CatchmentCARE
Community Actions for Resilient Ecosystems


Alternatives to willow coppice plantations (WCPs) for the treatment of small-scale waste-water treatment works (WWTWs) effluent

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Contents

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Executive Summary

This report, for the SEUPB funded CatchmentCARE project, is part of T1 *"Scoping and Action Targeting"* and specifically part of Activity A.T1.5 *"Hydrological connectivity and Point Source Pollution: Scoping studies for willow biomass as point and diffuse pollution mitigation actions"*. The specific Deliverable is D.T1.5.2 *"Alternative treatments available for small WWTW"*.

Nutrient enrichment of freshwater ecosystems, especially from waste-water treatment works (WWTWs) is a major environmental concern, with high phosphorus (P) emissions potentially resulting in excessive algal growth and degradation of water courses via eutrophication.
Willows have several characteristics (such as being non-edible and having high evapotranspiration rates) that enable their use for environmental remediation. Their use for treating municipal waste-water is also well established, not least because of the similarity between willow nutrient requirements and the typical nutrient characteristics of waste-waters. However, difficulty sourcing nearby land for growing willow coppice plantations (WCP), along with the potential for eventual soil P saturation with continued irrigation, may necessitate the identification of alternative methods to the use of WCPs.

These alternative methods include physico-chemical treatments such as precipitation, sorption and ion exchange; biological treatments such as Enhanced Biological Phosphorus Removal (EPBR); and/or a combination of both. Precipitation, for example, involves adding (dosing) metal salts such as ferric chloride or trivalent metal salts to the waste-water that precipitate out the P, while EPBR relies upon phosphorus-accumulating organisms (PAOs – e.g. Accumulibacter), that accