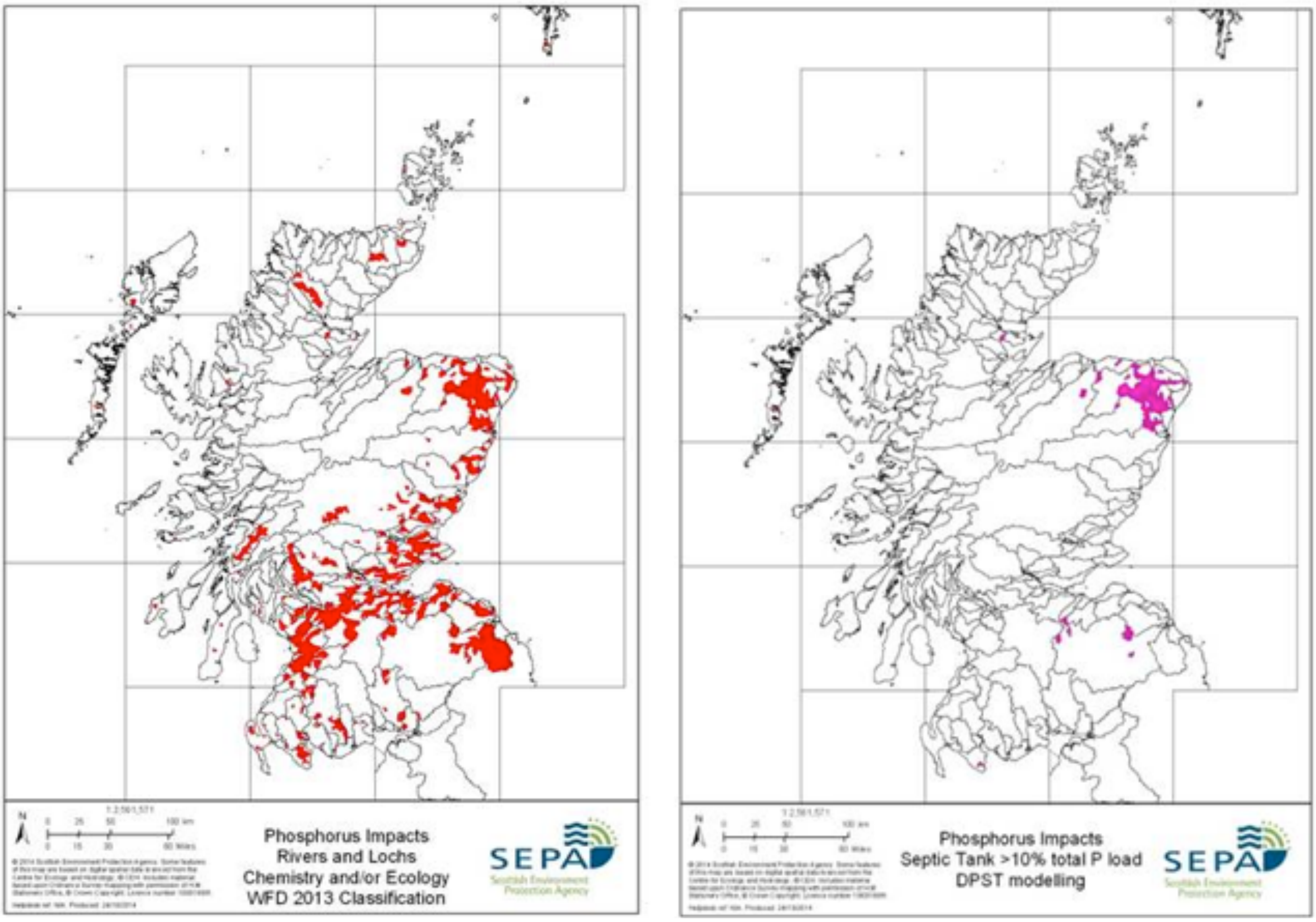


Figure 1 Left panel: Location of WFD rivers and lochs in Scotland that are affected by elevated phosphorus concentrations. Right panel: Impacted water bodies where the contribution of P from septic tanks is estimated to be > 10% of P load.

environmental impacts from pathogen contamination that are less well studied, such as impacts on the structure of microbial communities, mineral cycling and soil fertility (Lowe et al. 2007).

Microbial pollution (*E. coli* and intestinal enterococci) can be an important factor in the downgrading of bathing and shellfish waters. Direct discharges to small coastal streams and waters are likely to be a contributing pressure on shellfish waters, especially in rural areas on the west coast of Scotland.

In Scotland, the environmental standards for inland, coastal and transitional bathing waters provide an indication of acceptable levels of faecal indicator organisms (FIO) in the environment. The Scottish Bathing Waters Standards to be met by 2015 are displayed in Table 1 (Scottish Environment Protection Agency (SEPA) 2014).

Table 1 Scottish Bathing Water Standards

	Bathing water standards (cfu /100 ml)		
Pathogen	**Excellent***	**Good***	**Sufficient+**
Enterococci (inland water)	200	400	330
Enterococci (coastal and transitional)	100	200	185
E. coli (inland water)	500	1000	900
E. coli (coastal and transitional)	250	500	500

**95-percentile evaluation; +90-percentile evaluation*

Of eighty-four classified bathing waters, twenty-one do not meet "sufficient" status and are thus classified as being of "poor" status in 2015 according to the revised Bathing Waters Directive. The locations of these are shown in Figure 2. A further six currently are classified as "sufficient" but at risk of becoming "poor".

Under the Water Environment (Shellfish Water Protected Areas: Designation) (Scotland) Order 2013, eighty-four sites have been designated as Shellfish Water Protected Areas (SWPA); waters requiring protection to ensure the quality and productivity of shellfish. These designations include previously designated sites, extended sites and new designations. For these sites, the Scottish Government is currently designing new environmental standards. Proposed standards for these are listed in Table 2, however these are subject to change.

In 2013, fifty-three of the eighty-four SWPAs were considered to be of either "not sufficient" (2) or "sufficient" (51) status. SWPAs with waters at less than "good" status are considered to be impacted and may require the implementation of measures to improve their status. These sites are generally found on the west coast and the Shetland Islands (Figure 3).

Table 2 Proposed standards for SWPAs (Scottish Government 2013a)

	E. coli /100g of flesh and intra-valvular liquid		
Pathogen	**Good (Class A)**	**Sufficient**	**Not Sufficient**
E. coli	≤230	> 230 and ≤4600	> 4600

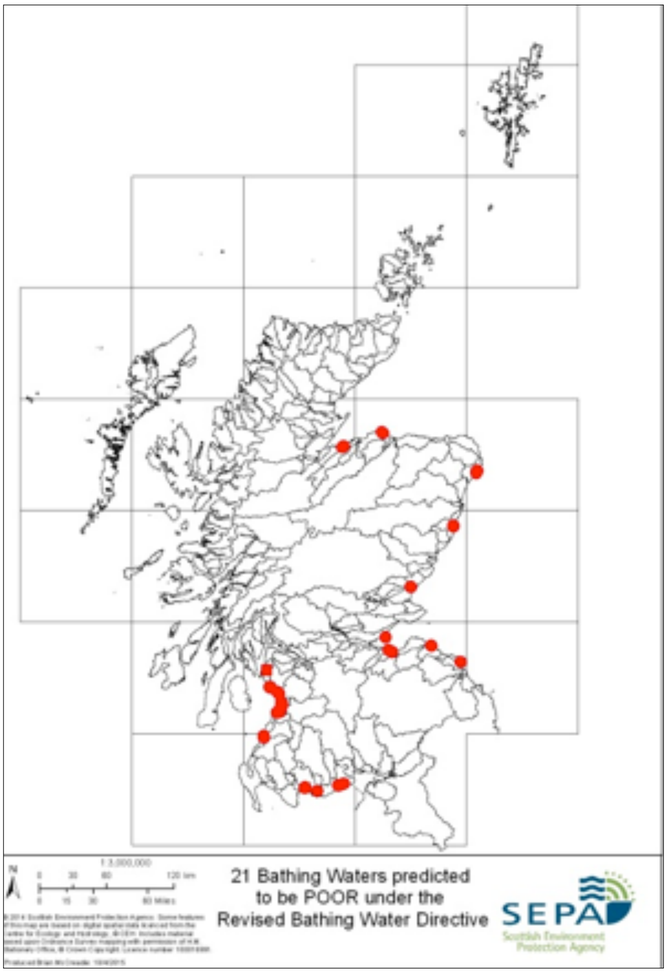


Figure 2 Bathing Waters predicted to be poor status under the revised Bathing Water Directive (B. McCreadie 2015, pers. comm. 10 April).

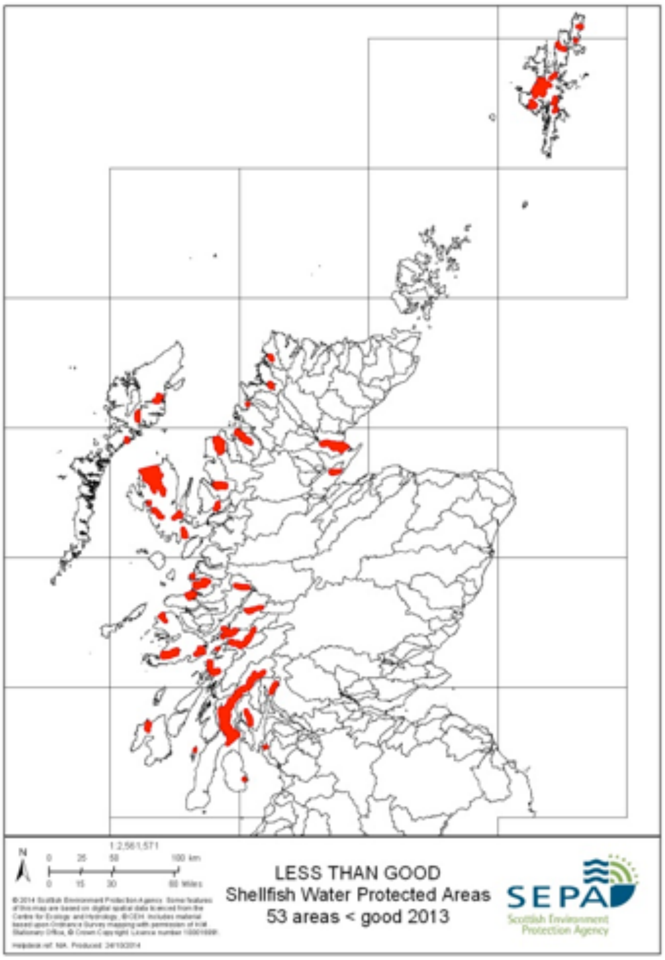


Figure 3 Shellfish waters at less than "good" status based on 2013 classification data.

The relative impact of septic tanks on SWPA failures is likely to be significant due to the absence of intensive agriculture and urban activities within the highly impacted areas. The contribution of septic tanks to total microbial loads within a water body is difficult to calculate with precision, but it has been estimated that 23.5% of the diffuse source *E. coli* load or 7.6% of the total load (diffuse and point source) to Scottish groundwaters and surface waters may result from OWTS discharges (SNIFFER 2006). Prevention of septic pollution of the environment is, therefore, essential in order to ensure environmental standards are achieved.

SEPA modelling predicts there are approximately 161,000 private sewage discharges in Scotland (B. McCreadie 2014, pers. comm. 28 Nov). Through the registration process, SEPA has detailed information on 61,819 private sewage discharge locations (Table 3). Of these, 76% discharge to soakaway or land and 21% discharge to inland waters. In addition, 91% of all registered discharges and 80% of discharges directed to inland waters undergo primary treatment. Only 18% of discharges to inland waters receive secondary treatment.

Based on the information presented in Table 3, it is likely that most of Scotland's 161,000 private sewage discharges undergo only primary treatment. SEPA has set default levels of treatment for small registered OWSTs that discharge to, or impinge upon, identified water bodies. These are shown in Table 4.

Table 3 Locations and treatment types of registered private sewage discharges in Scotland (B. McCreadie 2014, pers. comm. 18 Nov)

Type of discharge location	Private untreated	Private preliminary	Private primary	Private secondary	Private tertiary	Number of discharge locations	Percentage discharge location
Sea	64	1	861	88	8	1,022	1.7
Inland water	118	3	10,439	2,439	196	13,195	21.3
Soakaway/ land	351	8	44,481	1,990	102	46,932	75.9
Insufficient data	11	2	466	179	12	670	1.1
Total	544	14	56,247	4,696	318	61,819	100
% of all discharges	0.9	0.0	91.0	7.6	0.5	100	

Table 4 Currently accepted levels of treatment for small registered sewage effluents of ≤15 pe (SEPA 2014)

Treatment	Design
Septic tank and proprietary filtration system	As described in the Building (Scotland) Regulations 2004 Domestic Technical Handbook
Secondary treatment followed by reed bed treatment	Designed in accordance with requirements of the BRE, Good Building Guide (GBG) 42, Parts 1 and 2 or equivalent
Membrane bioreactor treatment plant system	Membrane treatment to achieve ultrafiltration of effluent
Other treatment system	Must demonstrate, to SEPA's satisfaction, they treat effluent to the required microbiological standard

2 Aims

Septic tanks contribute P and faecal microbial loads to waterbodies across Scotland. SEPA require a review of the available information to identify which additional treatments could be implemented at the domestic or larger licensed/communal scale to reduce their impact. A review of how effective these measures are likely to be in reducing discharged loads from STs is also required. This work will help SEPA to develop appropriate pollution mitigation strategies, predict improvements to water quality, set targets for improvement, and achieve GES.

The main aims of this project are to:

- Identify which mitigation measures are available for reducing P and pathogen loads to the environment from septic tanks and other types of OWTS
- Determine whether these measures are feasible/practical for use by single domestic or larger private/communal onsite treatment systems, especially in relation to retrofitting
- Estimate the level of load reduction that could, potentially and realistically, be achieved through the implementation of each measure, individually and collectively
- Consider the potential for measures to deliver sustainable waste management solutions, including opportunities for energy generation and/or nutrient recovery that may be of value to customers.

3 Sources of data and information

A literature review was undertaken to assess current evidence on the effectiveness of different OWTSs in removing P and pathogens from domestic wastewater before it is discharged to the environment. The primary sources of information for this report were refereed publications and academic literature, including journal articles, conference proceedings, reference books and government publications. Input from project partners, especially SEPA and Scottish Water, has also been included. Other sources of literature included unpublished reports from regulatory bodies, conservation agencies, research organisations and utility companies, and *ad hoc* leaflets providing advice to householders, such as those currently being circulated in many rural areas.

4 Septic Tanks

Septic tanks usually consist of a one- or two-chamber system (Figure 4) that holds sewage for a short period of time. This allows the solids to settle as sludge in the bottom of a tank, where it undergoes anaerobic digestion, with oil and grease forming a scum at the top.

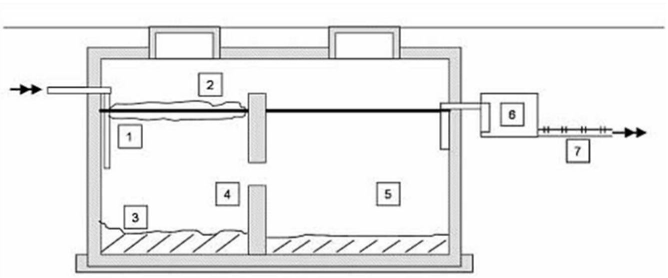


Figure 4 A standard two chamber septic tank design.
1. inflow;
2. floating scum;
3. settled sludge;
4. connection between chambers;
5. secondary chamber;
6. outflow and effluent inspection
7. soakaway or drainage system;

(reproduced from Hilton et al. unpublished)

Raw ST effluent contains suspended solids, dissolved P and nitrogen (N), and potentially contains pathogenic bacteria and viruses. Effluent quality is dependent upon the constituents of the raw wastewater that enters the septic tank and the degree of 'purification' that occurs within it. The main sources of P in raw domestic waste water are summarised in Table 5.

Examples of published P concentrations (mg/l) in influents and effluents from OWTSs are shown in Table 6.

Table 5 Source apportionment of P in raw domestic waste water (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%

Table 6 Examples of published P concentrations (mg/l) in influents and effluents from OWTSs

Influent TP concentration	Effluent concentration SRP	TP		Type of system	Notes	Reference
Unknown	1.9	3.3		Old ST; no soil adsorption bed	Concentrations entering a ditch (field drain discharge, including ST effluent); average from 1 year	Ockenden et al. 2014
Unknown	1.4	1.9		Old ST supplemented with modern tank	As above	Ockenden et al. 2014
Unknown	4.83 (0.32-10.56)	9.06 (4.45-18.01)		ST	Median concentrations from 4 STs	Brownlie et al. 2014
Unknown	8.82 (2.26-11.91)	11.86 (5.79-14.43)		ST with mechanical mixing	Median concentrations from 1 ST; 4 month monitoring period	Brownlie et al. 2014
Unknown	5.54 (1.42-10.60)	9.31 (1.91-14.44)		ST with chemical dosing and tank with aeration and filter system	Median concentrations from 2 STs; 4 month monitoring period	Brownlie et al. 2014
19.1 (13.05-25.8)	Unknown	12.2 (3-39.5)		ST	Average concentrations based on literature search (n=8 for influent, n=49 for effluent)	Lowe et al. 2007
13.3[a] 6.6 26.8 18.2	Unknown	7.07[b] 5.5 24.0 14.0	0.22[c] 0.04 1.22 0.02	Filter bed system	Results for 4 of the systems tested	Jenssen et al. 2010
Unknown	11.6 14.5 9.4 13.4 10.7 6.6	15.0 18.4 17.4 15.0 12.9 11.6		ST (concrete) ST (brick) ST (concrete) ST (brick) Klargester PTP Unknown	Sampled STs chosen from a range of locations across England	May et al. 2014
Unknown	Unknown	9.00		ST	Estimated value for 3 bedroom household using a desk based calculation method described in Section 4.1.2.	Brownlie et al. 2014
		4.50		Secondary treatment		
		1.80		Tertiary treatment		

[a]septic tank effluent; [b]outlet of biofilter; [c]outlet of filter bed.

A wide range of bacteria are essential to support the biological processes that occur within a septic tank such as the digestion and breakdown of complex organic compounds into simpler compounds. Some bacterial groups derived from human waste can be harmful or disease causing (i.e. pathogens). Human health risks are a primary driver for measurement and treatment of pathogens released from wastewater treatment systems into the environment. The bacterial groups of interest in this study are *E. coli* and intestinal enterococci, which are used as faecal indicator organisms (FIO) in shellfish and bathing waters in Scotland.

There is limited recent data available on the influent and effluent concentrations of pathogens from septic tanks, particularly for *E. coli* and enterococci. Table 7 provides estimates of pathogen strength in raw wastewater and basic septic tank effluent from the literature. Onsite treatment systems may show a higher level of variability than centralised treatment works due to variations in household water use and wastewater production on a local scale.

Table 7 Mean pathogen concentrations in raw wastewater and septic tank effluent (UK and Ireland)

Parameter	Mean concentration (cfu /100 ml) in raw wastewater	in septic tank effluent	Reference
Total coliforms	3.9×10^7	2.5×10^7	Kay et al. 2008
	-	7×10^8	Gill et al. 2007
	$2.0\text{-}3.5 \times 10^8$	-	Kadam et el. 2008
Faecal coliforms	1.2×10^7	-	Harrison et al. 2000
	$2.0\text{-}8.0 \times 10^7$	-	Kadam et el. 2008
	1.7×10^7	7.2×10^6	Kay et al. 2008
	-	2.9×10^5	Pundsack et al. 2001
Enterococci	1.9×10^6	9.3×10^5	Kay et al. 2008
	1.0×10^6	-	Blanch et al. 2003
E. coli	$1.2\text{-}3.3 \times 10^6$	-	Kadam et el. 2008
	-	5.0×10^5	Gill et al. 2007

Previous work by SNIFFER (2010) estimates the average concentration of septic tank effluent to be 1×10^8 cfu /100 ml microbial contaminants, with OWTSs contributing 23.5% of the diffuse microbiological loadings to receiving waters. This contribution will vary significantly by water body, density of OWTSs, and level of treatment.

Additional background information on the workings of septic tanks and septic tank pollutants is provided in Annex 1.

5 Mitigation measures for reducing pollutant loading to the environment from septic tanks and other OWTS.

There are a number of reasons why inadequate treatment of effluent by onsite wastewater treatment systems (OWTSs) occurs. These could include inadequate maintenance and desludging frequency, saturated or compacted drainage fields, inadequate capacity, structural problems within septic tanks (STs) such as broken baffles or lids, site specific issues related to soil properties, proximity to a water body, or inadequate separation from the water table (Ahmed et al. 2005, Swann 2001). Hydraulic failures, caused by clogged drainage fields or distribution systems can result in the discharge of untreated wastewater, the backup of wastewater into systems or surface breakthrough in the drainage field. Sites that have poor hydraulic retention due to soil type (i.e. sandy soils under rapid or high hydraulic loading) can allow effluent to move rapidly towards receiving waters as subsurface plumes. Systems that have not been designed with adequate capacity may be overloaded too rapidly resulting in releases of untreated wastewater, or site conditions may allow movement of partially treated effluent into the water table. This section explores the measures available for reducing discharges of phosphorus (P) and pathogens from OWTS into the environment.

5.1 Measures for reducing phosphorus discharges to the environment

Processes inside a ST break down particulate P into soluble P, with soluble reactive phosphorus (SRP) comprising about 85% of the total phosphorus (TP) in the discharged effluent (Canter and Knox 1985). Most aquatic systems are naturally low in biologically available P. So, when P availability increases, aquatic plants tend to grow rapidly and cause degradation of water quality (e.g. algal blooms). The most bio-available forms of P are those that are already dissolved in water or are readily soluble. In general, most bio-available P in the environment is in the form of SRP (Reynolds and Davies 2001). Most other forms of P, including the particulate P found in raw sewage, has limited availability to plants for growth until the breakdown processes inside a sewage treatment system transforms it into soluble P.

Septic tanks without soakaways are inefficient at removing P from wastewater (Canter and Knox 1985) and can be a source of P inputs to surface and groundwaters (Canter and Knox 1985, Ockenden et al. 2014). The potential level of importance of the pressure from septic tanks on waterbodies is well illustrated by an example from the River Chew, Somerset, UK. Here, decommissioning of a series of STs and directing the effluent to a nearby waste water treatment facility in 2002 resulted in a significant reduction of orthophosphate-P (OP) concentrations in the river, decreasing from an average of 251 µg P /l before 2002 to an average of 86 µg P /l after 2002 (May et al. 2010, Withers et al. 2014).

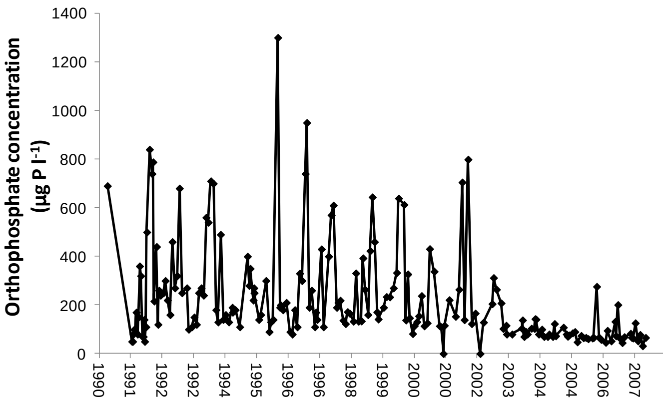


Figure 5 The impact of a first time sewerage scheme on orthophosphate concentrations in the River Chew (after Barden 2007).

In some areas of Scotland there are policies in place to limit discharges of P from STs to water. For example, the Kinross Area Local Plan (2004) recommends and enforces a desk based calculation procedure for estimating P output from OWTSs as part of local planning regulations (Box 1). The aim of this policy is to prevent any increase in P inputs to Loch Leven from new developments in rural areas that require OWTSs (Brownlie et al. 2014). In countries such as Sweden, statutory effluent criteria have been introduced for OWTSs. These require 70-90% removal of P from the effluent of a traditional ST to give a final effluent TP concentration of less than 1 mg/l (Nilsson et al. 2013a). Average effluent concentrations of 1 mg/l are also required in many parts of Norway (Jenssen et al. 2010).

BOX 1 Kinross Local Area Plan procedure for estimating P output from OWTS

The P output (g /day) is calculated by multiplying a person equivalence (pe) value, based on the number of bedrooms in the property (n) where pe = n +2, by an estimated *per capita* water usage value of 180 l/day and a likely TP discharge concentration according to OWTS type. The OWTS type is divided into those with primary (septic tank only), secondary (OWTS with wetlands, reed beds or mechanical treatment plants) and tertiary treatment (OWTS with sand filters, drum filters, membrane systems, chemical dosing), with pre-defined TP discharge concentrations of 10 mg/l, 5 mg/l and 2 mg/l for each of these treatment types, respectively (Brownlie et al. 2014). According to this calculation procedure, the TP concentrations in septic effluent from, for example, a three bedroom house with secondary treatment would be 4.5 mg/l, compared to 9 mg/l from a three bedroom house with primary treatment. These expected concentrations can be measured for regulatory purposes.

It should be noted that the P concentrations of ST and PTP effluents vary over time (Heistad and Paruch 2006, Gill et al. 2009a) and that a single instantaneous sample may not provide a reliable estimate of P discharges from these systems in the longer term. The reason for this is unclear.

5.1.1 Pre-tank measures for reducing phosphorus discharges to the environment

This section examines measures that can be implemented prior to effluent reaching an OWTS.

Change of diet

Vegetarian households have been shown to produce around half the amount of P in their sewage compared to households that consume meat (Cordell et al. 2009). Although the impact of this on the P content of OWTS effluent has never been quantified, it has been raised as a concern in relation to consequent environmental damage in countries that are increasing their consumption of meat products, such as China and India. The estimated P output from humans is summarised, and compared to that from cattle, in Table 8.

Table 8 Estimated P output from cattle and human sources (kg/capita/year)

Source	Urine	Faeces	Excreta Vegetable based diet	Excreta Meat based diet	Reference
Cattle	3.1	5.3	-	-	Cole and Todd (2009)
Human	0.7–1.0	-	-	-	Karak and Bhattacharyya (2011)
Human	0.3	0.16	-	-	Jönsson et al. (2004)
Human	-	-	0.3	0.6	Cordell et al. (2009)

It is unlikely that change of diet is a practical solution to reducing P discharges from OWTSs within the UK.

Using phosphate free detergents

Since the 1980s, there have been increasing concerns about the contribution of phosphates present in domestic waste water (especially those coming from laundry and dishwasher detergents) to the eutrophication of waters that receive effluents containing domestic waste. Initially, this resulted in a voluntarily reduction in the phosphate levels in detergents by the cleaning product manufacturers. By 2008, the amount of phosphate in raw sewage that was attributable to dishwasher and laundry detergents was estimated to be 7% and 18%, respectively (Defra 2008).

In 2010, the EU announced its intention to ban phosphates in domestic cleaning products with effect from 1 January 2013. In 2012, this resulted in amendment Regulation (EU) No. 259/2012 of Regulation (EC) No. 648/2004, which placed restrictions on the phosphate content of domestic laundry and dishwasher detergents. Limits were set at ≤ 0.5 grams per standard dosage for laundry detergents from June 2013 and ≤ 0.3 grams per standard dosage for dishwasher detergents from January 2017 (European Union 2012). Regulation (EU) No. 259/2012 was transposed into UK law in 2013 and is expected to result in a significant reduction in the amount of phosphate in raw domestic sewage over the next few years. As the level of P input to OWTSs tends to be reflected in the amount of discharged P, it is expected that the use of phosphate-free detergents in line with this new legislation will decrease the amounts of P being discharged from these systems and entering the environment.

A ban on the sale of detergents containing more than 0.5% of P in sixteen states in the U.S. supports this assumption. This resulted in a P reduction in wastewaters of about 40-50%

(US EPA 2002). Foy et al. (2003) indicated that a reduction in the use of P-based detergents within the catchment of Lough Neagh, Northern Ireland, was probably the main reason for the decrease in P loads from households to the lake between 1990 and 2000. This decrease was observed despite an overall increase in the size of the rural population in the area over this period. A positive effect of using P free detergents was also observed by Alhajjar et al. (1990) in a study comparing the effluents from newly constructed STs serving four volunteer households that used phosphate based detergents with four that used P free products. The use of phosphate free detergents reduced average concentrations of TP in ST effluents from 19 mg/l to 9 mg/l and average concentrations of OP from 18 mg/l to 8 mg/l. There was also a difference in proportions of soluble to particulate P in the effluent, with TP in effluents comprising 95% OP if P based detergents were used and 88% OP if non-P detergents were used.

The results outlined above suggest that encouraging householders to use P free detergents will help reduce levels of P in OWTS influent and, as a consequence, levels of P in OWTS effluents in the short term. In the longer term, new EU legislation (European Union 2012) will ensure that the amount of P in cleaning products is reduced over the next few years, with very low values being achieved by 2017.

Reducing P inputs to OWTSs to reduce effluent P concentrations is likely to be effective and result in a significant reduction in P-loading from domestic premises and will result from new EU legislation that limits the P content of household detergents.

Reducing phosphate additions to domestic water supplies

One factor that is increasing P inputs to OWTSs within the UK is the addition of orthophosphate to domestic water supplies to reduce plumbo-solvency in areas where water supplies are still delivered, at least in part, by lead pipework. Such treatment of drinking water can lead to high P concentrations in tap water, although the amount of orthophosphate added varies across regions. Average concentrations as high as 1.9 mg/l have been reported in some areas (UKWIR 2012), which is equivalent to about 20% of the P concentration in the effluent of an average septic tank (May et al. 2010).

The results of a study comparing P concentrations in the effluents of three STs in England to the P content of the corresponding domestic tap water found relatively high levels of SRP, ranging from 0.9 to 1.1 mg P/l in the tap water, and a sufficiently strong positive relationship between tap water and effluent P concentrations to raise concern and justify further investigation (May et al. 2014). In light of proposals to reduce levels of lead in drinking water, higher additions of orthophosphate to domestic water supplies may be required to achieve new lead standards. There is some evidence that the reduction in lead standards may be difficult to achieve, even with phosphate dosing, and lead pipe replacement/ rehabilitation may be the only option for achieving these standards in the longer term (UKWIR 2010). The cost of carrying out lead pipe replacement/rehabilitation may be offset by avoided costs of extra phosphate dosing and subsequent wastewater treatment. A potential consequence of reducing phosphate in tap water could be the impact on copper concentrations in effluent. Any reduction in P-dosing could result in increased copper concentrations in drinking water; however the level of impact has not been quantified (UKWIR 2010).

Given the human health implications of lead in tap water, reducing levels of orthophosphate additions is unlikely to be a viable option in the short term for reducing P inputs to OWTSs. In the longer term, a programme of lead pipe replacement or rehabilitation may reduce the requirement for tap water dosing.

Reducing levels of food waste being flushed down the sink or toilet

Although the effect of putting food waste down the sink on the P content of OWTS effluent has never been tested, this practice will increase the P input to the OWTS and is likely to result in higher effluent P values. For this reason, avoiding disposal of food waste down the sink or toilet is often listed amongst best management practices for on-site systems when guidance leaflets are distributed to householders. This could also be a consideration for rural businesses such as hotels and restaurants with high levels of food preparations. In-sink macerators are likely to allow more food waste to be flushed into a septic tank system. Although these systems will be banned under the Waste (Scotland) Regulations from use to dispose of food waste to sewer as of 1 January 2016, domestic premises and food producers in rural areas are exempt from the regulation. Best management practice guidance for these properties, particularly in at-risk areas, could include information on the use of macerators and disposal of food waste.

The effect of reducing levels of food waste being flushed down the sink or toilet on effluent P concentrations is unclear, but this is a cheap and viable option and should be explored.

5.1.2 In-tank measures for reducing phosphorus discharges to the environment

This section examines measures related to improving the P removal performance of a ST.

Anaerobic digestion in OWTSs, which results in partial degradation of organic materials, involves several groups of micro-organisms that convert complex organic matter into, mostly, methane and carbon dioxide (Appels et al. 2008). These groups are sensitive to, and can be inhibited by, several process limiting parameters such as temperature, alkalinity, pH, concentrations of hydrogen, sodium, potassium, ammonia, sulphite, heavy metals, volatile fatty acids (Appels et al. 2008) and level of enzyme activity (Diak et al. 2012). Physical removal processes such as precipitation may also be influenced by in-tank parameters. The potential for manipulating these process limiting parameters to reduce P discharges from OWTSs is discussed below.

Manipulating pH

The optimal pH range for microorganisms taking part in digestion processes varies depending on their group. For example, while less sensitive fermentative microorganisms can function in a wider range of pH values, between 4.0 and 8.5, others such as methanogenic bacteria are extremely sensitive to pH and their optimal range is much narrower (between 6.5 and 7.2) (Appels et al. 2008). The exact relationship between pH and the efficiency of breakdown processes that occur within OWTSs, and how this affects P retention and discharge, is unknown and requires further investigation. In addition to effects on biological processes, pH is important to chemical precipitation processes (discussed later under '*Introducing chemical additives*'). P removal by calcium (Ca) precipitation occurs at alkaline pH between 7 and 12 (Higgins et al. 1976, Johansson and Gustafsson 2000, Moon et al. 2007) although iron precipitation of P may dominate at a lower alkaline pH (i.e. pH 7-8) (Bastin et al. 1999). If chemical additives are used to reduce P concentrations, the effectiveness may be impacted by effluent pH.

Managing pH to improve the efficiency of waste breakdown in the tank may be possible, but the effects of this on P concentrations in effluent are unclear and may lead to an increase in the discharge of soluble forms of P.

Managing in-tank temperature conditions

Temperature can influence digestion processes of organic materials in two ways: by having an effect on the physicochemical properties of the digestion substrate compounds, and, by influencing metabolism and the growth rate of microorganisms (Appels at al. 2008). Benefits of higher temperatures include increasing the solubility of the organic components and enhancing reaction rates of biological and chemical processes. However, high temperatures can lead to an increase in free ammonia, which is an inhibiting factor for the microbial population. A stable temperature is important for the bacterial communities within OWTSs, as they can be adversely affected by sharp and frequent fluctuations in temperature (Appels et al. 2008).

Heistad and Paruch (2006) monitored TP concentrations in the effluent from a 7 m^3 septic tank over a 21 month period that received domestic wastewater from a house in Norway. They found that the observed TP concentrations in the effluent ranged from 2 mg/l to about 11 mg/l and varied seasonally, with lower values recorded in winter and higher in summer. This may indicate a possible effect of seasonal changes in temperature on the rate of decomposition of sewage within the tank. However, temperature seems to be an unlikely factor influencing such changes in effluent quality, because tanks are usually buried quite deeply in soil, and are often set in concrete, which should provide good insulation from short term seasonal changes in air temperature.

Although variations in temperature can affect in tank processing and effluent P concentrations, it is unclear how this can be managed to reduce P discharges to the environment.

Controlling waste water retention time

The amount of time that waste is retained in the holding tanks of OWTSs affects the amount of processing that it undergoes before being discharged to the environment. Retention time is affected by the size of the tank and the volume of wastewater passing through it over time. The appropriate size of a tank, in terms of meeting the needs of the population that it serves, is critical for its effective performance and for reducing impacts on the environment. The current recommendation for ST volume is 2.7 m^3 for a household of up to four people, plus an additional volume of 0.18 m^3 per person for each additional user (The Building Regulations 2000 – Approved Document H). This recommended value is based on the assumption that a ST will be desludged on an annual basis, because a build-up of sludge reduces the effective volume of the tank and, as a consequence, its retention time. In addition, a tank with insufficient capacity will overflow if it is not emptied frequently, whereas an adequately sized tank would be expected to reduce the risk of P pollution problems. However, there is no evidence to support this assumption or to indicate the optimal size of a tank for a given household.

Many older OWTSs receive additional water from roof runoff. Although input of rainfall may dilute the effluent P concentration, it will not decrease the amount (load) of P being discharged to the environment. The flushing through of additional solid material may overwhelm the infiltrative capacity of the soakaway, the size of which may not have been calculated to accommodate this additional input (May et al. 2010). Systems that receive roof runoff are likely to overflow during high rainfall events.

Regular desludging of OWTSs is often recommended to householders as a way of maintaining the effective performance of their systems and reducing environmental pollution. When full of sludge, OWTSs do not function properly because retention times are reduced. As higher levels of sludge accumulate in these systems, reducing the effective volume of the tanks, the flushing rate will increase and the distance between the influent pipe and surface of the sediments will decrease. This will create increased disturbance of settled material during high flushing events caused by sudden discharges of domestic wastewater or roof runoff into the system. This can result in increased levels of organic matter being discharged in tank effluent and in turn can lead to clogging of drainfields and poor infiltration, which affects the contaminant absorption function of the drainfield (Withers et al. 2014). Although design standards for STs are generally based on annual desludging, advice on this varies. Beal et al. (2005) suggested that desludging of septic tanks should occur every three to five years and this has been the observed frequency in some catchments in Scotland (May et al. 2010). Scottish Water recommend that this should occur every one to two years (Dudley and May 2007), while the US EPA (2002) recommends that tanks should be emptied when they are more than 30% full of solids (Withers et al. 2014).

The link between waste water retention time and effluent quality is unknown, especially in terms of P discharges. However, if a link can be found, it is feasible that retention time can be managed effectively by diverting roof runoff, and optimising tank size and the frequency of desludging.

Optimising frequency of desludging

Regular desludging of OWTSs is assumed to increase the efficiency and effectiveness of waste breakdown processes and improve the quality of effluents, especially in terms of P discharge. Although it is important to recognise that desludging is beneficial to the effective operation of OWTSs, (e.g. it reduces the amount of solids passing into the soakaway and the risk of hydraulic failure), the impact of desludging on P concentrations in the effluent is unclear. Canter and Knox

(1985) suggested that increasing the efficiency of breakdown of the solids may be more likely to increase, rather than decrease, the concentrations of soluble P discharged. Consequently, it is possible that regular desludging may increase the amounts of P discharged to the environment. There is a need to determine the optimal frequency of desludging when all of these factors are taken into account.

A study of 28 ST systems (STS) conducted in Ireland (Gray 1995) showed a steady decline in the accumulation rate of sludge in septic tanks tested over time, due to increases in the degradation of solids within the tank, increased compaction of settled solids and, to some extent, losses of solid material from the tank. The sludge accumulation rate in these septic tanks decreased from 0.254 l/capita/day/6 months after desludging to 0.178 l/capita/day/60 months after desludging. Gray (1995) suggested that increasing the desludging interval resulted in less sludge production, an increase in sludge age and, consequently, a much more stable sludge. There is a risk that, as sludge ages more suspended solids may be discharged from the outflow as a result of the disturbance by methane gas bubbles generated in the sludge. This may increase the amount of particulate P discharged from the system. Gray (1995) suggested that incorporating slightly larger primary chambers into septic tanks (compared to the 2720 l septic tank recommended for a population of four) would allow an optimal desludging interval to be achieved. He estimated this to be more than five years. Increasing the frequency of desludging and using multi-chambered tanks, which reduce the possibility of solids being carried to soakaways (Gray 1995), could also decrease the concentrations of total P leaving the tank.

There is little information in the literature about the impact of desludging on effluent quality, especially in terms of P concentrations. Further research is needed to provide information on the impact of sludge age on the breakdown of in-tank solids and its effect on P discharges to the environment. The effect of desludging interval on P discharges may differ for particulate and soluble P. It should be noted that optimising in tank processes to reduce sludge accumulation may, as a side effect, increase P discharges to the environment.

Frequency of desludging affects the level of sludge accumulation within the tank and the effectiveness with which particulate waste material is broken down. However, decreasing rates of sludge accumulation by breaking down solids more effectively into soluble components that can be discharged from the outflow, may conflict with aims to increase P retention in the tank.

Introducing chemical additives

A variety of septic tank additives are commercially available and widely used by householders to 'improve' waste processing in their septic tanks (Diak et al. 2012). The ability of phosphorus to bind to some elements, such as aluminium (Al), iron (Fe) or calcium (Ca) provides the potential for chemical amendments to be used to increase P retention within septic systems (Eveborn et al. 2014). Chemical additives, such as aluminium sulphate (alum) and sodium aluminate (Long and Nesbitt 1975), can be used to increase P retention within a tank through a chemical precipitation process (Brandes 1977). As previously mentioned, the process efficiency can be impacted by effluent pH. The chemically precipitated P is incorporated into the sediment at the bottom of the tank, which is subsequently removed during desludging. This has been shown to reduce TP concentrations in septic tank effluents by as much as 85% (Brandes 1977). Azam and Finneran (2014) demonstrated that adding ferric iron amendments to enhance biological removal of P by Fe (III)-reducing microorganisms, promoted phosphate removal from wastewater in septic tanks. This process involves the adsorption of the phosphates present in solution by reducing iron (Fe(II)) and precipitation of the stable mineral vivianite ($Fe_3(PO_4)2{\cdot}H_2O$), which then accumulates in septic solids for removal during desludging (Azam and Finneran 2014). The use of chemical additives may allow for an increase in P removal from discharge water, however, there is an increase in sludge production, which may increase the required desludging frequency. There is also an element of chemical handling required by householders.

Despite the apparent effectiveness of the chemical precipitation in reducing P concentrations in tank effluent (85% reduction in effluent TP), the application of chemical amendments to tanks raises concerns about personal and environmental safety associated with the use of some of these additives, in addition to considerations on increased frequency of desludging.

Introducing biological additives

The purpose of most biological additives available for adding to OWTSs is to enhance the efficiency and rate of biological activity within septic tanks through the addition of microorganisms, e.g. bacteria or enzymes (Pradhan et al. 2011a). Biological additives are relatively inexpensive and simple to use. They are believed to enhance tank performance by increasing metabolic activity and digestion rate, and helping to maintain a healthy population of microorganisms (Diak et al. 2012). There is very little published research on the effectiveness of these biological additives in the field. Most assessments have focused on laboratory based process studies or literature reviews and assessments, often conducted by product manufacturers and not substantiated by an independent third party (Pradhan et al. 2011b). Pradhan et al. (2008) observed that there was no significant impact of three different bacterial additives on total bacterial concentrations within 48 full-scale septic tanks during a field experiment. In addition, Pradhan et al. (2011a) found that the overall effects of such additives on decomposition of sludge and scum across a range of septic tank maintenance levels were non-significant. However, reduced sludge accumulation rates were observed for two out of three additives tested, when added to tanks that had been desludged within the 2-3 years prior to the additive usage (Pradhan et al. 2011a).

Degradation of organic compounds during anaerobic digestion can decline due to decreased enzyme activity within the holding tanks. This is because enzymes are needed to break up organic compounds (proteins, lipids and carbohydrates) into small molecules that are available to microorganisms for utilisation. Decreased enzyme activity can result in the incomplete degradation of biodegradable solids and their increased accumulation, together with non-biodegradable solids, over time (Diak et al. 2012). Although it is likely that decreased enzyme activity within the tank affects P outputs, there are no available data to confirm this. In a laboratory based study, Diak et al. (2012) investigated the effect of enzyme additives on the performance of septic tanks. Enzyme additives are believed to increase hydrolysis (solubilisation of suspended particulate matter) and digestion rates within tanks, and help maintain a healthy microbial population. The results showed no significant effects of added enzymes on these processes in the sludge.

A further type of biological additive that is available commercially, although not as common as other biological or chemical additives, is worms. Worms are believed to improve treatment of sewage sludge and food waste by increasing the break-down of organic matter (Worm Smart Waste Systems 2014). The suppliers suggest that one of the advantages of using worms is that it is a 'one-time' solution (Septic Tank.co.uk 2014). Once inside a waste treatment system, these organisms are believed to reproduce and maintain a viable population, removing the need for regular dosing. The solutions offered by companies advertising worms for waste treatment include specially designed waste treatment systems with worms, e.g. 'Wormsmart' (Worm Smart Waste Systems 2014) and 'Worm Farm Waste System' (A & A Worm Farm Waste Systems 2014), or more simple kits containing worms with bedding material that can be added to an existing tank, e.g. 'Soakaway Worms' (SepticTank.co.uk 2014). The latter solution assumed that worms added will move through the soakaway pipes, cleaning out the accumulated material inside and keeping the effluent free flowing. In contrast, the waste treatment systems are designed as single chamber, non-mechanical systems that process sewage and other organic waste by using worms to enhance the breakdown of solid waste. The treated effluent from these systems can then be put through a low pressure sub surface irrigation system (e.g. see the 'Wormsmart Waste System') and reused as an underground liquid fertiliser. These treatment systems can also have an above ground entry chute installed that allows for easy disposal of food and garden waste. In both a traditional septic tank and a specially designed treatment system, the worms can only survive in the presence of oxygen.

None of the studies reviewed looked at the impact of biological additives, either bacterial or enzymatic, on the level of P in tank effluents or implied any potential effects of such additives on nutrient concentrations. No studies have been found that assess the effects of worms on P concentrations in system outflows.

There is no evidence that biological additives, such as microorganisms, enzymes or worms, affect the level of discharge of P from OWTSs. By increasing the level of breakdown of solid wastes, they may lead to an increase in the discharge of soluble forms of P.

Replacing old tanks with new tanks

The design life of many older septic tanks (mainly built from brick or concrete) was approximately 10-15 years when they were installed (Canter and Knox 1985), although many have been in constant use for much longer. In contrast, most new systems have been constructed from stronger and more watertight material designed to last for up to 50 years (Canter and Knox 1985). CMHC (2006) suggested that older septic systems have a relatively higher risk of failure than newer systems, with systems over 30 years old being up to 12 times more likely to cause water pollution issues than those under 10 years old.

A study conducted in the River Armagh catchment, in the border area of Northern Ireland and Republic of Ireland, showed that replacing high impact, defective septic tanks within a catchment with modern systems (sequential batch reactors) and polishing filters achieved a reduction in low flow P concentrations in nearby rivers by 0.032 mg TP/l (Macintosh et al. 2011). May et al. (2014) found some evidence that older tanks (among sampled tanks aged between 2 to 50 years) were discharging higher levels of P to the environment than those recently installed. The results were, however, not conclusive due to the small sample size that did not allow effect of tank age to be separated from that of other influencing factors, such as the lifestyle of the household and the level of repair and maintenance of the system.

Supplementing an old septic tank with a modern prefabricated one may help reduce the output of P from existing tanks that do not have a soil absorption system. This measure increases the capacity of the tanks and, as a consequence, extends the residence time allowing more particulate matter to settle out (Ockenden et al. 2014). Such an upgrade to an existing septic tank in Cumbria was shown to reduce the mean concentrations of TP and SRP of effluent entering a ditch wetland system from 3.3 mg/l and 1.9 mg/l to 1.9 mg/l and 1.4 mg/l, respectively, with little change in the TP/SRP ratio. These measurements suggested that allowing more particulate matter (and consequently more particulate P (PP)) to settle within the tank can decrease P discharged from a tank but does not change the efficiency of processes responsible for changing PP to SRP (Ockenden et al. 2014). In addition, the upgrading of the septic tank in Cumbria was combined with the construction of a wetland system. The latter measure was shown to be more successful in reducing P in effluent concentrations than increasing the capacity of the tank.

In contrast, to the above, a study on the effectiveness of seven OWTSs within the Loch Leven catchment, Scotland, found no significant difference in TP discharge concentrations from OWTSs with primary treatment (single septic tank only), secondary treatment (mechanical mixing) and tertiary treatment (fitted aeration and filter system, chemical dosing) (Brownlie et al. 2014). The results suggested that assumptions about the level of P discharge based on the OWTS type alone may not be accurate, and that efficient performance of OWTS and human domestic behaviour are two important factors influencing the quantity and quality of P discharged from septic treatment systems to the environment.

Nasr and Mikhaeil (2013) compared TP concentrations in the effluent from a 95 litre three-chamber septic tank to TP concentrations from a one-chamber septic tank. In theory, the smaller 'baffled' system should improve the level of contact between anaerobic accumulated sludge and influent wastewater, because the baffles force the wastewater to flow under and over the baffles between the inlet and outlet of the tank, resulting in faster accumulation of sludge. Despite the significant increase in removal of suspended solids (10-15% of difference in accumulated sludge volume) the differences in TP removal between the one-chambered and three-chambered tanks were small, 29.3% and 33.1% respectively, after 72 hours of hydraulic retention time.

Emerging technologies may be used to augment the treatment capacity of existing and new septic tanks, such as microbial electrolysis cells (MEC). It has been shown that the TP content of the output from a septic tank with a MEC, which provides electrochemical assisted anaerobic digestion, was on average 39% lower than the output from a conventional septic tank, when tested under laboratory conditions (Zamalloa et al. 2013). In this case, the MEC was installed in a septic tank using stainless steel mesh as electrodes. The implementation of a MEC coupled with a digester can facilitate production of H_2 and O_2 by applying an electric field (Zamalloa et al. 2013). Microorganisms consuming an energy source release electrons and protons. The protons are then reduced by the additional voltage supplied to the cell from an outside source. This leads to hydrogen production. The MEC has been designed to improve biogas production in septic tanks and, as a result, increase the potential of these systems for energy generation and organic matter removal. Experimental operation of such a system showed 77% removal of H_2S from the biogas in the MEC-septic tank compared to the standard ST and 500% higher biogas production. These results indicate potential of MECs to improve the quality of ST gaseous output, as well as generating small amounts of renewable energy (Zamalloa et al. 2013).

Replacement or supplementing of older septic tanks with newer systems may help reduce P losses to the environment by 20-50%. More research is needed to determine whether this results from improved in-tank waste water processing or a lower likelihood of tank failure and leakage.

Replacing old tanks with package treatment plants

Upgrading older style, traditional septic tank systems to modern alternative solutions, such as package treatment plants (PTPs) is often assumed to be more effective in treating domestic waste water due to the improved processing of influent. A PTP can be installed as an alternative to a ST, or to provide subsequent treatment of ST effluent before it is discharged to a soil soakaway. The removal of malfunctioning STs and replacement with PTPs has been advocated in some UK catchments (e.g. Loch Leven, Scotland) to reduce P loading to surface waters. This is based on the assumption that a PTP is more efficient at removing pollutants from effluent than a traditional septic tank. For example, a recent Scottish Natural Heritage report (SNH 2011) states that a discharge from a PTP will contain about 5 mg P/l compared to an average ST discharge of 10 mg P/l. Because discharges from PTPs are assumed to be 'safer' for the environment than those from traditional septic tanks, they are often allowed to discharge directly to water. The controlled aerobic environment in PTPs accelerates microbial degradation of organic matter. The design focus of modern systems is to reduce biological oxygen demand (BOD), and concentrations of suspended solids and ammonia in effluents, however P removal is not a main focus in a PTP design. With P removal levels less than 50%, it is possible that replacing traditional STs with PTPs might not achieve the expected reduction in P loads to the environment (Ockenden et al. 2014). There are also concerns about the reliability of their performance and challenges associated with reaching an average effluent P concentration of 1 mg/l, as is required in some countries (e.g. Sweden, parts of Norway; Jenssen et al. 2010).

The "Klargester Guide to EPP2: Water Discharge Consent" (Kingspan Environmental 2010) provides an example of performance results (percentage reduction) relating to their Biodisc® rotating biological contactor (RBC) PTP. These test results are summarised in Table 9. The Klargester PTP is shown to effectively reduce chemical oxygen demand (COD), BOD5, SS and NH4, by almost 90%, but the level of reduction of TP is low. Data collected from six septic tank systems sampled in England by May et al. (2014) seem to confirm this. Concentrations of both SRP (10.7 mg/l) and TP (12.9 mg/l) in the effluent from a Klargester Biodisc® system were similar to those recorded from the more traditional types of tank, which ranged from 6.6 to 14.5 mg/l SRP and from 11.6 to 18.4 mg/l TP, respectively.

Table 9 Klargester RBC performance results

Parameter	% Reduction in concentration
COD	89.4
BOD_5	95.7
SS	94.8
NH_4	88.6
TP	47.6
Total N	45.7

Akunna and Shepherd (2001) compared the treatment efficiency of a RBC PTP and a sequencing batch reactor (SBR) PTP. TP concentrations in influent were found to range from 13 to 18 mg/l. After 90 days operation, the effluent TP concentrations from the RBC ranged from 13 to 16 mg/l and from the SBR ranged from 1 to 2 mg/l. The SBR was found to be better at producing a higher quality effluent than the RBC, however system specific design features and operating conditions affected the SBR performance.

To assess and compare the efficiency of various types of OWTSs, such as STs, STs combined with additional treatments and PTPs, information on the P levels in influent and effluent is required. However, data on the P concentrations in raw sewage and discharged effluent are scarce, with most research being focused on determining the effectiveness of post-tank treatment (Lowe 2007). In addition to the above, reducing P discharge from OWTSs appears to be a complex problem and applying expensive engineering solutions as a sole measure, whilst significantly increasing the costs to householders, may not bring expected results. A comprehensive approach covering the impact of human domestic behaviour on P input and factors influencing the efficiency of OWTSs in retaining P (e.g. design, location, age, condition, maintenance, frequency of desludging, effectiveness of soil treatment systems) is required to better understand the risk that OWTSs pose to the environment and how to significantly reduce TP discharges from these systems (Brownlie et al. 2014).

Replacing traditional septic tanks with package treatment plants may reduce levels of P discharge from OWTSs by about 50% however some PTPs show little impact on P reduction. Based on limited data, SBR appear to show better removal potential than RBC systems. Additional research is required to confirm this.

5.1.3 Post-tank measures for reducing phosphorus discharges to the environment

This section examines measures that can be implemented once effluent is discharged from a ST.

Removal of a certain amount of P from influent waste water is likely to occur within STs, simply due to the adsorption of P to solids. In conventional OWTSs, most P removal occurs within the soil treatment system, drainfield or soakaway (Lowe 2007). Very little is known about how efficient septic tank systems (STS - ST plus soil soakaway) are at removing P from domestic wastewater, or how much P they release to surface waters. In theory, STSs can be effective in P removal if they are sited, used and maintained properly. However, there are many inter-related factors influencing the performance of STSs in removing P, such as proximity to the water table and surface water, capacity of the system compared to the number of people in the household, chemical composition of the waste water received by the system, frequency of desludging, and soil composition and grain size in the soakaway. In addition, effectiveness of the ST itself can be dependent upon several inter-related processes that are responsible for capturing solids and breaking down organic materials.

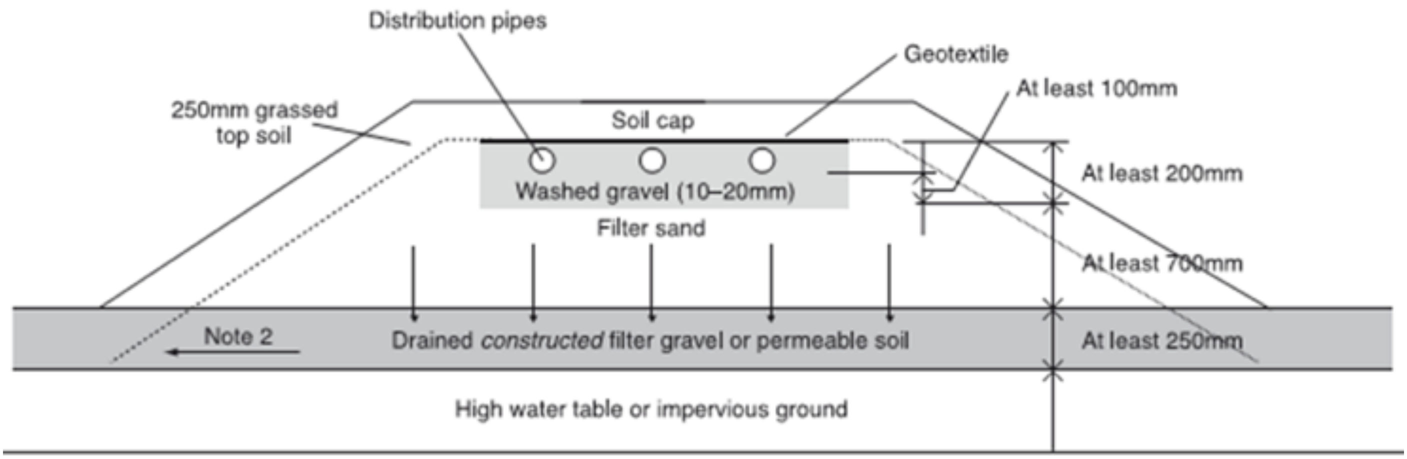
Installing a soil based soakaway

Septic tanks with natural soil infiltration systems (soakaways) are the most common on-site treatment measure for wastewater in developed countries (Eveborn et al. 2012). However, the level of effectiveness of soil infiltration for the treatment of septic tank effluents has been questioned in terms of their P removal efficiency.

Removal of P in soil infiltration systems is attributed to the processes of chemical sorption and precipitation within the soil matrix, such as the formation of Al(III) and Fe(III) (hydr)oxides surface complexes or precipitation of Al(III), Fe(III) and/or Ca phosphates (Eveborn et al. 2014). The physical characteristics of soil can affect the attenuation of contaminants, as can the chemical characteristics, for example, the amount of organic matter present and capacity for cation exchange (Beal et al. 2005). The P removal capacity of the soil systems can be easily exceeded (Eveborn et al. 2014). The long term performance of soil treatment systems and the number of years it takes to saturate the system depend mainly on the wastewater P load (i.e. a combination of the P concentrations in the wastewater leaving the ST and the hydraulic loading rate) and the soil volume available for treatment (Eveborn et al. 2014). The efficiency of P attenuation in soil can also be decreased when low soil permeability restricts dispersion and attenuation of septic tank liquids. In this case, the liquid effluents may by-pass the soil soak trenches and exert a high trophic impact on receiving waters (Palmer-Felgate et al. 2010).

Research conducted by Eveborn et al. (2012) on four filter beds aged 14-22 years with comparably high P loads, indicated a low overall capacity of soil to absorb P with, on average, only 12% of the long-term P load being removed. A study of six soil treatment systems (aged 11-28 years) in Sweden showed that P in the most P retaining soils systems accumulated mainly as aluminium phosphates or by association with aluminium hydroxides surfaces, with organically bound P being an important sink for P (Eveborn et al. 2014). Aluminium chemistry proved to have a significant effect on P removal efficiency, with the P absorption capacity and vulnerability to wash-out from sandy soil treatment systems correlated with the aluminium chemistry of the system (Eveborn et al. 2014).

An alternative to a soakaway is a mound system (Figure 6), which may be used particularly in areas where the water table is too high to meet Building Regulations for the installation of a soakaway. Mounds can allow for an increase in the aerobic contact distance between the discharged effluent, subsoil and the upper level of the water table.



Notes:

1. To provide venting of the filter, the upstream ends of the distribution pipes may be extended vertically above mound level and capped with a cowl or grille.
2. Surface water runoff and uncontaminated seepage from the surrounding soil may be cut off by shallow interceptor drains and diverted away from the mound. There must be no seepage of wastewater to such an interceptor drain.
3. Where the permeable soil is slow draining and overlaid on an impervious layer, the mound filter system should be constructed on a gently sloping site.

Figure 6 Drainage mound for septic tank effluent (after The Building Regulations 2000).

Little is known about the effectiveness of mounded systems in removing P from ST effluent in comparison to traditional soakaways, but a study by Alhajjar et al. (1990) suggested that mounded drainfields are not as efficient as conventional ones at absorbing P before it enters groundwater. The results of this study also showed that when phosphate containing detergents were used in households, P leaching from STS increased by a factor of 4.3 if a drainage mound was used. However, mounded soakaway systems may be needed in areas where the water table is too close to the soil surface for a traditional soil soakaway to work effectively. Research suggests that combining this with the use of phosphate free laundry and dishwasher detergents (see 5.1.1) will help decrease P leaching to groundwater.

Soil or mound based soakaway systems can provide some P removal, however effectiveness is site specific and depends largely upon local soil conditions. Over time, ground can become saturated and P removal capacity can decrease.

Installing constructed reed beds for effluent 'polishing'

Constructed wetlands (CW) can improve the quality of effluents discharged by OWTSs before they reach the wider environment, but they cannot be used as a primary source of sewage treatment. They are increasingly incorporated into sewage treatment systems in rural communities where the full-scale use of chemical and biological treatments of sewage effluent is uneconomical. Being efficient in removing nitrogen (N) from the effluents, constructed wetlands are, however, believed to be less effective in removing P (Cooper et al. 1996). Such systems can change from a sink to a source under certain environmental conditions (Gabriel et al. 2008) or as a result of saturation of P binding sites (Mann 1990).

Initial results of a study conducted by the Wildfowl and Wetlands Trust and the NERC Centre for Ecology & Hydrology at Slimbridge, Gloucestershire, showed the constructed wetlands installed removed 20% of TP, 5.2% of SRP and 100% of particulate phosphorus (PP) from the influent water (Duenas et al. 2007). The water reaching the wetland contained 0.78 mg/l of TP, 0.35 mg/l of SRP and 0.43 mg/l of PP. After 10 years of operation, however, the P retention values had fallen to 10% of TP, -40% of SRP and 50% of PP in the effluent water, indicating that the reed bed had become a source of SRP due to P release from decaying vegetation (Duenas et al. 2007). Maintaining wetlands by regular changes of substrate and cutting/removal of plant biomass is important to ensure good performance.

Ockenden et al. (2014) demonstrated that constructing a system of three small wetlands along, or adjacent to, an open ditch that collected drainage from a farm resulted in longitudinal improvements in P, with its concentrations decreasing between the inlet of the site and the outlet from the last wetland. Concentrations of TP in water going through the last of the wetlands were reduced by 60% during the first year. This was attributed to the long hydraulic residence time of the wetland, which was estimated to be at least 7 hours (compared to 3 minutes in an equivalent length of a ditch). Long hydraulic residence times increase opportunities for sedimentation and processing of nutrients, which was reflected in significant sediment and nutrient net retention within the studied wetlands over a period of three years (Ockenden at al. 2014).

Subsurface flow CWs, with pre-treatment bio-filters, have been shown to perform very well in Norway, where they have been pioneered. The investment cost of these CWs, however, is high due to their relatively large size and the use of P-sorbing light-weight aggregate, which is likely to require renewal over time due to saturation (Jenssen et al. 2005). The potential use of these systems has not been explored within the UK.

Constructed wetlands can be used for polishing nutrient laden effluent before it is discharged into the environment. P removal efficiency for TP and PP may decrease by about 50% after 10 years unless they are maintained through regular changes of substrate and the cutting/removal of plant biomass; they may also change from a sink to a source of SRP over the same period.

Removing P from discharged effluent using ochre

A number of studies have shown that successive use of iron-based (ferruginous) media can reduce P levels in waste water (Heal et al. 2004). Heal et al. (2004) investigated using air-dried 'ochre' (i.e. precipitated iron), recovered from settlement ponds and constructed wetlands designed for remediation of polluted mine drainage, as a low-cost reagent for the removal of P from sewage effluents. Such reagents could, for example, be placed in filter units or used as a substrate for a constructed wetland, and have the potential for re-use as a slow-release fertiliser for agricultural use. Phosphorus removal from sewage effluent by ochre occurs predominately by sorption onto iron and aluminium oxides and hydroxides, and calcium carbonate, and, to a smaller extent, by precipitation (Heal et al. 2004). In laboratory experiments, ochre demonstrated a high capacity for P removal, increasing with the level of P concentration in solution. The estimated maximum P adsorption capacity of the two types of ochres tested in the study were orders of magnitude higher than those determined in other wetland substrates, suggesting a high potential for additions of ochre to improve P removal in constructed wetlands. During a 9 month experiment that simulated a horizontal flow filtration system, ochre also proved to be an effective material for long-term P removal. The effectiveness of removal remained consistently high throughout the whole period of the experiment, with the concentration of P dropping from 20 mg/l in the applied solution to less than 1 mg/l in the effluent throughout. In addition, tests on the desorption of adsorbed P showed that only 2% was easily released to water. Heal et al. (2004) recognised the need for further research to scale up the laboratory findings to next step implementation, and to find a method of granulating ochre that would not compromise its P sorption properties and allow widespread applications. A granular form was suggested to address the potential vulnerability of ochre to erosion by running water and wind disturbance. Based on the high capacity of ochre for P removal demonstrated in the study, Heal et al. (2004) indicated that incorporation of this material into constructed wetlands has a high potential for improving their treatment efficiency in relation to P removal and highlighted the potential use of ochre for treating of sewage effluents more directly by dosing with ochre and mixing in holding tanks.

Chemical treatment of discharged effluent using products with an affinity for P, by direct dosing or by incorporation into another form of treatment such as a constructed wetland, may help to reduce P discharges from OWTSs to the wider environment by about 95%. This approach warrants further investigation.

Installing on-site filter systems

Filters can be used as on-site secondary treatment systems when space allows. Such systems include filter beds (Jenssen et al. 2010, Nilsson et al. 2013a) and/or subsurface flow CWs (Mæhlum and Stålnacke 1999, Kõiv et al. 2009). The advantage of using filters over conventional treatment methods are their large P adsorbing areas, typically long retention time, diverse microbiological populations and flexibility in alternating aerobic-anaerobic zones (Kõiv et al. 2009). Contaminants are removed from wastewater through physical, chemical and biological processes, especially sorption (Kõiv et al. 2009). Robertson (2012) reported an example of an effective post-tank treatment system, where almost 100% of the lifetime P loading to the domestic STS was adsorbed and retained by a sand filter bed within 2 m of the effluent distribution pipes. The filter media that are most commonly used for P removal from wastewater can be categorised as natural materials, industrial by-products and man-made products (Vohla et al. 2011). Performance of filters may vary, with filter particle size, hydraulic retention time and rates of organic load likely to impact P removal efficiency (Jenssen et al. 2010, Nilsson et al. 2013a, Nilsson et al. 2013b).

Examples of different filter media efficiency in P removal in subsurface flow CWs (from the review conducted by Vohla et al. 2011) are presented in Table 10. Using a pre-treatment filter, e.g. a vertical flow pre-treatment system, to reduce the amount of organic material in wastewater was demonstrated to improve the efficiency of filter beds and subsurface flow CWs in terms of P and other substances removal (Mæhlum and Stålnacke 1999, Nilsson et al. 2013a). Mineral-based sorbents incorporated into filter systems can be easily replaced and have potential to be reused as an agricultural fertiliser (Nilsson et al. 2013b), which is particularly important as global resources of P are diminishing (Cordell and White 2011).

Table 10 Examples of P removal efficiency by different filter materials (adapted from Vohla et al. 2011)

Material category	Material	Study type	P retention
Natural products	Alunite	Batch	Average of over 80% removal
	Apatite	Batch/column	0.28-1.09 g P /kg material in 24 hours isotherms
	Bauxites (raw and activated)	Batch	Adsorption for raw and activated bauxite: 0.82 and 2.95 mg P /g, removal > 95% for activated bauxite
	Gravel	3 gravel based CWs, batch	-40 - 40% P removal, adsorption capacity: 25.8-47.5 mg P /g
	Gravel	Batch	33-50% P removal, sorption 3-3.6 g P /kg in 24h
	Limestone	Full-scale CW (SSF wetland cell treating wastewater from dairy farm, 1.5 years)	P retention on average 4.3%, mean reduction 14.5%
	Marl gravel	Full-scale CW	37-52% P removal efficiency
	Peat	Domestic wastewater lab-scale biofilter (1 month), household biofilter	P removal in lab-scale filters: 44%, biofilter: 12% for TP
	Peat	Small-scale CW in field (landfill leachate)	TP reduction during the first 6 months: 77% from sludge water, 93% from biopond water
	Peat	A vertical flow (VF) and horizontal flow (HF) filters (municipal wastewater and landfill leachate)	58% and 63% P removal, P binding capacity 0.081 g /kg, decrease of P removal efficiency after 6 months of operation
	Sand	Full-scale CW, horizontal subsurface flow (HSSF) sand filter	P in soil after 8 years: 0.117 g P /kg, 72%
	Sand	Batch	8.6-27.2% P removal or 2.45 g P /kg
	Sand	Full-scale CW, HSSF sand filter	P in sand after 5 years: 52.8 mg /kg, purification efficiency 78.4%
	Shellsand	Batch; Meso-scale CW in field, HSSF filter in greenhouse for household	0.8-8 g P /kg in 24h; 335 g P /kg, saturated before 2 years, Ca-P: 37-57%, Al-P: 8-23%
	Wollastonite	Batch	PO_4-P reduction in 20 h: 90-93%, P-sorbed: 0.1-12,000 mg /kg
	Wollastonite tailings	Full-scale CW (wastewater from dairy farm)	12.8% soluble P retention, mean reduction: 27.5%
Industrial by-products	Fly ash (acidic)	Batch	PO_4^{3}-immobilization: 75-100% in 24h
	Fly ash	Full-scale CW, 3 stage system, one filled with fly ash	Majority of TP absorbed by fly ash, TP removal about 83%
	Iron ore		67% (aerated) and 53% (anaerobic)
	Ochre	Batch	0.026 g P /kg, 90% of all forms were removed after 5 and 15 min shaking
	Blast furnace slag	Small-scale CW (dairy farm wastewater, 7 months)	Up to 72% of P was retained, working area 200 μg P /g
	Blast furnace slag	Full-scale CW (7 months)	Slag filters reduced TP up to 99%
Man-made products	Filtra P	Column	Retained 98.2% of P, effluent pH fell from 12.9 to 11.6, clogged after 971 pV
	Filtralite P™	Small, meso and full-scale CW	Extracted P: 3887 mg P /kg (small-scale), 4500 mg P /kg (meso-scale), 52 mg P /kg (full-scale)
	Filtralite P™	Full-scale CW	P reduction in the filter system: 99.4%
	Leca (Estonian)	Full-scale CW, VSSF + HSSF filter bed, 2 years	89% TP removal
	LWA	Full-scale CW (wastewater from households, 4 years)	89%
	Leca (Swedish Norlite)	Full-scale CW (wastewater from dairy farm)	34% P removal

On-site filter systems based on specially engineered media have been suggested as an effective alternative to natural soil infiltration methods and these are now commercially available in Sweden, Norway and US (Nilsson et al. 2013a). Jenssen et al. (2010) tested the performance of nine filter bed systems operating in four Scandinavian countries for three years. The systems benefitted from two kinds of filters sequentially combined with other components in the following order: a septic tank, a pump well, a vertical flow single pass aerobic bio-filter, a subsurface horizontal or vertical flow filter and an outlet well (Figure 7 and Figure 8). The bio-filter component contained a 0.6 m deep, light-weight, aggregate filter in the size range of 2-10 mm and a distribution system, with the latter (or optionally both elements) confined in a fibreglass dome or tank. The subsequent flow saturated filter consisted of light-weight aggregate Filtralite®P that was specially designed for sorption of P in constructed wetlands (Jenssen et al. 2010). The filter beds were 1 m deep, with volumes varying from 5 to 40 m^3. They were covered with grass, but could have been planted with wetland vegetation instead.

The filter beds, both 'compact' (5-10 m^3) and 'normal' sized (20-40 m^3), showed high average levels of performance, with removal of organic matter (measured as BOD) >80% and of total P >94%, with concentrations of TP in the final effluents of <1.0 mg/l. The biggest reduction in BOD occurred in the aerobic bio-filter compared to other elements of the system, i.e. the septic tank and filter bed. During the first 6-12 months of operation, the removal of P showed a slow, but stable, increase, with substantial amounts of P being removed by the bio-filters. The majority of long-term P removal occurred in the filter beds due to the saturation of the bio-filters with P after the first year and the high P-sorption capacity of the LWA used in the filter beds. The estimated lifetime of the filter medium, for saturated filters in a 40 m^3 bed with P inlet values of about 10 mg/l, is about 15 years. Jenssen et al. (2010) argued that the incorporation of P-sorbing chemicals in the porous media within filter beds allows for stable high performance, in contrast to package treatment systems that require dosing of chemicals from containers in amounts that need to be adjusted to flow, water quality and pH. The investment costs of the seven systems tested was approximately 16,000 USD (£10,000) per household, which is comparable to the costs of other onsite options, such as PTPs or STs with soil adsorption systems in Scandinavia. It was also suggested that the LWA medium used in the study, once saturated with P, was suitable for use as an agricultural P fertiliser, with additional positive impacts caused by liming as a side effect (Jenssen et al. 2010).

Peat is a known natural attenuant in the environment, purifying water that passes through it (Kennedy and Geel 2000). It has been also shown to be an effective filter medium for P removal from wastewater (e.g. Xiong and Mahmood 2010, Kõiv et al. 2009). It is a cellular structured, complex material, with its major constituents (lignin, cellulose and humid acids) carrying polar functional groups. This gives peat a large adsorption potential for dissolved solids (Xiong and Mahmood 2010) and makes it favourable sorption medium over sand or gravel, which are characterised by lower adsorption capacity (Kennedy and Geel 2000).

Results of a laboratory experiment conducted by Xiong and Mahmood (2010) showed 94-99% and 76-95% removal of dissolved and total phosphate, respectively, from secondary effluent by a peat column. This indicates its potential as a substrate bed material for adsorptive removal of P from secondary wastewaters. P adsorption on peat decreased with increasing temperature and was the greatest at pH 6.5. At pH both above and below 6.5, peat adsorption showed a decreasing trend (Xiong and Mahmood 2010). The level of compaction of peat was indicated as another important parameter for efficiency of peat filter performance (Kennedy and Geel 2000). High level of compaction of peat will result in low saturated hydraulic conductivity, which in turn, increases the potential for clogging (Kennedy and Geel 2000). Peat filters are mostly design as vertical flow systems (Kõiv et al. 2009). Kõiv et al. 2009 investigated the treatment capacity of vertical and horizontal flow well-mineralised peat filters in terms of P

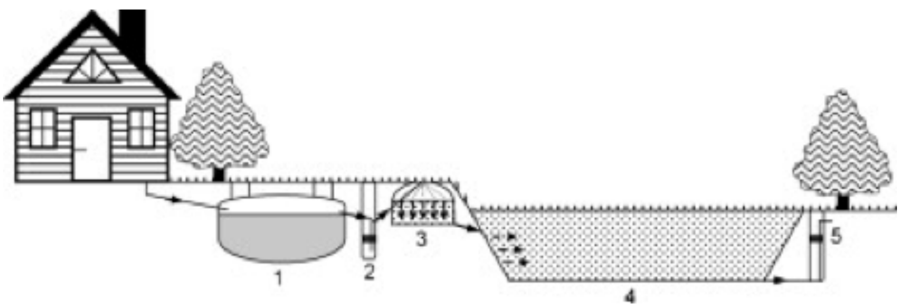


Figure 7 Nordic filter bed system
1. Septic tank;
2. Pump well;
3. Light-weight aggregate biofilter;
4. Subsurface horizontal flow filter bed;
5. Outlet
(after Jenssen et al. 2010).

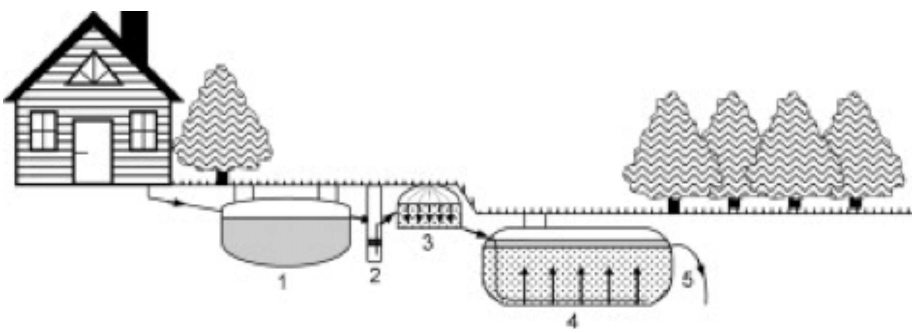


Figure 8 Norwegian compact filter system
1. Septic tank;
2. Pump well;
3. Light-weight aggregate biofilter;
4. Upflow saturated filter tank;
5. Outlet
(after Jenssen et al. 2010).

reduction from leachate and municipal wastewater in on-site conditions. The results of the two 6 month on-site experiments showed good P removal efficiency of the well-mineralised peat in vertical flow filters (81 mg /kg of P for dry peat) and poor efficiency in saturated horizontal flow filters. Patterson et al. (2001) suggested that addition of an approximately 600 mm deep bed of peat as a treatment mechanism for reducing the impact of septic tank effluent, made a significant difference to the quality of the leaving site effluent. Monitoring of five OWTSs in New South Wales, Australia, that were using Biogreen™ peat filter beds showed successful decrease of contaminants in the ST effluent, with faecal coliforms (FC), total nitrogen (TN) and TP being reduced by 99.5%, 44.2% and 83.6%, respectively. In these systems, effluents were distributed to peat treatment filter beds through a pressurised distribution network from a collection chamber connected to a ST, to maintain aerobic reduction conditions. The peat bed filters were dosed in small volumes many times during the day, depending on the rate of effluent passing from the ST to the collection tank. Peat beds were located adjacent to or on the top of drainfields. The effluent was carried away through slotted pipes from the bottom of the peat bed for distribution in soil (Patterson et al. 2001).

The Puraflo Peat Biofilter System from Bord na Móna is another example of an on-site system using peat media (Figure 9, Figure 10).

Effluent flows by gravity from a septic tank into a pump tank, then pumped in doses at specified intervals into Puraflo modules (treatment tank) where it is distributed onto peat media. Treated influent disperses from the bottom of the module into a gravel pad. The life expectancy of the Puraflo peat fibre, which is imported from Ireland, is 15 or more years. Table 11 presents parameter values of the treated effluent (Bord na Móna 2010).

Table 11 Parameter values for the ST effluent treated by Puraflo peat bed

Parameter	Concentration in treated effluent
CBOD (mg/l)	2
TSS (mg/l)	2
pH	6-7.5
Total N	>70% reduction
NH_3-N (mg/l)	<1
Faecal coliforms	99.9% removal

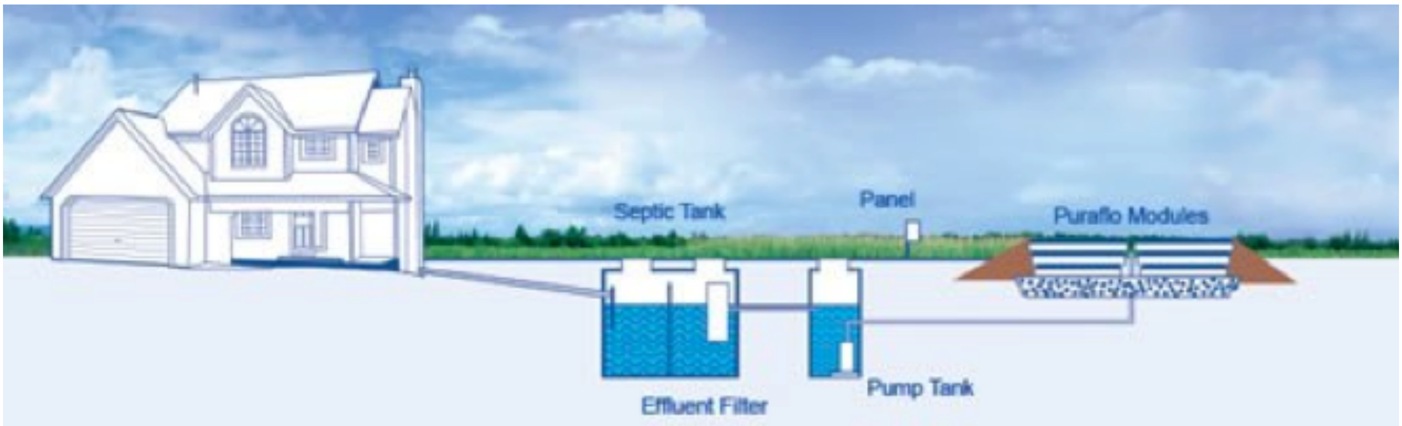


Figure 9 Puraflo peat biofilter system from Bord na Móna (Flemington Precast & Supply, L.L.C. 2014).

Figure 10 Puraflo peat biofilter system from Bord na Móna (Flemington Precast & Supply, L.L.C. 2014).

5.2 Measures for reducing pathogen discharges to the environment

Many OWTSs have been designed for the removal of solids, BOD, COD, and nitrogen, therefore efficiency of pathogen removal has been evaluated less widely. Where data on pathogen removal are not available, understanding the mechanisms for pathogen inactivation can assist in evaluating various treatment options. A full discussion of factors affecting pathogen survival is included in Annex 2.

The removal of pathogens from raw household effluent by a primary treatment system such as a ST can result in approximately a 1-2 $\log_{10}$ reduction in discharges of pathogens to the environment (see Table 7). This reduction may be due, in part, to predation by protozoa within the system, but is more likely to result from adsorption onto particles, flocculation and settlement in sludge. These processes can be influenced by characteristics of the effluent such as quantity and size of suspended particles, retention time within the system and chemical composition of the effluent.

In addition to processes occurring within a ST, the movement of pathogens from a ST into a secondary treatment system, or into the environment, can be affected by a number of physical, biological and chemical processes. Site characteristics, properties of infiltration surfaces, soil media, pathogen size, degree of saturation, presence of predators, temperature, pH, aeration and UV radiation among other factors can influence the level of pathogen removal that can be achieved (Stevik et al. 2004, Chabaud et al. 2006). Design of treatment systems for removal of pathogens from wastewater must therefore take account of a number of physical, chemical and biological elements as summarised in Table 12. Properties at each site location will vary, and some elements cannot be controlled such as local weather conditions. However, understanding the key principles of pathogen control can help identify suitable mitigation measures.

Table 12 Factors affecting pathogen survival and their potential use in mitigation activities

Factor	Considerations for pathogen reduction
Loading rate	Loading rate should allow adequate retention time for settlement of solids; misconnections of roof or surface drains increase hydraulic loading and reduce retention time
Seasonality	Treatment regime should take account of periods of high/low usage. Seasonal properties may not develop drainage fields with functioning biomats
Use of chemicals	Excessive dosing with disinfectants should be avoided to reduce potential deactivation of useful protozoa and reduced predation of bacteria
Settlement of particles/Flocculation	Finer particles enhance binding of pathogens; design and operation that reduces turbulence and increases retention time can increase pathogen settling in sludge
Site characteristics	Characteristics such as slope, percolation rate, soil type, type of bedrock, and proximity to watercourse can impact movement of wastewater and potential for contamination and should be considered in siting of systems
Filter media	Particle size can affect ability to bind pathogens; fine clay soils adsorb pathogens but can easily clog; too freely draining soils can reduce adsorption and filtering of smaller pathogens
Pathogen shape/size	Pathogen size can reduce in dry conditions; filter media could be chosen based on the pathogen of interest (i.e. bacteria or viruses)
Biomat formation	The biomat is important to pathogen retention. Favourable conditions for formation include unsaturated and aerated drainage field, and moderate and regular loading rate even dispersion
Aeration	Aerated systems can show improved performance due to enhanced biological removal processes
Moisture, rainfall	High rainfall events can increase the risk of flushes through the OWTS particularly where misconnections of roof or surface drains exist. This can increase treatment volume and influence the ionic strength of effluent, reducing adsorption
Temperature	Higher temperatures reduce pathogen survival. For treatment systems, insulation can reduce the impact of winter conditions/freezing temperatures on the functioning of drainage field or wetland
UV radiation	Increased opportunities for exposing effluent to UV radiation can enhance pathogen removal; lower turbidity reduces shading and sheltering of pathogens from exposure
pH	pH extremes can reduce pathogens; above pH 7 there is enhanced potential for ammonia toxicity
Exposure time	Increased retention time within a treatment system increases the potential for inactivation prior to discharge to the environment

5.2.1 Pre-tank measures for reducing pathogen discharges to the environment

Siting and setback distances

Proper siting of an OWTS is an essential first step in reducing potential pollution issues. It cannot be overstated that regardless of technological advances in onsite wastewater treatment, unsuitable locations for STs will increase the risk of release of pollution to the environment. Site conditions such as slope, soil type, distance to water table, distance to receiving water are all sensible measures to consider prior to installation of any OWTS. Scottish Building Standards provide some guidance on site design, with reference to BS 6297 (BSI 2008). This specifies that an appropriate distance for siting drainage fields to minimise the risk of pollution is 50 m away from a drinking water source and 10 m away, horizontally, from any water course, permeable drain, road or railway (Scottish Government 2013b). The Standards specify system design requirements and design of infiltration fields. While new systems should comply with these specifications, older systems may have been installed prior to these requirements being in place. For locations that are considering retrofitting measures to address poorly operating systems, a review of whether systems comply with existing standards would help site owners address any potential issues that may not have been accounted for previously.

Regulators should also consider whether a one-size-fits-all standard for locating OWTS is appropriate, especially in relation to the recommendation for siting distances of 10 m horizontally from any water course, permeable drain, road or railway. Risk based, site specific setback distances may be more appropriate. For example, the Cariboo Regional District in Canada has developed a shoreland management policy that sets recommended distances for OWTS from watercourses as a function of soil type, water body sensitivity and development density, with separation distances based on combined criteria scoring (Cariboo Regional District 2004). Pang et al. (2003) attempted to define an optimal setback distance for STs in New Zealand for the protection of groundwater and surface waters and found that a minimum setback distance of 16 m was sufficient to provide a 5-log reduction in bacterial concentrations. However, only a 48 m set back was sufficient to provide a 10 log reduction in viral concentrations, as required to comply with New Zealand drinking water standards. This example highlights how measures to reduce faecal bacteria contamination to acceptable levels may not be sufficient to reduce viral contamination. Within sensitive areas or high density areas, additional precautions may be needed in local planning regulations, such as minimum setback distances, vertical separation distances, the utilisation of reserve or alternating drainage fields, the use of alternative or innovative systems, or the use of shared systems or other management tools. In January 2015, new general binding rules for small sewage discharges came into effect in England that require new discharges to be more than 500 metres from a Special Area of Conservation (SAC), Special Protection Area (SPA), Ramsar site, biological Site of Special Scientific Interest (SSSI), aquatic local wildlife site, freshwater pearl mussel population, designated bathing water, or protected shellfish water and more than 50 metres from a chalk river (Defra 2014a). A similar requirement is not in place in Scotland.

In order to implement any changes to recommended set back distances, further research may be required to identify optimal setback distances for protection of sensitive water bodies and consultation with other public bodies such as the Scottish Government and planning authorities.

Risk based approaches

SEPA modelling predicts there are approximately 161,000 private sewage discharges in Scotland with about 61,819 currently registered (B. McCreadie, pers. comm. 28 Nov). Registration is on-going through the property conveyancing process. It is possible that some of the 100,000 unregistered sites will not be captured by the registration system for some time. Although the registration process has improved knowledge on the locations and types of systems in place, there is no evidence to suggest that registration has reduced pollution from OWTS. In 2014, Defra proposed changes to the registration process for STs in England. These took effect in January 2015 and include a requirement for registration of STs at sites located in the most sensitive areas where greater levels of environmental protection are required, such as bathing waters and shellfish waters (Defra and EA 2014). The changes automatically require all existing small sewage discharges in England to comply with general binding rules. This is a risk based approach to environmental permitting based on local conditions and evidence, attempting to focus the purpose of the registration process on the reduction of pollution in the most sensitive areas.

A number of authors propose using a risk based approach to identify areas likely to have either a high density of treatment systems, site characteristics that infer greater likelihood of system failure, or proximity to sensitive water bodies or drinking water supplies (Coffey et al. 2007, Kenway and Irving 2001, May et al, 2015). Swann (2001) recommended combining indicators such as ST density and predicted system age to identify problem areas within a catchment. May et al. (2015) combined distance from a water body, depth to high water table and slope to map zones within the catchment where STs would pose low, medium or high risks of contaminating freshwater SSSIs in England. Tools such as the On-site Sewage Risk Assessment System (OSRAS) for decentralised systems developed under the Australian Septic Safe programme in New South Wales can help identify potential problem sites for further investigation. The proposed changes by Defra recommend the use of proximity to protected areas as the criteria for registration.

Applying a risk based approach could be effective in focusing measures on the most sensitive or "at risk" locations. This could be based upon site designation, water body classification (at risk of not achieving good ecological status based on sewage related contaminants) or other criteria based on consultation with stakeholders.

Behaviour change and awareness

Ahmed et al. (2005) found well maintained systems are less likely to contaminate surface waters. Therefore, measures that inform and compel site owners and occupiers to inspect and maintain their OWTSs could, potentially, result in reduced incidence of system failures. Naughton and Hynds (2014) found that, while 92% of Irish residents surveyed were aware of the location of their OWTS, 42% did not know whether any secondary or tertiary treatment was in place, 71% had received no information about system operation or maintenance, and only 67% had previously had their system desludged. The study recommended that providing residents with information on how to carry out an effective inspection, general data on contamination of private water supplies, and information on the health and well-being benefits of effective ST management could be an effective approach to raising awareness amongst site owners. Naughton and Hynds (2014) also found that tenants were less aware than homeowners, and younger residents (who represent fewer homeowners) were less aware than older residents, suggesting that communication strategies should consider targeting awareness-raising at the young adults as this was the least aware group. The type of communication mechanism used (leaflets, social media or informative talks) and the emphasis of the message (health and environmental protection, monetary or EU compliance drivers) should be considered to make educational initiatives more effective. For example, the Dee Catchment Partnership has developed a UK Septic Tank Guide in a flyer format (Dee Catchment Partnership 2013) that lists practical information about STs including inspection tips and "Dos and Don'ts" of ST management for site owners and occupiers. This includes tips that could reduce pathogen releases, such as avoidance of misconnections of rainwater drainage pipes into a ST and the careful use of harmful chemicals that could disrupt microbial populations. The mechanism for distribution for information such as this should be considered by regulators. Targeted leafleting, information raising events in prioritised areas or electronic distribution could improve awareness and potentially improve management practices.

Communication and awareness-raising is a feasible and practical measure for helping site owners and tenants to identify potential issues with their OWTS and carry out appropriate maintenance. Registration has identified a large number of addresses where OWTSs are located, and the point of registration provides an opportunity for information to be provided to homeowners. Requesting contact email addresses during registration would enable paperless communication, and widespread, and repeat, awareness-raising at very little cost. The level of load reduction possible from awareness raising is difficult to quantify. However, any measures that improve OWTS management and maintenance are likely to help reduce pathogen loading to the environment.

5.2.2 In-tank measures for reducing pathogen discharges to the environment

The basic treatment process in STs is primary settlement, followed by gravity or pressure distribution of effluent into the environment, or secondary treatment system (Beal et al. 2006). Some removal of pathogens occurs within the ST, with estimates of 0-2 $\log_{10}$ reductions, or up to 99% possible (Feachem 1983). A range of factors can influence the quality of effluent released from a ST, hence some systems will be more effective than others at reducing pathogen releases as discussed in further detail in Annex 2. Kay et al. (2008) report that, under base flow conditions, conventional STs can provide only a reduction of 36% total coliforms, 57% faecal coliforms and 52% enterococci. In order to meet environmental quality standards, most ST effluents will require some level of additional treatment. The level of treatment required can be reduced if in-tank measures are implemented to maximise the potential for pathogen removal within the ST.

Tank design

The ability to remove solids within the ST is important in relation to reducing releases of pathogens within the effluent. Systems designed to maximise removal of solids will also increase the removal of pathogen from STE. Tank design can have a significant impact on the removal of solids by ensuring that the correct hydraulic retention time (HRT) is obtained, e.g. by reducing short-circuiting or channelling during high hydraulic loads (i.e. bathing, washing). Good inlet and outlet design can prevent short circuiting and can reduce turbulence and release of solids. The use of baffles within an upflow STS has been shown to reduce COD, suspended solids and BOD (Nguyen et al. 2007). The shape of the tank may also be important in preventing re-suspension of solids, particularly during tank desludging and wastewater inflow events. A flat bottomed tank will be more likely to cause re-suspension of particles compared to a parabolic shaped tank, particularly during desludging. A parabolic shape with baffles will help maximise the settling of solids.

Hydraulic retention time (HRT)

Increasing retention time in both a baffled or un-baffled system reduces COD and suspended solids in effluent (Nguyen et al. 2007). At times of peak loading or excessive water usage, a tank with insufficient capacity will not provide sufficient retention time for the system to work effectively. In addition, it is likely that peak flows can increase turbulence within a system more readily if capacity is reduced. This is likely to result in greater re-suspension of particles. The addition of a second chamber, or holding tank with an HRT of one to two days and the ability to trickle effluent into the main tank, can reduce the impact of turbulence and allow for a longer HRT in the main tank to increase settlement of pathogens within the sludge. Although misconnections of roof drains to STs are not likely to occur in newer systems, it is possible that these may be present in some older STs. Systems connected to roof drainage will experience flushes during high rainfall events, using up the hydraulic retention capacity of the system. This can result in the loss of solids and pathogens from the system.

One option for reducing hydraulic loading into a ST is the diversion of grey water from the system. Shower, bath, wash hand basin and laundry drains could be discharged to a separate treatment system, relieving the hydraulic burden upon the main ST. Hernandez Leal et al. (2011) carried out an extensive chemical characterisation of household grey water and found that, despite grey water being assumed to be "clean", it can account for up to 50% of the household COD and also contains a high level of surfactants. Average concentrations of TP were found to be 7.2 mg/l and of phosphate-P were found to average 2.36 mg/l. A study carried out on the impact of grey water diversion on ST function in New Zealand found that grey water diversion reduced suspended solids in a poorly functioning tank, and reduced the loading of solids to a soakaway by between 13% and 56% for a well-functioning and a poorly-functioning system, respectively (Siggins et al. 2013). A significant reduction in both TP and *E. coli* loading was observed when grey water was diverted from the poorly functioning site, with TP loading falling from 6.5 g/day to 4.6 g/day and *E. coli* falling from 3.1 x 10^9 MPN[1]/day to 5.6 x 10^8 MPN/day. Although a reduction in both TP

[1] Most probable number (MPN) – unit of measure for bacteria

and *E. coli* was observed at the well-functioning tank, this was not found to be significant. Overall the study concluded that grey water diversion had little impact on the reduction of *E. coli* concentration in STE, but the reduced volume resulted in reduced loading to the soakaway, which could, potentially, extend the life span of the post tank treatment area. These results suggest that grey water diversion may be a useful measure at sites with poorly functioning OWTSs, however there is no evidence that grey water diversion significantly improved effluent quality from a well-functioning site. Levels of E.coli in the grey water stream were approximately two orders of magnitude lower than in the STE. The results also suggest that pathogen levels in diverted grey water remain high, and as such would require additional treatment to reduce these to acceptable levels.

As discussed in 5.1.2, HRT can be increased by maintaining an adequate treatment capacity through regular desludging. Over time, pathogens held within a ST will become deactivated. The ability of solids to be retained in the tank reduces the opportunity for pathogens to be transported out of a tank attached to solid particles. A reduction in HRT due to sludge build up can reduce the length of time a pathogen may be held within the tank, and the ability of the tank to retain additional solids. There may be some conflict between the optimal desludging frequency for pathogen reduction and phosphorus reduction, with some suggestion that frequent desludging could increase the release of soluble P (Canter and Knox, 1985). Unlike pathogens, that are deactivated over time, P held within a tank is retained within the sludge, and can change between soluble and insoluble forms over time. Further research may be required to determine the optimal desludging frequency that balances the retention of both pathogen and P releases to the environment.

Addressing drain misconnections is a low cost and practical way of reducing pollutant flushes from STs. Limited evidence on the diversion of grey water from STs suggests that it could be a useful measure for reducing loading to the environment, particularly for poorly functioning systems. Grey water contains high levels of pollutants and may still require additional treatment. Although diversion of grey water would be easier, and potentially cheaper, in new build properties as opposed to retrofitting to existing homes, this measure may be less effective in new builds with well- functioning STs or OWTSs. Desludging to maximise HRT and reduce the release of solids in effluent will reduce pathogen releases to the environment. Further research may be required to determine the potential impact on P releases.

Addition of treatment agents

Gaikwad and Gupta (2010) experimented with the use of biological control agents to reduce wastewater pathogens. They found that the use of *lactobacillus* spp. inhibited the activity of pathogens by either the production of lactic or acetic acid or via the production of substances such as bacteriocins, proteinaceous toxins that inhibit bacteria. The study found that *lactobacillis* may be useful in the inactivation of *E. coli*. Off the shelf ST additives are available that claim to improve tank function. However, there is currently inadequate information on the efficiency of these additives in terms of pathogen reduction. Other types of treatment agents that could be added to tanks include those that enhance particle settling or precipitation, thus reducing transport of solids and pathogens. Any additional precipitation within the tank results in a greater production of sludge, increasing the need for tank desludging. Addition of lime or similar additives to the system requires a dosing mechanism, and additional expense for the treatment materials. The resulting effluent may also have an increased pH, which may enhance pathogen die off through ammonia toxicity, but may require additional treatment to adjust the pH to acceptable levels for discharge.

There is a lack of evidence that in-tank treatment agents are effective in reducing the release of pathogens. Considerations for use include the additional cost of treatment materials, dosing systems and the increase in the frequency of desludging required.

5.2.3 Post-tank measures for reducing pathogen discharges to the environment

The majority of pathogen removal occurs after effluent has left the ST. In general, post tank treatment options for STE can vary from simple to complex and include a range of potential treatment processes such as adsorption, predation, biotransformation, oxidation or filtration (Beal et al. 2006).

Soakaway/drainage field

British Standard 6297:2007 distinguishes traditional soakaways from drainage fields, referring to a traditional soakaway as an area of ground backfilled with bricks or granular material for assisting the drainage of clean surface water into the ground (BSI 2008). BS 6297 suggests that this type of soakaway does not provide adequate treatment for use in wastewater effluent disposal (BSI 2008). Commonly, the term “soakaway” is used interchangeably with terms such as drainage field, infiltration area, percolation field, or soil absorption system. In contrast to a traditional soakaway, the latter types of system refer to a network of infiltration pipes arranged in trenches to allow even distribution of effluent for infiltration into the ground (BSI 2008).

There are a number of factors that influence the effectiveness of the drainage field (see Annex 2: Factors affecting pathogen loading and survival). Downward flow of materials through a drainage field is generally gravity driven and allows the movement of solids, microorganisms, and organic matter through the subsoil. Even distribution of effluent in a percolation area can affect the effectiveness of biomat formation and, hence, the functioning of the drainage field and its ability to absorb pollutants. Unsaturated drainage fields with suitably small particle sized media can be effective in removing additional pathogens from STE. In the region of 90% faecal coliforms and 85% of *E. coli* can be removed in the drainage field (Tomaras et al. 2009), although these figures can vary from site to site. Mounded systems can be used to increase the separation distance between a STE outlet and a receiving water body by increasing the height of the drainage field above ground (see *“Installing a soil based soakaway”*). Mounded systems are, in simple terms, an above ground drainage field constructed using layers of sand, gravel and soil, and possibly covered with geotextile material to encourage the growth of grass to cover the mound. They are used in areas where water tables are high. A pump may be required to move effluent to the top of the mound. The function of a mounded system will not differ significantly from a conventional drainage field.

Drainage fields, soakaways and mounded systems provide a basic level of treatment via filtration and absorption.

Lagoon/waste stabilisation pond

The aerobic conditions and shallow bed of a lagoon or waste stabilisation pond (WSP) can assist in providing additional polishing treatment for wastewater by increasing exposure to predation and sunlight. The additional retention time of effluent enhances the settling of solids, and the potential for natural pathogen die off. The reduced level of solids within the effluent can also help in making UV inactivation more effective in surface water, and any chemical dosing to improve effluent quality will be more effective after effluent leaves the ST and before it reaches the settlement lagoon. Chemical dosing, such as lime dosing or chlorination, can also be applied to effluent within a WSP relatively easily to provide additional polishing.

Waste stabilisation ponds are very effective in warm, sunny climates where removal rates of 3-6 log units of bacteria are possible. In cooler climates, a longer retention time may be required to maximise removal rates (Jimenez et al. 2010). The use of additional lagoons or in-stream reservoirs, where sediments are protected from high storm flow or tidal action that would cause a re-suspension of particles and hence pathogens, could also be considered. Gannon et al. (2005) found that lower flow velocities and higher residence times within reservoirs increased faecal coliform (*E. coli* and *Streptococcus*) sedimentation and die off, and improved downstream water quality. Retention systems require impermeable soils and sufficient storage capacity to allow settlement of pollutants and to provide protection from wave or flow action to reduce turbulence. Caution should be used where in-stream reservoirs are placed in areas of recreational use because, as noted by Gannon et al. (2005), die off rates of bacterial contaminants in sediments are much lower than in water and can present a risk to human health. Care should therefore be taken to minimise public access to in-stream reservoirs.

Lagoons or WSPs may be effective as a polishing measure, increasing retention time of effluent before release to the environment. These systems are only practical where space is available and where their presence does not present increased risk of human exposure to pathogens.

Constructed wetlands (CWs)

Constructed wetlands comprise a bed of soil, sand or gravel populated with aquatic or wetland plants, receiving effluent directly from a ST or secondary treatment system. Constructed wetlands can be either surface flow (SF), subsurface horizontal flow (HF) or vertical flow (VF) (Jimenez et al. 2010). The process of pathogen removal in constructed wetlands involves a combination of predation, filtration, adsorption and UV inactivation (Jimenez et al. 2010). The effectiveness of using CW for pathogen removal can be site specific. Jimenez (2003) reviewed a number of studies that demonstrated levels of faecal coliform removal of 90-99% in wetland systems, but efficiency was related to climatic conditions. Mbuligwe (2005) found that, compared to standard ST treatment only, an engineered wetland system improved FC removal from 40.17% to 99.99% and TC removal from 37.4% to 99.99%. Chang et al. (2014) compared pathogen and nutrient removal in subsurface flow wetland systems and observed removal of FC, *E. coli* and TP. The removal rates were 97.06 99.98% for FC, 99.80-100.00% for *E. coli* and 95.7-98.3% for TP (Table 13). The study by Chang et al. (2014) also showed that the presence of plants did not significantly affect the removal rate of pathogens in comparison to a control wetland system without plants. The control system, however, did show a reduced removal of TP compared to the planted systems. This suggests that the removal mechanism for pathogens is much more reliant upon physical removal processes in the wetland system as compared to the presence of plants.

Table 13 Pathogen and total phosphorus removal in subsurface flow wetland systems (Chang et al. 2014)

Wetland Cell	FC removal (%)	*E.coli* removal (%)	TP removal (%)
1.	99.98	100.00	98.3
2.	97.06	99.94	95.7
3.	99.76	99.80	98.0
Control (un-planted)	99.74	100.00	85.7

O'Luanaigh et al. (2009) also found reductions in the region of 99.5% (2-3 $\log_{10}$) of TC and 99% (1.9 $\log_{10}$) FC in reed bed constructed wetlands. Nguyen et al. (2007) found a reduction in FC in a two-step vertical flow CW of 97% (1.3-1.4 $\log_{10}$ units). However, ponding caused anaerobic conditions to develop, which reduced removal efficiencies in this study. Solids were reduced significantly in this system from 189.6 to 6.8 mg/l and TP was reduced from 5.6 mg/l to 1.6 mg/l. However, the 97% reduction in FC was insufficient to meet local environmental standards. In addition, the system demonstrated reduced efficiency after two years, indicating a need for biannual desludging.

Constructed wetlands can include modifications that can increase aerobic biological removal processes. These modifications can include introduction of direct aeration through the use of air diffusers incorporated in the system, or indirect aeration through the use of drops, baffles, or step feeding within the system (Wu et al. 2014). Similarly, recirculation using pumps or novel system designs can also enhance removal. These modifications have an additional cost and maintenance implication that should be considered alongside additional treatment capacity (Wu et al. 2014).

Constructed wetlands can be effective at reducing pathogens in STE. The practicality and feasibility of utilising CW systems relates to the level of additional treatment required (secondary or tertiary) and the availability of space in which to locate them. Where space is available, CW systems provide additional pathogen removal and should, therefore, be considered where existing systems require enhancement. Constructed wetlands require regular maintenance to remove weeds, to keep inlets and outlets clear, to harvest or trim wetland plants, and to remove dead vegetation to ensure that the system is operating optimally (Mbuligwe 2005). For systems where filtration media are used within the wetland, there may be a requirement for renewal of the substrate over time. There are costs associated with this process that need to be considered.

Filter systems

Sand Filter: Harrison et al. (2000) found that installation of a sand filter increased pathogen removal in the drainage field from 98.6% in a soil-only system to 99.8% with a sand filter. The study showed that less than 0.1% of faecal coliforms (FC) passed through the sand filter as compared to 9.1% through the soil only system. Pundsack et al. (2001) found that a sand filter was effective in removing up to 4 $\log_{10}$ units of FC from STE. However, the removal rate varied seasonally (Table 14). Gill et al. (2009b) found that most treatment occurred within the first 1.1 m and that the optimal loading rate for bacterial removal was 30 l /m per day, and for TP removal was 60 l /m per day, and that the treatment efficiency of a stratified sand filter was similar to that of a percolation trench field. However, the latter required a much smaller area for installation. The efficiency of a sand filter can decrease over time as adsorption sites are used up, so filter media may need to be replaced occasionally.

Peat filter: Peat filters, such as the Puraflo® secondary treatment system, have been trialled in studies by the Environmental Protection Agency (EPA) in the Republic of Ireland (Gill et al. 2001). This system showed a significant reduction in bacterial load along with organic load into the percolation area. Pundsack et al. (2001) also showed almost complete removal of FC using peat filters, with seasonality having less of an impact on efficiency than when using sand filters (Table 14). Patterson (1999) showed a reduction of FC to less than 99.6% of the influent concentration and Gill et al. (2007) reported a 3-4 $\log_{10}$ reduction of enteric bacteria across a peat filter treatment system. While there is evidence that peat filters are effective at reducing bacterial loads, the efficiency of the system can reduce over time as shown in Figure 11. This suggests that the system may need to be renewed as treatment efficiency begins to drop below acceptable levels.

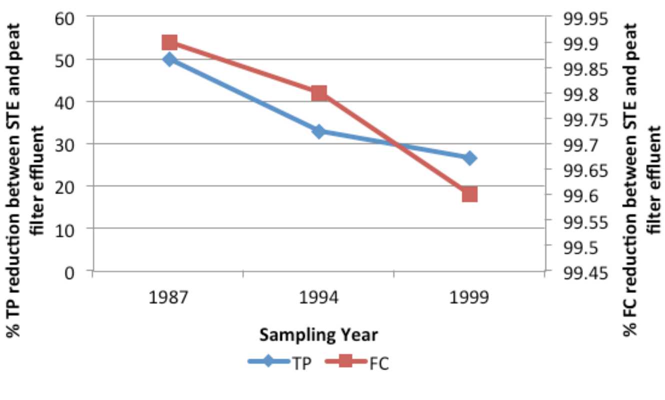


Figure 11 Removal of TP and FC from septic tank effluent by a peat filter between 1987 and 1999 (Patterson 1999).

Pundsack et al. (2001) compared the pathogen removal efficiency of three secondary treatment systems: a constructed wetland, a sand filter and a peat filter. The results of this experiment are presented in Table 14.

Although all of these systems demonstrated a drop in treatment efficiency in the winter, the peat filter was able to maintain higher temperatures than the other systems and hence greater removal efficiency. Pundsack et al. (2001) determined that CW were the least efficient treatment systems throughout the year, with a number of observations suggesting why this was the case. The medium size used in the sand filter and peat filter had a smaller particle size, thereby providing additional filtering capacity and adsorption sites for both pathogens and nutrients. In addition, the vertical flow of the sand filter and peat filter allowed for improved aeration of the effluent compared to the CW, reducing the likelihood of saturation in the system.

Other types of filter: A range of other filter configurations and types are available on the market, ranging from alternative materials for use in filter beds to ultra- and micro-filtration systems. Kadam et al. (2008) investigated the efficiency of a constructed soil filter. The system differed from a simple sand filter by the incorporation of media, native microflora, geophagus worms, and bio-indicator plants. The system was described to remove pathogens by filtration, creation of an unsuitable physico-chemical environment, and predation. The study showed an approximate 3 $\log_{10}$ reduction in FC and TC, but the resultant effluent was still at a level that exceeded acceptable standards (1.5-3.6 x 10^5 cfu /100 ml for TC and 3.1-8.3 x 10^4 cfu /100 ml for FC). *E. coli* was reduced from 1.2-3.3 x 10^6 cfu /100 ml to 1.5-2.5 x 10^4 cfu /100 ml. A range of alternative sorption media, such as ochre, slag, zeolites, lime, gypsum and fly-ash have been assessed for use in wastewater treatment systems. These materials are generally tested in relation to nutrient removal, with impacts on pathogens less well studied. However, Chang et al. (2014) investigated the use of a mixed sorption medium to augment soil media, using a combination of 50% citrus grove sand, 15% tyre crumb, 15% sawdust and 20% limestone within a subsurface upflow wetland system. This system reduced *E. coli* concentrations by between 99.99670% and 99.99987%. The study indicates that alternative filter media may be effective in reducing pathogen concentrations, and that filter media used in combination with CW may provide more complete pathogen removal in comparison with traditional CW systems.

Jenssen et al. (2010) compared the efficiency of nine different pilot plants located in Nordic countries (Figure 7) that had been designed to maximise P removal and provide a potential

Table 14 Comparison of innovative secondary treatment systems (Pundsack et al. 2001)

System Type	Design and Features	$\log_{10}$ decrease in FC	Treated effluent FC concentration (cfu /100 ml)
ALL SYSTEMS	Designed to treat single family home effluent to secondary treatment standards Septic tank effluent strength was 2.9 x 10^5 cfu /100 ml faecal coliforms; 15 mg/l TP; 96 mg/l TN; 294 mg/l BOD		
Subsurface flow constructed wetland	Two cell system planted with cattails and bulrush	2.8 - 2.9 (summer) 1.6 - 1.7 (winter)	4.0-5.0 x 10^2 (summer) 5.9-7.1 x 10^3 (winter)
Intermittent sand filter	Depth of 1.2 m and 3 m	3.9 - 4.1 (summer) 3.3 - 3.4 (winter)	3.5-6.0 x 10^1 (summer) 1.1-2 x 10^2 (winter)
Intermittent peat filter	Depth of 1.4 m	4.5 (summer) 4.8 - 4.9 (winter)	6 (summer) 4-5 (winter)

route for P reclamation. These systems were also assessed in terms of pathogen removal. Each system was designed with a ST, a pump well, a vertical flow single pass aerobic biofilter, a subsurface horizontal flow filter with a proprietary product (Filtralite P®), and an outlet well. The biofilter consisted of 0.6 m of lightweight aggregate (LWA) within a fibreglass tank and a distribution system fed by a high pressure pump. All systems were extremely efficient in reducing FIO to very low or non-detectable levels and, apart from at one test site, removed pathogens to a level that fell within bathing water quality limits (<500 *E. coli* /100 ml; <200 enterococci /100 ml). No viruses were detected in the Norwegian compact filter system (Figure 8) with a high initial pH of the LWA (pH 12-13). Even after three years of treatment, the pH remained high at 9-10. The high pH may contribute to pathogen inactivation, but may also influence the suitability of this system for use in areas where discharges with a high pH could be detrimental to the receiving environment.

In addition to alternative media use in filters, ultra and microfiltration systems are also available, which can be added onto treatment systems. Membrane bioreactors contain very small pore sizes that offer ultra and microfiltration. They are therefore effective at removing almost all pathogens including viruses but can be complex to operate, expensive and prone to fouling, requiring regular maintenance.

Filter systems can be effective at removing pathogens from STE. The level of treatment can be related to filter media, with peat filters showing greater treatment capacity for pathogens in comparison to sand filters. Alternative filter media have shown promising results and may be more practical than CWs due to their smaller land requirement. Combination systems using alternative filter media and CW together appear to be very effective at reducing pathogen concentrations to acceptable levels. These systems may have high installation and ongoing maintenance costs, including filter media renewal over time.

Disinfection

The addition of precipitating chemicals (e.g. lime dosing, ferric chloride addition) has been shown to be effective at reducing pathogens when used for tertiary treatment (Nieuwstad et al. 1988, Koivunen et al. 2003). Use in an onsite treatment capacity has been less widely assessed. Chlorination has been shown to be effective at reducing coliform bacteria, but less effective at reducing viruses. Used within a tank system, chlorination may adversely affect useful microorganisms that are involved in the treatment process, hence it is better suited to use as a tertiary treatment if required. Ozone treatment may also be used to kill or inactivate *E. coli* (Coffey et al. 2007) as can UV treatment. There are additional costs associated with additional treatment steps such as chlorination, ozonation and UV treatment that must be considered, as well as additional upkeep and maintenance of treatment systems.

Disinfection may be undertaken as a polishing step where the need for additional pathogen reduction cannot be achieved by other means.

5.2.4 Pre-fabricated package treatment plants

In locations where conventional systems are not practical due to poor soil conditions or lack of space for CW or filter systems, a package treatment plant (PTP) may be the only reasonable option for improving on-site sewage treatment. Scottish Building Standards define PTPs as "*systems engineered to treat a given hydraulic and organic load using prefabricated components that can be installed with minimal site work*" (Scottish Government 2013b). The most commonly available PTPs in the UK tend to be designed to maximise the physical and biological processes that reduce BOD, COD and TSS. There is little available data on their pathogen removal capacity. General principles of solids removal suggest that they will reduce pathogen concentrations, but further research is required to identify the level of reduction possible. A description of the most common systems is provided below.

Balmoral systems

System	Description
Two chamber septic tank	System designed specifically to reduce the suspended solids in tank discharges. Can serve single dwellings of 1-4 pe or larger or multiple premises of up to 22 pe
Continuous aeration plant (CAP)	Two chamber system that uses aeration to assist biological treatment processes. Outer chamber allows primary settlement of solids to occur. Central chamber contains a small compressor that supplies air to a bubble diffuser to assist bacteria that provide secondary treatment. Systems range in size from single family to 12 pe capacity
Hydroclear™	System that utilises a moving bed biological reactor (MBBR). Manufacturers claim that this overcomes the issue of short circuiting that is common to process designs such as submerged aerated filters, RBC or trickling filter designs. The system has three chambers. 1. Settlement – desludging area 2. Aeration – biological reactor, with induced area and biomedia where dissolved constituents are removed 3. Final settlement Manufacturer suggests that benefits include self-cleaning, and resistance to blocking, and 97% removal of pollutants with a final BOD:SS:NH4-N at 10:13:6 mg/l. System suitable for single family dwellings or larger units of up to 50 pe
Sequential Batch Reactor (SBR)	Finer particles enhance binding of pathogens; design and operation that reduces turbulence and increases retention time can increase pathogen settling in sludge

Viltra systems

System	Description
Viltra IWOX Premium system	Two tank, three compartment system. Initial tank is a pre-settlement tank. Second tank is aerated and microorganism rich. As sludge settles, wastewater becomes clarified for release. Settled sludge is recirculated back into the biological compartment. Manufacturers claim a BOD:SS performance standard of 10:10 and possibly 5:5
Vitra IWOX Clear system	System carries out a similar process to the Premium system, but also includes a denitrification zone and a second clarifier. System reduces the BOD:SS to 5:5 (Edens et al. 2010) and has a smaller footprint (0.8 m2) than a reed bed (8 m^2)

Klargester systems

System	Description
BioDisc® Rotating biological contactor (RBC)	System uses a combination of settlement and biological processes to reduce BOD and solids in effluent. Rotating biological contactor (RBC) used to enhance an active microbial population to increase biological breakdown of organic compounds. Systems can be used for single family or small communal units
BioTec ™	System uses a combination of settlement and biological processes to reduced BOD and solids in effluent. System filters coarse solids, redistributes filtered liquid through a suspended filter, and includes a final settlement stage

O'Luanaigh et al. (2012) assessed the effectiveness of the Klargester Biodisc® RBC in comparison to treatment capacity of a ST plus reed bed system and a ST-only system. The results indicated that the RBC system reduced *E. coli* and TC better than the ST-only system. Compared to the reed bed system, the RBC showed a similar reduction in pathogen numbers (slightly better for E. coli, slightly worse for TC). *E coli* and TC were both reduced by 1-2 $\log_{10}$ units compared to ST only treatment. A review carried out by Kay et al. (2008) showed that a RBC used as a secondary treatment system provided a 98.26% reduction in TC, 99.06% reduction in FC and 99.5% reduction in enterococci compared to untreated effluent under base flow conditions. The quality of the effluent was still in excess of 1 x 105 cfu /100 ml for TC, FC and approaching 1 x 104 cfu /100 ml for enterococci.

The pathogen removal capacity of many PTPs is not widely available, with published performance results primarily stating reductions in nutrients, BOD and suspended solids. Akunna and Jefferies (2000) compared the effectiveness of SBR and RBC package plants and found that the SBR system produced effluent with TP of 2 mg/l, BOD <10 mg/l, and suspended solids <15 mg/l. The RBC system produced similar levels of BOD and solids removal, however, the level of nutrient removal was much reduced. The SBR system was observed to tolerate shock loads of flow better than the RBC system. It is expected that the reduction of solids by both systems would have resulted in a reduction of pathogen discharges, although, this was not quantified in the study.

There is a lack of evidence that PTPs provide significant pathogen removal capacity and further study is required to quantify the impact of these systems on pathogen removal. Limited evidence suggests that these systems can reduce suspended solid loading and, as such, should help reduce pathogen levels more than a traditional ST.

5.3 Estimated load reductions of practical measures

5.3.1 P-reducing measures

The following measures have been proven to be successful in reducing P outputs to the environment from OWTSs and are likely to be the most feasible and practical P-reducing measures.

Pre tank measures:

- Reducing P inputs to OWTSs, e.g. by using P free detergents (Alhajjar et al. 1990) or avoiding flushing food waste down the sink or toilet.

In tank measures:

- Augmenting old systems with a modern ST (Ockenden et al. 2014)
- Using chemical additives (Azam and Finneran 2014)
- Replacing old systems with modern systems (Macintosh et al. 2011)

Post tank measures:

- Combining existing systems with tertiary treatment such as constructed wetlands/reed beds (Ockenden et al. 2014)
- Incorporating subsurface flow constructed wetlands with pre-treatment biofilters (Jenssen et al. 2005)
- Installing on-site filter systems (Nilsson et al. 2013a; Nilsson et al. 2013b)

The level of load reduction possible from some measures is difficult to quantify, particularly pre-tank measures. Details on efficiency and load reductions possible for various filter media were presented in Table 10 and the level of performance of the above measures based on literature, where available, is summarised in Table 15.

5.3.2 Pathogen reducing measures

The effective reduction of faecal pathogen loading resulting from pre-tank measures is difficult to quantify, and the impact of in-tank measures can be very site specific. Table 16 summarises the mean bacterial concentrations for treated effluent or percent pathogen removal for various treatment systems. This provides an indication of the relative load reductions possible for the various measures as identified in the literature.

Table 15 Performance measures available for reducing P discharges from OWTSs

Measure		% TP reduction in wastewaters	Reference
Ban on detergents containing more than 0.5% of P		40-50	US EPA 2002
Chemical additives		85	Brandes 1977
Constructed wetlands	Initial results	20	Duenas et al. 2007
	After 10 years	10	
Constructed wetlands (during the first year)		60	Ockenden et al. 2014
Microbial electrolysis cell (MEC)		39	Zamalloa et al. 2013
Three-chambered septic tank		33	Nasr and Mikhaeil 2013
Klargester package treatment plant (PTP)		47.6	Kingspan Environmental 2010
Soil filter beds (aged between 14 – 22 years)		12	Eveborn et al. 2012
Sand filter bed		almost 100	Robertson 2012
Peat filters		84	Patterson 2001
Filter bed systems (with a biofilter and aggregate (LWA) Filtralite®P)		94	Jenssen et al. 2010

Table 16 Mean bacterial concentrations within wastewater effluent after treatment (cfu /100 ml) Where pathogen concentrations were not reported, removal rate is indicated in italics

Treatment type	Total coliforms	Faecal coliforms	*E. coli*	Enterococci	Reference
Settled septic tank	2.5×10^7	7.2×10	-	9.3×10^5	Kay et al. 2008
Soil filter plus alternative media	2.55×10^5	5.7×10^4	2×10^4	-	Kadam et al. 2008
Trickling/sand filter	6.4×10^5	2.1×10^5	-	2.1×10^4	Kay et al. 2008
Sand filter	-	(99.8% *removal*)	-	-	Harrison et al. (2000)
Intermittent sand filter	-	4.75×10^1 (summer) 1.55×10^2 (winter)	-	-	Pundsack et al. (2001)
Intermittent peat filter	-	6 (summer) 4.5 (winter)	-	-	Pundsack et al. (2001)
Rotating biological contactor (RBC)	6.8×10^5	1.6×10^5	-	9.6×10^3	Kay et al. 2008
Reed bed/grass plot	3.7×10^4	1.3×10^4	-	1.9×10^3	Kay et al. 2008
Reed bed CW	99.5% *removal*	-	-	-	O'Luanaigh et al. 2009
SSF CW	-	99.135% *removal*	99.935% *removal*	-	Chang et al. 2014
Two step VF CW	-	97% *removal*	-	-	Nguyen et al. 2007
SSF CW plus alternative filter media (citrus grove sand, tyre crumb, sawdust and limestone)	-	-	99.998% *removal*	-	Chang et al. 2014
Combination filter system VF and HF and alternative media	-	-	< 500	< 200	Jenssen et al. 2010
UV disinfection	1.5×10^3	2.8×10^2	-	8.3×10^1	Kay et al. 2008

Note: Data from Kay et al. (2008) are under baseflow conditions

These data suggest that, for many of the treatment systems, additional measures would be required to treat effluent to acceptable standards. Peat filters, sand filters, constructed wetlands and other filter media systems, and UV disinfection, all have the potential to reduce pathogens to acceptable concentrations. However, variability in the data suggests that treatment efficiency may be site and system specific, and may also show seasonal fluctuations. Systems that are effective at removing solids (filters) are also effective in reducing pathogen releases. Additional factors, such as slowing the movement of effluent to the receiving environment, can enhance the adsorption processes and increase the levels of predation and UV inactivation in order to reduce pathogen loading.

Table 17 summarises the potential pathogen reduction measures and their potential impact on pathogen loading to the environment. The load reductions possible using these measures may be very site specific and, hence, a full site assessment would be required to verify suitability of any measure to a particular site or system. Many of these measures could be used in combination to achieve a greater level of effectiveness. Practical considerations such as cost, availability of land, and site characteristics may exclude some measures being utilised.

Table 17 Pathogen load reduction measures

Measure	Type of treatment	Potential load reductions	Feasibility/practicality
Pre-Tank Measure			
Appropriate site and setback distances	Increased separation distance from receiving water should enhance removal efficiency	Site specific	Further research is required to identify optimal setback distances for protection of sensitive water bodies; consultation with Scottish Government, planning authorities etc. would be required
Risk based measures	Identification of high risk systems	Yes – site, load rate and design specific	Risk based approach could be effective in focusing measures on the most sensitive or "at risk" locations; further consultation and changes to regulation may be required. This approach is currently being implemented in England
Awareness raising	Increased identification and remediation of poorly performing systems	Yes – site, load and design specific	Feasible and practical measure for helping site owners to identify potential issues with OWTS and carry out appropriate maintenance. Difficult to quantify level of load reduction possible; any measures that improve OWTS management and maintenance practices likely reduce pathogen loading to the environment
In-Tank Measure			
Tank design (baffles and shape)	Increased retention of solids, buffering from shock loading and turbulence	Yes – site, load and design specific	Includes consideration of tank design for new build properties in 'at risk' locations and for tank replacement. Could be included in awareness raising activities
Increased HRT – correcting misconnections	Increase in solids retention and decrease in solids re-suspension		Feasible, practical and low cost measure that is likely to reduce pollutant flushes from STs on existing sites. Requires communication with site owners and/or inspection of problem sites
Increased HRT – desludging	Increased retention of solids, buffering from shock loading and turbulence	Yes – site, load and design specific	Desludging at recommended frequencies should help avoid excessive pathogen flushes from treatment systems; determining the optimal desludging frequency may require research, especially where unintended impacts on P releases could result
Increased HRT – grey water separation	Increased retention of solids, buffering from shock loading and turbulence	Yes – site, load and design specific, potentially more effective measure for retrofit than for new build	Could be a feasible measure to reduce pathogen discharges, particularly for poorly functioning systems. Practicality relates to cost of retrofitting, providing separate treatment of grey water, and need for additional space for onsite for grey water treatment
Post-Tank Measure			
Soak away/drainage field/ percolation field	Increased retention and settling; retention of solids and pathogens, increased predation and retention in the biomat	Minimal beyond ST treatment. Approximately 90% FC, 85% *E. coli*	Drainage fields, soakaways and mound systems provide a basic level of treatment through filtration and absorption. Most practical for 'at risk' sites with direct discharges to water bodies; less effective for seasonal use properties due to potential failure of biomat
Mound system	Similar to drainage field	Similar to drainage field	Similar to drainage field
Lagoons/WSP	Solid settling, predation and UV exposure; chemical treatment can inhibit pathogens by precipitation or disinfection	Yes, dependent on UV exposure and HRT; chemical treatments applied to WSP can provide final polishing	Systems only practical where space available and where their presence does not increase risk of human exposure to pathogens
Constructed wetland	Reduction of solids, filtering of pathogens, predation, UV inactivation	Up to 99.99% FC, TC and up to 100% *E. coli*	Practicality and feasibility of using CW systems relates to level of additional treatment required (secondary or tertiary) and availability of space required. CW require regular maintenance to remove weeds, keep inlets and outlets clear, harvest or trim wetland plants, remove dead vegetation, and renew saturated substrate; costs associated with this process need to be considered

Measure	Type of treatment	Potential load reductions	Feasibility/practicality
Sand filter	Reduction of solids, filtering of pathogens	Over 99% pathogen removal, seasonally variable	Level of treatment depends on filter media; peat filters showing greater treatment capacity for pathogens than sand filters; filter systems may be more practical than CWs due to a reduced land requirement; combination systems using alternative filter media and CW appear to be very effective at reducing pathogen concentrations. May have high installation costs and on-going maintenance costs, including filter media renewal. For alternative filter media (e.g. waste products) potential release of other pollutants or influence on effluent quality (i.e. pH) should be considered
Peat filter	Reduction of solids, filtering of pathogens	Over 99% pathogen removal, seasonally less variable	
Ultra and micro filtration	Filtration with reduced pore size	Yes complete but expensive and prone to clogging	
Alternative filter media	Retention of solids, and filtering of pathogens	Range of load reduction dependent on material	
Combination systems	Removal of solids, pathogen filtering, UV inactivation, aeration	Potential to achieve high bacterial pathogen removal	
Package treatment plants			
SBR (e.g. Balmoral)	Increased biological treatment, solids reduction	Variable and design dependent; likely requirement for polishing	Lack of evidence that PTPs provide significant pathogen removal capacity; further study required to quantify impact on pathogen removal. Some evidence that these systems provide a better level of treatment than traditional STs

5.4 Comparison of system costs

Cost data are not available for all of the systems reviewed. Jimenez et al. (2010) reviewed a number of mechanisms for pathogen removal and identified waste settling ponds and aerated lagoons as the least cost options. Ultra- and micro-filtration offered the most effective removal of all pathogens, but these systems are complex to operate, may be expensive and require a higher level of maintenance and inspection than other systems. High investment costs are associated with some of the retrofit measures outlined above, e.g. subsurface flow constructed wetlands with pre-treatment biofilters (Jenssen et al. 2005) or replacement of old STs with modern systems or package treatment plants (PTPs). The systems tested by Jenssen et al. (2010) were estimated to cost in the region of £10,000, with the high initial cost related to the proprietary material (Filtralite P®) filter media. Operation and maintenance costs for the Nordic systems were related, primarily, to electricity to run the distribution pump, annual inspection of the pump and biofilter, and tank desludging.

Dubber and Gill (2014) carried out an analysis of installation and annual running costs for a range of decentralised package treatment systems comparing single house systems and small decentralised systems. The data from this analysis are shown in Table 18.

The relative (cost) effectiveness in reducing P and pathogen pollution of the environment is unclear. This requires further investigation. It is also possible that limitations on space may affect the practicality of installing some potential solutions, e.g. upgrading STS by the installation of constructed wetlands. Most of the systems outlined above can be retro-fitted, except perhaps the complete replacement of old STs with more modern STs or PTPs.

Table 18 Average capital cost and annual electricity running costs for household and community scale systems (Adapted from Dubber and Gill 2014)

	Single family household[a] system (costs: £/capita)		Small community[b] system (costs: £/capita)	
System	**Installation cost**	**Running cost**	**Installation cost**	**Running cost**
ST with drainage field (percolation area)	894	17	n/a	n/a
Membrane Bioreactor	1422-1580	40-55	474-948	<24
Moving Bed Bioreactor (e.g. Balmoral Hydroclear)	1185	16-24	474-632	<8
Filter media	664-948	0-4	277-553	0-4
Sequencing Batch Reactor (e.g. Balmoral SBR)	490-711	3-6	237-395	3-6
Submerged aerated filter (e.g. Balmoral CAP)	375-664	16-24	119-198	<14
Conventional activated sludge	427-474	16-24	198-356	<12
Rotating Biological Contactor (e.g. Klargester)	n/a	13	332-474	<4

n/a = not available ([a]single family house of 3-6 pe, [b]small community ≥20 pe)

6 Sustainable waste management solutions

This section considers options for sustainable use of ST sludge removed from OWTSs. The practicality and economic feasibility of the options discussed below is largely dependent upon the scale of the treatment system, and level of demand for sludge material.

6.1 Nutrient recovery

Verstraete et al. (2009) suggest that, in many countries, the majority of the value in sewage will be water, followed by the methane potential. In Scotland, where water scarcity is not as prevalent, the water value is likely to be negligible. Hence value in sewage is more likely to be associated with nutrient and methane value. As yet, the level of potential nutrient recovery from OWSTs is small and, therefore, unlikely to provide a revenue stream for households. Verstraete et al. (2009) estimate that the P-value per m^3 of sewage is in the region of €0.01(Table 19). As global resources of P become depleted and its value increases, the ability to concentrate nutrients from waste sources will increase in importance.

Table 19 Estimated recoverable value from sewage

Potential recovery	Amount in 1 m^3 of sewage	Value per m^3 (Euro)
Water	1 m^3	0.25
Nitrogen	0.05 kg	0.01
Methane	0.14 m^3	0.05
Organic fertiliser	0.10 kg	0.02
Phosphorus	0.01 kg	0.01

The most basic form of nutrient recovery from recovered septic sludge is via composting. Septic tank sludge, as with all sewage sludge, contains high concentrations of pathogens, nutrients and organic compounds. However, to make septic sludge suitable for land application, effective treatment to stabilise and disinfect it is required to reduce the potential for environmental contamination. A number of authors have considered effective mechanisms for stabilising septic sludge. Valencia et al. (2009) found a reduction of 99% in total coliforms and 100% in faecal coliforms in compost co-digested with municipal solid waste (MSW). A number of mechanisms were proposed for die off of pathogens including high salinity, high H_2S, or high ammonia concentrations. Rodriguez-Canché et al. (2010) investigated the potential of using vermicomposting (worm composting) processes to inactivate pathogenic organisms. The study showed that a pre-composting period of 2 weeks was required to reduce ammonia levels to a less toxic level suitable for worms. Optimal conditions were 15-20°C with optimal growth rates above pH 4.5 but still within the acidic range, and with a humidity level of 80-90%. The study showed good results for pathogen reduction after 60 days. Carrying out a vermicomposting operation would require sludge to be kept in a well aerated system for up to 60 days, and consideration of local odour impacts would be necessary, along with potential end-uses of the material. It should be noted that biosolids derived from human sewage may not be included in compost seeking to achieve PAS 100:2011 standards. Where biosolids are incorporated into compost or spread on agricultural land, restrictions apply to the immediate use of that land for grazing or crop harvesting. This may limit some potential end uses for the material on land used for food production or grazing.

In addition to composting sludge directly, plant material grown in wetland systems has been shown to take up nutrients. These plants can be harvested and composted as a means of recovering nutrients from septic sludge, or they can be harvested for use in anaerobic digestion (AD) systems. A study by Curneen and Gill (2014) demonstrated effective uptake of nutrients from septic effluent utilising willow trees. Although the effluent only provided a small overall proportion of the nutrients available to the plant within the soil, there was a preferential uptake of the soluble form provided by effluent.

An additional means of nutrient recovery is the reuse of saturated filtration or sorption media used within treatment systems. The study by Jenssen et al. (2010) investigated the nutrient recovery potential from the light weight aggregate (LWA) and Filtralite-P™ material. Over time, the P-sorption capacity of this material became reduced suggesting the saturation of binding sites. Despite the retention of P, actual accumulation after 1-2 years was small and, therefore, not useful as a P fertiliser. Potted plant assays using saturated Filtralite-P™ (7500 mg P /kg) showed that ryegrass was able to utilise the P when grown in the Filtralite P™ filter media, suggesting that the P is in an available form for uptake. However, uptake was less than for an equivalent dosing of $Ca(H_2PO_4)_2$ fertiliser. The Filtralite P™ provided a smaller level of P-utilisation (24% *cf.* 37%) providing approximately 65% of the fertiliser value of the $Ca(H_2PO_4)_2$ fertiliser. No adverse impacts on plant growth were detected due to the use of the filter media. Any reuse of material in this manner should consider the presence of heavy metal and hygiene issues.

6.2 Energy generation potential

Use of sludge in a concentrated format in an AD plant is one potential energy generation process that could be used for ST sludges. This process could deactivate pathogens within the sludge and enhance its compostability. Although the capture of methane gas from the AD processes occurring within OWTS is common in some developing countries, it is not practised to any significant degree in developed countries. The reasons for this include hygiene and safety issues, and the availability and affordability of heating and cooking fuels such as mains gas and oil. In addition, the warmer ambient temperatures in many developing countries allow the AD process to operate more efficiently (van Haandel et al. 2006). The only likely utilisation of septic sludges in Scotland for energy production would be in the form of a communal AD system, or co-digestion with other feedstocks such as municipal waste or farm wastes. The economic feasibility of such a system would be related to the quantity of available feedstock. Verstraete et al. (2009) estimated the value of recoverable elements of wastewater from onsite treatment systems in the Netherlands (Table 19) and found that in the region of 0.14 m^3 of methane could be derived from 1 m^3 of sewage, with a value of approximately €0.05.

It is unlikely that single units or small communal systems for a few houses, which may only be desludged once every two years or more, could provide an adequate volume of feedstock to make a small communal system economically feasible. Due to the frequency of loading and volume of feedstock required, only a larger scale system would be viable and this would need staged inputs throughout the year to maintain effective operation. This would require a well-coordinated collection scheme that would balance desludging frequencies with system loading requirements. This would require cooperation between landowners and desludging firms to secure a regular and adequate supply of feedstock. The potential to combine sludges with AD systems at existing sewage treatment works could provide a more sustainable end-use of septic sludge and sufficient flexibility for addition of materials where a large feedstock already exists. Similarly, co-digestion with existing farm waste or food waste fed AD systems could be possible. Operators of these types of systems would have to consider the additional regulatory requirements for co-digestion of human sewage based sludges and potential end uses of digestate material. It should be noted that co-digestion of waste with sewage based sludges may make PAS 110:2014 AD digestate standards unattainable, thereby limiting the end use of the digestate.

It is unlikely that these options would provide significant cost savings to site owners, as the cost for desludging will be largely related to transportation and time related to the physical desludging process. However, companies offering desludging services may be able to offer reduced rates to secure feedstocks that have lower disposal costs than current arrangements, or could potentially generate a revenue stream.

An alternative energy generation option would be drying of sludges and incineration. This process would require facilities for drying of sludge that are adequately designed to prevent health or hygiene concerns. In addition, sludge incineration could potentially release increased particulates and unpleasant odours into the atmosphere. This would, potentially, make it unsuitable for combustion in biomass systems, and more suited to waste incineration units. This type of facility is of limited availability across Scotland, so this option would only be viable where logistics and transport to incineration sites were economical. Although drying and combustion of sludges may be possible in ambient conditions, it is likely that an energy source would be required to provide necessary drying in Scotland, particularly for sludges with a high moisture content. The costs of this could negate any potential revenue stream.

7 Conclusions

This study has reviewed a large volume of literature relating to the efficiency of various treatment measures in reducing P and pathogen concentrations in effluent. The previous sections have described measures suitable for P and pathogen reduction and sustainable waste management options for ST sludge.

A strong theme running through the literature is the site specific nature of treatment efficiency, in relation to the strength, volume and continuity of flow of effluents, and of site specific characteristics. Some receiving waters will require a reduction in either P or pathogen concentrations to achieve WFD objectives, whereas others will require a reduction in both. So, a broad recommendation of measures is not practical, as site specific issues, including the level of impact and sensitivity of the local receiving environment, will determine both the level of treatment required and suitability for various measures.

7.1 Summary of measures

Table 20 summarises the measures discussed for both P and/or pathogen reduction, along with factors affecting the practicality of applying these measures and specific requirements that may influence the possibility of their retrofit to existing sites.

Table 20 Summary of measures

Measure	Removal of P possible	Removal of pathogens possible	Practicality	Site requirements	Cost	Likely uptake
Change of diet	Yes - quantity unknown	No	Depends on user dietary preferences	None	None	Unlikely
Using phosphate free detergents	Yes - up to 50%	No	Legislation will ensure this is implemented	None	Low	Guaranteed
Reducing phosphate additions to domestic water supplies	Yes - quantity unknown	No	May be practical in the longer term if increased P-dosing fails to achieve reduced limits for Pb in drinking water	Household inspections, renovation works	Potentially high if replacement or renovation of lead piping carried out	Unlikely without funding and awareness raising of health benefits of lead pipe replacement
Reducing levels of food waste being flushed down the sink or toilet	Yes - quantity unknown	Unknown	Awareness raising could assist. May be more practical for hotels, restaurants	None	Low - awareness raising	Possible with awareness raising
Appropriate site and setback distances	Likely	Likely	May require change in building regulations (linked to risk based approach)	Increased distance to water body. Use depth of water table to determine location of treatment/effluent release	Related to increased land take and pipe distances	Possible
Risk based measures	Unknown	Unknown	Targeting measures to most at risk sites	None	Cost of consultation, deregulation	Currently being applied in England
Awareness raising	Unknown	Unknown	Practical if providing advice on operations, inspection and maintenance of systems to reduce pollution	None	Low if electronic; costs associated with leaflets or public events	Likely
Manipulating pH	Unknown	Unknown	Depends on processes, may enhance chemical precipitation	Access for dosing, may be more suited to multi-chamber system, may require electricity if automated dosing mechanism	Low	Unlikely
Managing in-tank temperature conditions	Unknown	Possible at higher temperatures	Functioning of system may be improved in winter with system insulation	Insulation of tank and/or CW may require site disruption for retrofit; may require electricity for heating tank	Cost of insulation or heat source	Unlikely
Replacing old tanks with new tanks: Tank design (baffles and shape)	Yes	Yes	Practical where current system is poorly functioning or leaking. Baffles may be more practical for pathogen reduction than P reduction	Removal of old tank and replacement with new requires access for machinery and adequate space for new treatment system	High	Possible
Increased HRT – correcting misconnections	Yes	Yes	Practical as an inspection measure for site owners/occupiers to improve function of their system	Access to pipe connections and knowledge of OWTS	Low	Likely
Increased HRT - desludging	Unknown	Yes	Practical as a maintenance measure for site owners/occupiers; potential conflict with P releases to environment	Access to desludging equipment; consideration of sustainable end uses of sludge	Relatively low	Likely
Increased HRT - grey water separation	Yes	Yes	May reduce effluent concentrations from poorly functioning sites; requires additional treatment system for grey water	Retrofit site requires re-plumbing and new treatment system; new build can incorporate from outset. May require electricity for pumping, or package plant for grey water treatment	High	Unlikely

Measure	Removal of P possible	Removal of pathogens possible	Practicality	Site requirements	Cost	Likely uptake
Introducing chemical additives	Yes	Yes	Depends on site, level of improvement required and dosing mechanism. In tank chemical use may destabilise microbes	Access for dosing, may be more suited to multi-chamber system. May require electricity if automated dosing mechanism used	Medium (depending on additive and dosing frequency)	Possibly as a polishing step
Introducing biological additives	Unknown	Unknown	Depends on site, the level of likely improvement required and the dosing mechanism used	Access for dosing, may be more suited to multi-chamber system	Medium (depending on additive and dosing frequency)	Unlikely
Soak away/drainage field/percolation field or mound system	Yes	Yes	Could provide additional treatment if retrofitted to sites with direct discharges to water body	Land availability for drainage field, suitable soil conditions and slope. May require electricity if distribution pumps used	High, depending on level of site work required	Likely
Lagoons/WSP	Yes	Yes	Depends on site and whether additional polishing required. Can allow for UV treatment or chlorination	Land availability, suitable protection against human exposure to pathogens	Installation and maintenance costs may be high depending on system	Possible
Removing P from discharged effluent using ochre	Yes	Unknown	Depends on site and whether additional polishing required, could be combined in WSP or as filter medium	Land availability for treatment area, or dosing mechanism	High	Possible for additional polishing
Constructed wetland	Yes	Yes	Practical where adequate space allows, facilitates increased adsorption of both P and pathogens. Increased retention time can enhance settling and pathogen inactivation through exposure and predation	Land availability, suitable protection against human exposure to pathogens, consideration of end uses for substrate and vegetation to reclaim nutrients. May require electricity if pumps required	Installation and maintenance costs may be high depending on system	Likely
Sand filter	Yes	Yes	Practical where adequate space allows	Land availability, electricity if pumps required	Installation and maintenance costs may be high depending on system	Likely
Peat filter	Yes	Yes	Practical where adequate space allows	Land availability, electricity if pumps required	Installation and maintenance costs may be high depending on system	Likely
Ultra and micro filtration	Yes	Yes	May be very efficient a pollutant removal but subject to clogging and require a higher level of expertise for operation and maintenance	Land availability, and access for operation and maintenance, will require electricity	Installation and maintenance costs may be high depending on system	Unlikely
Alternative filter media	Yes	Yes	Practical where adequate space on site allows, may require additional monitoring to assess impacts on other pollutant releases (i.e. pH, metals)	Land availability; consideration of end uses for saturated filter material; may require electricity if pumps required	Installation and maintenance costs may be high depending on system	Possible with further evidence of suitability
Combination systems	Yes	Yes	Practical where adequate space on site allows	Land requirement will be higher for a site with multiple treatment steps; may require electricity if pumps required	Installation and maintenance costs may be high depending on system	Possible for sites in sensitive areas, where high level of treatment needed
Package treatment plants	Yes	Possible	May allow for treatment where limited space available onsite	Similar to ST, requires electricity	Range of costs, can be cheaper than ST. Likely to have higher maintenance costs than ST	Possible

Measures such as awareness raising, site planning, and maintenance are likely to reduce the impact of OWTS on the environment. The level of load reduction possible from measures such as awareness raising is difficult to quantify, but it is low-cost and relatively easy to implement. There are steps that site owners/occupiers can take to reduce P loading, e.g. choosing to use phosphate free detergents. It is likely that changing legislation will result in a reduction of up to 50% of P-loading due to removal of P from household detergents, but this is unlikely to affect pathogen concentrations.

The most practical pre-tank measure for reducing pathogen release is the reduction in hydraulic loading to OWTS. This can be through addressing misconnections of roof drains and reducing household water consumption. Both of these are practical measures that can reduce P flushing from STs. Desludging of tanks at recommended frequencies is likely to help maintain HRT for pathogens. Further study may be required to better understand the impact of desludging on releases of P to the environment.

There are few data available on the effectiveness of package treatment plants such as SBRs and RBCs in terms of reducing pathogen and P in effluents. Limited evidence suggests that SBR systems have a greater P reduction capacity than RBC systems. The latter appear to provide slightly higher pathogen removal capacity than a traditional ST, but effluent from this type of system would still require additional polishing before release to the environment. Further research is required to identify the combined effects of PTPs on P and pathogens, with much of the existing research focusing on the reduction of BOD, TSS and N. There is some evidence to suggest these systems could be effective, but results may depend upon operating conditions (i.e. continuous flow).

The most effective measures for P and pathogen removal are post-tank measures that maximise physical removal, through adsorption and filtering, and maintain good conditions for biological breakdown of solids and predation of pathogens. Constructed wetlands show good removal potential for both types of pollutants. The practicality of CW relates to the availability of land for siting a system and the ability to carry out regular maintenance. Constructed wetlands can lose treatment efficiency over time, requiring renewal of substrate, maintenance of inlets and outlets, and removal of vegetation.

Filter systems have also been shown to be effective for both P and pathogen removal, with some variation in treatment capacity being shown by different filter media. Peat filters and sand filters have shown good treatment capacity for both P and pathogens. The practicality of these systems relates to the initial installation cost and the requirement for land (although less land-take may be required than for CW) and the requirement for renewal over time as treatment efficiency reduces. Alternative filter media used in combination systems have also shown high efficiency in reducing both P and pathogens to acceptable levels. These systems also require renewal over time.

The most practical sustainable waste management solutions for ST sludge include composting or co-digestion with other organic material in a centralised anaerobic digestion (AD) facility. Practical considerations for composting include the ability of the composting process to deactivate pathogens. End markets for human waste derived compost may be limited due to regulatory controls, therefore additional treatment measures required for the intended end use must be considered. Similarly, for co-digestion in an AD facility, practical considerations include the logistics of collection and transport of material, as well as potential end uses of AD digestate.

Other considerations for the reclamation of nutrients from ST sludge include the use of filter materials containing adsorbed P that can be returned to land once saturated. As discussed, some materials have already been tested and have shown promising results. A simple method for reclamation of nutrients is the harvesting and subsequent composting of plant material grown in constructed wetlands that are treating ST effluent. The removal of this vegetation is also essential to ensure that P is not returned indirectly to the environment through the die off and decomposition of constructed wetland plants onsite.

7.2 Further work

This study has uncovered a number of gaps in the literature that may require further study. Recommended areas of further study include the following:

- The impact of PTPs on P and pathogen reduction has not been widely evaluated. With the number and variety of PTPs now available, an in depth study on the performance of these systems for parameters other than BOD, TSS and TN would be beneficial.
- The impact of desludging frequency on releases of P and pathogens to the environment has not been widely evaluated. There is some suggestion that the optimum desludging frequency for pathogens may not be the same as for P, and desludging may result in unintended releases of P to the environment. Further study could assist in determining optimum desludging frequency.
- In consideration of pathogen reductions, viruses were not considered in depth in this study. Viruses have been attributed to human illness following shellfish consumption, with sewage overflows following rainfall events a potential link to viral contamination (Kay et al. 2007). Studies have associated an increased risk of illness from private water supplies with the presence of an OWTS such as a STs (Risebro et al. 2012). Viruses tend to be more infectious than bacteria, with the human rotavirus being one of the most infectious and widespread human enteric pathogens (Gerba and Smith 2005; Kargar et al. 2013). Although viruses may be removed by similar processes to indicator bacteria (i.e. adhesion to sand, clay, suspended colloids, silt and sediment, filtering, inactivation through Cl, O_3, UV exposure), their small size and increased mobility in the aquatic environment make them more difficult to control. A more in depth study of the impact of various treatment measures on viruses is needed.

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Annex 1: Development and workings of septic tanks

Septic tanks, which are the most commonly used OWTSs (Wood et al. 2005), originated in France and were introduced into England in 1885 (Canter and Knox 1985). They usually comprise a one- or two-chamber system (Figure 12) that holds sewage for a short period of time. This allows the solids to settle as sludge in the bottom of a tank, where it undergoes anaerobic digestion, with oil and grease forming a scum at the top. The tank produces a relatively clear liquid effluent that is discharged to the environment through an outlet pipe (Figure 13). The retention time and volume of the effluent varies over time, depending on the influent volume.

Originally, STs were designed as single chambers made of reinforced concrete or brick. Modern designs are frequently constructed from fibre-glass and are deemed more efficient in terms of removal of suspended solids (SS) and reduction of biochemical oxygen demand (BOD) (Canter & Knox 1985). Modern systems may incorporate electrically operated aeration pumps, which can be used in continuous or discontinuous (sequencing batch) mode. In addition, some may also incorporate processes of filtration or polishing through organic or inorganic based media prior to dispersion of the effluent to the soil soakaway or drainfield. Other systems include an above ground soil percolation mound for attenuation of final discharged liquids. These are installed as a solution to problems of high water table and low soil permeability (Macintosh et al. 2011).

Many older properties in the UK still rely on their original, often Victorian, STs for the treatment of waste water but these are often under-sized for today's patterns of water use, e.g. frequent bathing or showering, and use of domestic appliances (Selyf Consultancy 2002). This can result in the discharge of the untreated waste into the environment. Older tanks may also receive runoff from roofs, which flush the system through very quickly during heavy rainfall. Connections of roof drains to STs is not recommended, however older properties and OWTSs may have connections that site occupiers are unaware of.

Raw ST effluent contains suspended solids, dissolved P and nitrogen (N), and potentially pathogenic bacteria and viruses. Therefore, it is not of suitable quality for discharge directly to a water body and requires additional treatment. Today, STs are typically used in conjunction with a drainfield or soakaway. Together, these are known as a septic tank system (STS) (Figure 14). Abiotic and biotic processes within the drainfield, such as filtration, adsorption, nitrification and denitrification, purify the tank effluent, which then disperses to groundwater (Withers et al. 2014). Digestion of waste in STs occurs naturally through a range of biological reactions that involve anaerobic and facultative anaerobic bacteria, and larger organisms growing in the wastewater (Pradhan et al. 2011a). During the early development of STs, their effectiveness in relation to P or pathogen removal was not considered to be a key issue and, therefore, this aspect of ST performance has not been studied extensively (Eveborn et al. 2012).

The effectiveness of STS in terms of their wastewater treatment capability varies widely, with little or no integrated regulatory control of these systems (Withers et al. 2014). The presence of many septic tank systems in rural areas is often overlooked when tackling the problem of diffuse water pollution. Applying additional measures to improve performance of existing septic tanks, and improving the quality of septic effluents, could lead to a substantial decrease in nutrients (and pathogens) entering waterbodies (Ockenden et al. 2014). However, data on the scale of potential water quality improvement and the economic costs of upgrading and regulating existing systems are scarce (Macintosh et al. 2011).

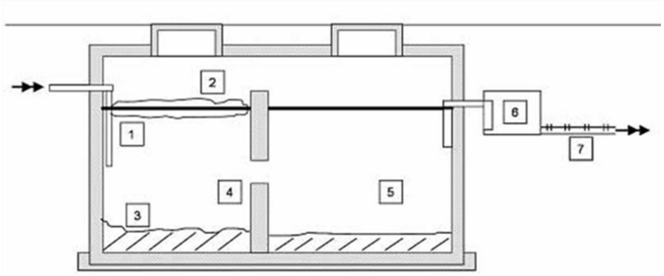

Figure 12 A standard two chamber septic tank design.
1. inflow;
2. floating scum;
3. settled sludge;
4. connection between chambers;
5. secondary chamber;
6. outflow and effluent inspection
7. soakaway or drainage system;
(reproduced from Hilton et al. unpublished)

Figure 13 Clarified effluent from a septic tank.

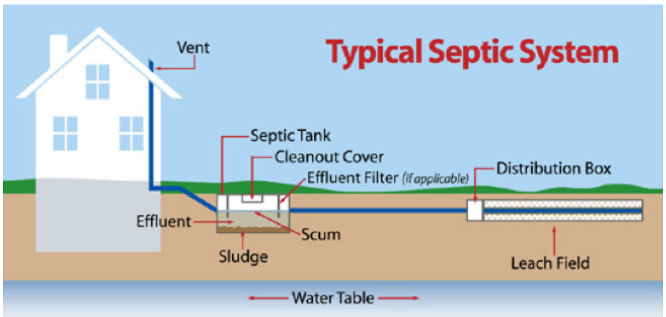

Figure 14 Schematic of septic tank and drainfield in relation to water table (http://www.typesofsepticsystems.com/wp-content/uploads/2012/02/septic1.jpg).

Sources and concentrations of P in effluent

Effluent quality is dependent upon the constituents of the raw wastewater that enters the OWTS and the degree of 'purification' that occurs within it. The main sources of P in raw domestic wastewater are summarised in Table 21.

Brownlie et al. (2014) suggested that decreasing P inputs from households to OWTS (i.e. reducing the source of the problem) may be more effective than decreasing P outputs to the environment by applying expensive engineering or chemical based solutions (i.e. treating the effects of the problem).

Table 21 Source apportionment of P in raw domestic waste water (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%

There are few data on the level of P in ST influents and effluents available in literature, with most studies on the performance of OWTSs focusing on biochemical oxygen demand (BOD), total suspended solids (TSS) and, sometimes, either a form of N or of P. This research focus is not unexpected, because the main design parameters of these systems are often BOD, TSS and N (Lowe et al. 2007) and this is reflected in the current UK certification of effluent quality from these systems (e.g. EN12566-3 UK Standard). In addition, these studies provide little or no information on factors that have an impact on P output from STs, such as number of people in a household or human domestic behaviour (Brownlie et al. 2014). In general, wastewater treatment in conventional septic tanks is limited, with most of the influent nutrients not receiving significant treatment (Zamalloa et al. 2013).

Examples of published P concentrations (mg/l) in influents and effluents from OWTSs are shown in Table 22. Lowe et al. (2007) found that the average concentrations of total phosphorus (TP) in raw wastewater and in ST effluent were 19.1 mg/l and 12.2 mg/l, respectively, indicating more

Table 22 Examples of published P concentrations (mg/l) in influents and effluents from OWTSs

Influent TP concentration	Effluent concentration SRP	TP		Type of system	Notes	Reference
Unknown	1.9	3.3		Old ST; no soil adsorption bed	Concentrations entering a ditch (field drain discharge, including ST effluent); average from 1 year	Ockenden et al. 2014
Unknown	1.4	1.9		Old ST supplemented with modern tank	As above	Ockenden et al. 2014
Unknown	4.83 (0.32-10.56)	9.06 (4.45-18.01)		ST	Median concentrations from 4 STs	Brownlie et al. 2014
Unknown	8.82 (2.26-11.91)	11.86 (5.79-14.43)		ST with mechanical mixing	Median concentrations from 1 ST; 4 month monitoring period	Brownlie et al. 2014
Unknown	5.54 (1.42-10.60)	9.31 (1.91-14.44)		ST with chemical dosing and tank with aeration and filter system	Median concentrations from 2 STs; 4 month monitoring period	Brownlie et al. 2014
19.1 (13.05-25.8)	Unknown	12.2 (3-39.5)		ST	Average concentrations based on literature search (n=8 for influent, n=49 for effluent)	Lowe et al. 2007
13.3[a] 6.6 26.8 18.2	Unknown	7.07[b] 5.5 24.0 14.0	0.22[c] 0.04 1.22 0.02	Filter bed system	Results for 4 of the systems tested	Jenssen et al. 2010
Unknown	11.6 14.5 9.4 13.4 10.7 6.6	15.0 18.4 17.4 15.0 12.9 11.6		ST (concrete) ST (brick) ST (concrete) ST (brick) Klargester PTP Unknown	Sampled STs chosen from a range of locations across England	May et al. 2014
Unknown	Unknown	9.00 4.50 1.80		ST Secondary treatment Tertiary treatment	Estimated value for 3 bedroom household using a desk based calculation method described in Section 4.1.2.	Brownlie et al. 2014

[a]septic tank effluent; [b]outlet of biofilter; [c]outlet of filter bed.

than 36% removal of P by OWTSs. However, the range of concentrations varied widely, with actual TP values ranging from 13.05 to 25.8 mg/l for raw sewage, and 3.0 to 39.5 mg/l for ST effluents, suggesting considerable variation in overall levels of P removal across systems. This is further evidenced by values recorded by May et al. (2014) in effluents from six STs in England. These ranged from 6.6 to 14.5 mg/l for SRP and from 11.6 to 18.4 mg/l for TP, with average concentrations across all tanks of 11.0 mg/l and 15.1 mg/l for SRP and TP, respectively.

Sources and concentrations of pathogens in effluent

Pathogens in surface and ground waters are generally derived from the guts and faeces of warm blooded animals. Sources can include failing cesspools, animal derived sources such as feedlots, dairy farms or intensive animal husbandry, and human derived sources such as sewage works, combined sewer overflows or septic tank effluent (Macler and Merkle 2000). Runoff from animal housing and grazing areas can carry pollutants diffusely to surface waters, and grazing animals in proximity to or within water courses can introduce faecal bacteria directly. Improved farmyard management and collection of slurries can prevent some escape of faecal matter into the environment. The process of spreading slurries onto land can enhance pathogen die off by exposure to sunlight, desiccation and filtration through the soil. Additional non-human, sources of faecal contamination to water courses can include run-off that carries diffuse microbial pollutants from grazing wildlife or domestic animals (e.g. dog walking areas). *E. coli* have been found naturally within soils, on plant surfaces (potentially insect or non-mammalian sources) and in pulp and paper effluent (Santo Domingo and Edge 2010). Butler and Davies (2004) estimated that storm water can provide a unit load of faecal coliforms of 0.9-3.8 x 109 counts per impervious hectare of ground.

Microorganisms are necessary for the effective functioning of any wastewater treatment system, and a wide range of bacteria are essential to support the biological processes that occur within a septic tank. These microorganisms are involved in the digestion and breakdown of complex organic compounds into simpler compounds. Some bacterial groups derived from human waste can be harmful or disease causing (i.e. pathogens). Human health risks are a primary driver for measurement and treatment of pathogens released from wastewater treatment systems into the environment.

Human waste contains a number of potential pathogens, including bacteria, viruses, protozoa and helminths. Bacterial species include *Salmonella spp., Vibrio cholera, Shigella spp, Escherichia coli,* and coliforms. Protozoans include *Cryptosporidium spp., Giardia spp*, and viruses include Adenoviruses, Noroviruses, Hepatitis A, Echnoviruses and Coxackieviruses (Malham et al. 2014). Bacteria, at 0.2-2.0 µm diameter, are larger than most viruses, which are typically 0.02 – 0.08 µm diameter (Lowe et al. 2007, Metcalf and Eddy 2003). These size differences will affect the ability of some treatment processes to remove both types of pathogens. As some bacteria and viruses can also react differently to environmental conditions, and to disinfection processes, some treatment measures may not be effective in controlling both.

The bacterial groups of interest in this study are *E. coli* and intestinal enterococci, which are used as faecal indicator organisms (FIO) in shellfish and bathing waters in Scotland. These bacteria belong to the phyla *Proteobacteria* (including *E. coli*) and *Firmicutes* (including Enterococcus spp). Enterococci and *E. coli* are found in the guts of warm blooded animals including humans (Ahmed et al. 2005). *E. coli* is a faecal coliform. Total coliform measurements generally refer to a range of coliform organisms that can occur in the faeces of warm-blooded animals, cold blooded animals, and on plant surfaces and in the soil. Some coliform species may not require a host to survive in the environment. However, faecal coliforms and enteric pathogens tend to have a reduced survival time in the environment (Alhajjar et al. 1988). Much of the literature on faecal pathogen monitoring presents measurements of faecal coliforms (FC), total coliforms (TC) and *E. coli*. A significant body of research has been carried out to assess how to improve the use of indicator groups to identify the source and intensity of wastewater releases to the environment (Alhajjar et al. 1988, Ahmed et al. 2005, Savichtcheva and Okabe 2006, Stedfelt et al. 2006, Yan and Sadowsky 2007, Kronlein et al. 2014). The FIOs currently in use provide an indication of the presence and intensity of faecal contamination within the environment, but it should be noted that these indicator organisms are principally indicators of bacterial contamination only. More suitable indicators of faecal contamination may be required to fully assess the environmental and public health risks associated with the wider range of pathogens released from wastewater treatment system, particularly viruses.

Pathogen concentrations: The basic composition of raw domestic wastewater can vary by the number of users, household activities and behaviours. There is limited recent data available on the influent and effluent concentrations of pathogens from septic tanks, particularly for *E. coli* and enterococci. Table 23 provides estimates of pathogen strength in raw wastewater and basic septic tank effluent from the literature. Onsite treatment systems may show a higher level of variability than centralised treatment works due to variations in household water use and wastewater production on a local scale.

Table 23 Mean pathogen concentrations in raw wastewater and septic tank effluent (UK and Ireland)

Parameter	Mean concentration (cfu /100 ml) in raw wastewater	in septic tank effluent	Reference
Total coliforms	3.9×10^7	2.5×10^7	Kay et al. 2008
	-	7×10^8	Gill et al. 2007
	$2.0\text{-}3.5 \times 10^8$	-	Kadam et el. 2008
Faecal coliforms	1.2×10^7	-	Harrison et al. 2000
	$2.0\text{-}8.0 \times 10^7$	-	Kadam et el. 2008
	1.7×10^7	7.2×10^6	Kay et al. 2008
	-	2.9×10^5	Pundsack et al. 2001
Enterococci	1.9×10^6	9.3×10^5	Kay et al. 2008
	1.0×10^6	-	Blanch et al. 2003
E. coli	$1.2\text{-}3.3 \times 10^6$	-	Kadam et el. 2008
	-	5.0×10^5	Gill et al. 2007

Previous work by SNIFFER (2010) estimates the average concentration of septic tank effluent to be 1×10^8 cfu /100 ml microbial contaminants, with OWTSs contributing 23.5% of the diffuse microbiological loadings to receiving waters. This contribution will vary significantly by water body, density of OWTSs, and level of treatment.

Annex 2: Factors affecting pathogen loading and survival

Pathogen loading

The volume and strength of wastewater entering any treatment system can vary by household makeup, behaviour, seasonality and activity types (Butler and Davies 2004). Working families may show different patterns of wastewater production than retired. In addition to more general variations in effluent quantity and strength, peak flows can occur on a daily basis, seasonally or in conjunction with household events. As discussed in Section 5, the misconnection of roof drains or surface water drains to a ST can increase hydraulic loading and, during storm events, could increase turbulence and reduce effluent retention time in the ST. For households that use water more efficiently, with low flow taps and/or low volume flush toilets, wastewater quantities may be reduced but may also be less dilute.

Pathogens can be transported into the environment attached to suspended particles of waste, so processes that enhance settlement of solids within a treatment system can reduce the release of pathogens. For example, flocculent (particles of organic matter, silt and clay) can act as a pollutant sink for pathogens by providing binding sites that enhance their settlement within the tank. However, in an overloaded or turbulent system, pathogens adsorbed onto solid particles can be flushed through the system. In this case, attachment to particles may enhance their transport.

The use of chemical disinfectants and washing agents may have some impact on pathogens. Microorganisms involved in the digestion and removal of solids within a ST can be inactivated by high concentrations of household chemicals. Ip and Jarret (2004) demonstrated that the use of household disinfectants adversely affected the ability of OWTS to reduce BOD and solids. Although the study did not show a change in the concentration of faecal coliforms (FC) in effluent between dosed and un-dosed systems, Chabaud et al. (2006) showed a correlation between removal of protozoa in septic effluent and subsequent pathogen removal in the receiving environment, probably due to reduced predation. It is likely that the high level of dilution present in most household systems will be sufficient to compensate for typical household chemical usage. However, excessive household use, or small commercial premises utilising high volumes of disinfecting chemicals, may have a negative impact on the microbial community within a ST or drainage field, reducing treatment efficiency. Currently, there is inadequate research available to quantify the level of impact of disinfectants on pathogen loading to the environment.

Pathogen survival

The movement of pathogens from a ST into a secondary treatment system, or through the environment, can be affected by a number of physical, biological and chemical processes. Site characteristics, properties of infiltration surfaces, soil media, pathogen size, degree of saturation, presence of predators, temperature, pH, aeration and UV radiation among other factors can influence the level of pathogen removal that can be achieved (Stevik et al. 2004, Chabaud et al. 2006). The process of flocculation in the environment is affected by salinity, turbulence, sediment concentration, pH, and organic matter content. Hence the movement and survival of pathogens in the environment is different in freshwater and marine environments where river flow and tidal movements impact turbulence, and physicochemical properties are different (Stedtfelt et al. 2006, Malham et al. 2014). This section provides additional details on the factors that are important to enhancing pathogen inactivation, and thus reducing loading to the environment.

Site characteristics

Maximising the residence time of effluent between its source and a receiving water body increases the likelihood of adsorption, filtration or die off (Cave et al. 1999). The linear velocity of effluent within the soil is related to the gradient, hydraulic conductivity and porosity of the medium. The slope of the drainage field can affect contact time, with more extreme slopes resulting in reduced contact time with soil particles. Wolf et al. (2006) list a number of site specific considerations in relation to protecting groundwater from OWST contamination. These include lot size and distance to the water body, soil type and percolation rate, separation distance between the water table and bedrock, topography of the site, flooding frequency and density of development. These factors are also considered in British Standard BS 6297:2007 on the design of drainage fields (BSI 2008).

Direct contact with the soil can result in pathogen removal through adsorption onto soil particles or through chemical binding. The process involves either surface adsorption to particles by overcoming repulsive forces and forming a weak interaction (reversible attachment, affected by changes to ionic strength) or adhesion through the formation of polymer connections by the pathogen to adsorbent particles (irreversible attachment) (Stevik et al. 2004). These processes are influenced by a range of physical and chemical properties of the filter media, the presence of organic matter, temperature, hydraulic loading, ionic strength, pH, surface charge of particles, and bacterial concentrations.

Drainage fields need to be sited in locations that do not add risk to either surface or ground water bodies. Locations in close proximity to surface waters, or in areas of high or seasonally variable water tables, can experience saturated conditions that do not promote removal of pathogens but, instead, enhance their transport in the environment and increase the likelihood of contamination events. Bedrock conditions can also impact on the risk of groundwater contamination within areas of high bedrock reducing infiltration capacity. Proximity to fractured limestone aquifers can present a particular risk of pathogen contamination reaching groundwater (Borchardt et al. 2011). Katz et al. (2010) found that, where the depth to limestone bedrock was shallow coupled with a high hydraulic loading rate, there was a higher detection of FIOs and enteric viruses in drainfield wells. The authors concluded that areas with karstic aquifer systems are more at risk of septic pollution than non-karstic systems. There are few locations in Scotland where limestone bedrock may be of concern, however specific measures may be required to address existing OWTS in these areas.

Filter media structure in the drainage field

Soils that are free draining, such as sandy soils, may not allow for sufficient contact time for pathogen removal. There is evidence to suggest that coarse-textured soils are less effective at treating septic system effluent than finer textured soils (Harrison et al. 2000). Less free-draining soils such as clays may allow for increased contact time with finer soil particles, hence increasing adsorption. Small clay particles are effective at pathogen removal due to their relatively high surface area and positive charges on their edges, which increase adhesion to negatively charged bacteria (Stevik et al. 2004). However, clay soils are more prone to clogging. The presence of organic matter in the soil can increase adhesion of bacteria but dissolved organic matter can also compete for binding sites. A study by Pang et al. (2003) found that the mechanism of removal of faecal bacteria within the soil was mainly due to filtration (87-88%) as compared to natural die off (12-13%). In contrast, the study also found that filtration only accounted for 55% of the removal of phages (viruses) as compared to 45% of the removal attributed to die off, which suggests a different mechanism for virus removal compared to bacterial removal. Gill et al. (2001) showed that COD, N and pathogen removal (*E. coli*) occurs in the percolation gravel and upper 300 mm of the subsoil in ST soakaways.

Pathogen shape and size can also influence the ability of filter media to remove pathogens, with larger pathogens more readily filtered out of effluent than smaller ones. Pathogen size may not be constant throughout the year, with the potential for desiccation or starvation reducing size and, hence, greater transport through filter media (Stevik et al. 2004). Pathogen shape can also affect the rate of transport through filter media, with long rod shaped cells being more readily transported than spherical cells (Stevik et al. 2004). Cell appendages may also affect removal rates, with those with appendages showing reduced transport in comparison to smooth surfaced cells. These factors may impact on the control measures implemented for specific pathogens.

Biomat formation

For drainage fields receiving septic tank effluent (STE), movement of microbes and organic matter through the soil over time results in the formation of a biomat. Meschke and Sobsey (1999) suggest that the biomat provides the majority of pathogen removal in a drainage field, with the mechanisms of filtration, adsorption and predation dominating. The presence of the biomat can also enhance irreversible adhesion of pathogens to particles due to the presence of polysaccharides that enhance bonding. Beal et al. (2006) propose that the biomat can be a limiting factor affecting the efficiency of the drainfield to retain microbial pollutants and this is supported by field experiments by Postma et al. (1992), O'Luanaigh et al. (2009) and Gill et al. (2007). The presence of the biomat can increase the retention time of effluent in the filter media, and can enhance filtration by clogging pores (Stevik et al. 2004). Once formed the biomat becomes a regulator of downward flow in a drainage field (Beal et al. 2006). The biomat may be most effective in the retention of larger bacteria, with smaller bacteria and viruses passing through (Meschke and Sobsey 1999). The soil matrix becomes more important for these smaller microbial groups through adsorption to particles or chemical binding.

The formation of the biomat is impacted by the loading rate, BOD, suspended solids, aeration and soil properties as well as concentration of STE (Beal et al. 2006; Stevik et al. 2004, O'Luanaigh et al. 2009 and 2012). Tomaras et al. (2009) found that open infiltration surfaces released higher numbers of FC and *E. coli* in comparison to gravel-laden surfaces, suggesting that gravel surfaces may be more favourable to biomat formation. Less free draining soils can be prone to clogging and, where this occurs, may cause a reduction in aerobic conditions in the biomat (Meschke and Sobsey 1999). A properly functioning biomat needs to be unsaturated and aerobic to allow the established microbial community to consume or out-compete pathogens derived from STE. In seasonally occupied properties, where wastewater effluent releases are variable throughout the year, biomats may not become fully established due to seasonal variations in effluent flow, particularly in porous soils. This can cause localised plumes of untreated effluent to travel through unsaturated soil towards ground water or surface waters and the drainfield alone may not be sufficient to remove bacterial contaminants (Postma et al. 1992).

Aeration/ Dissolved Oxygen (DO)

There is evidence that dissolved oxygen concentration correlates with bacterial die off in the environment (Kadam et al. 2008). An aerated system may result in more acidic conditions that could impact pathogen survival (Potts et al. 2004). However, the impact of aeration on the microbial community in the soil or biomat is likely to be a more important factor. Removal of pathogens may be enhanced where the conditions for the micro and mesofauna involved in removal are improved, such as through adequate aeration. Potts et al. (2004) demonstrated that increasing aeration levels in the drainage field enhanced pathogen removal from 98-98.6% to 99.2-99.9%. The process for removal by the microbial community is likely to be by competition for resources and predation (Kadam et al. 2008). Malham et al. (2014) suggest that predation is a more important factor in faecal bacteria inactivation than UV degradation once they have reached the water column. Chabaud et al. (2006) also found that there was a significant correlation between the presence of active protozoa (bacterial predators) and the die off rate of pathogens in septic effluent. Hence, conditions that enhance the survival of protozoa can reduce bacterial survival. Chabaud et al. (2006) found that predation by protozoa removed about 66% of total coliforms and 45% of faecal coliforms in laboratory based experiments. Survival was approximately 10 times lower in a septic effluent that contained protozoa in comparison to effluent free from protozoa.

Moisture and rainfall

In dry conditions, some pathogens can become inactivated through desiccation. However, during high rainfall events, desorption can take place, remobilising pathogens (Meschke and Sobsey 1999, Arnade 1999). Unsaturated filter media performs better at retaining pathogens than saturated media. High rainfall events that increase saturation and flow rates reduce the possible contact time with filter and adsorption media resulting in releases of pathogens (Pundsack et al. 2001, Stevik et al. 2004). Similarly, spring thaw events in areas of snow or ice accumulation can cause seasonal flushes of material through systems. Rainfall can also impact the ionic strength of effluent released into the environment. High ionic strength can reduce the repulsive forces and increase adsorption. However, the type of ion present may be important and there is evidence that trivalent and divalent cations (e.g. Fe^{3+}) adsorb bacteria differently to monovalent cations (e.g. K^+) (Stevik et al. 2004, Chen and Walker 2007). The low ionic strength of rainwater may be a reason why high rainfall events remobilise pathogens (Okoh et al. 2010). In addition, rainfall events may transport pathogens from other environmental sources and can cause the re-suspension of pathogens that had previously settled in the sediments of receiving waters.

Temperature

It is widely recognised that pathogen survival is enhanced in cooler temperatures, which suggests that some systems may exhibit reduced removal efficiency in winter (Pundsack et al. 2001). At colder temperatures, there is reduced adsorption due to changes to the surface properties and physiology of bacteria, such as increased viscosity of surface polymers, which reduces their adsorption to filter media particles (Stevik et al. 2004). Bell et al. (2009) also note that predation by protozoa is reduced at lower temperatures, which could enhance faecal bacteria survival. Pundsack et al. (2001) cite evidence of an inverse relationship between temperature and faecal coliform survival below 15°C. This relationship may not hold below 0°C, as freezing temperatures can inactivate some pathogens by causing cell damage (Meschke and Sobsey 1999) and freezing winter temperatures can affect the efficiency of treatment systems, such as constructed wetlands, by halting efficient effluent flow through treatment systems. Most below ground STS are sufficiently insulated from winter freezing. However, above ground treatment systems may show some reduced treatment efficiency. There is evidence that insulating constructed wetlands can prevent winter freezing and, hence, reduce hydraulic failures thus allowing treatment to continue year round (Wallace 2000).

Ultraviolet radiation (UV)

High light intensity (> 450 nm) can contribute to pathogen inactivation (Jimenez et al. 2010). Natural UV radiation can be important to pathogen inactivation in the environment. Maiga et al. (2009) demonstrated that exposure to sunlight accounted for the majority of *E. coli* inactivation in algal waste stabilization ponds in Burkina Faso, and that the effectiveness of UV exposure was greater in shallower ponds, showing a synergistic effect with increased temperature and dissolved oxygen. The study recommended a depth of 0.4 m to enhance UV inactivation in waste stabilisation ponds. In Scotland light intensity is much reduced over the winter months, hence the level of effectiveness may be reduced as compared to locations closer to the equator. Schultz-Fademrecht et al. (2008) found a correlation between increased light intensity and inactivation of FC and enterococci in treated effluent, estimating the average inactivation rate coefficient for a site at 50° N latitude in Germany was 12. 7 days for faecal coliforms and 9.3 days for Enterococci. In Scotland, at 5 to 8 degrees of latitude further north, the winter UV exposure would mean an increased inactivation time. The Schultz-Fademrecht et al. (2008) study also found greater survival of pathogens within an in-stream biofilm exposed to UV. Pathogens attached to flocculent in the water column, or stabilised within the biofilm, may be less susceptible to UV inactivation due to shading (Malham et al. 2014). High levels of turbidity can also reduce the transmission of UV radiation within the water, reducing pathogen inactivation.

pH

Although bacteria tend to have a relatively wide range of pH tolerance, most show reduced survival at extreme pH levels below 4 or above 9.5 (Metcalf and Eddy 2003, Jimenez et al. 2010). Pundsack et al. (2001) found that systems with a lower pH showed decreased pathogen survival compared to systems with a higher pH. In contrast, other studies showed higher pathogen survival at low pH compared to high pH (Stevik et al. 2004, Bhat et al. 2012). Bacteria such as *E. coli* have shown pH tolerance down to pH 2.5 in the water environment (Coffey et al. 2007).

The effect of pH on pathogen survival can depend on the overall chemical composition of the wastewater, with the influence of pH on survival being more important at low ionic strength. At lower pH, the net negative surface charge of bacteria becomes increasingly positive. At higher pH, these same bacteria generally have a net negative surface charge and therefore the presence of increased cations enhances the adsorption process, immobilising pathogens (Meschke and Sobsey 1999). In addition, at pH above 7, the ammonium ion (NH_4^+) becomes less dominant and ammonia (NH_3) concentrations begin to increase. An increase in NH_3 can increase deactivation of enteric bacteria (Vinneras et al. 2008); hence high pH can result in indirect ammonia toxicity.

Exposure time

Pathogens may survive in soil and water environments for an extended time. Coffey et al. (2007) reported that *E. coli* can survive 13-245 days in the water environment. *E. coli* have a relatively low die off rate in the water environment and can, therefore, be transported further than some pathogens. *E. coli* numbers can also increase. Gerba and Smith (2005) estimated pathogen survival in soil as a common maximum of 2 months and an absolute maximum of one year for bacteria. Table 24 presents the approximate survival time of the main pathogen groups in soil as reported by Gerba and Smith (2005).

Table 24 Pathogen survival in soil (Gerba and Smith 2005)

Pathogen	Absolute maximum	Common maximum
Bacteria	1 year	2 months
Viruses	6 months	3 months
Protozoa	10 days	2 days
Helminths	7 years	2 years

Table 24 suggests that, although pathogens may be bound within soil and sediments, they may not be permanently inactivated for some time. Tidal condition, river hydrology, rainfall and storm events can impact the persistence of microbes in the environment by causing the re-suspension of materials that have settled into river or estuarine sediments (Stedtfelt et al. 2006).

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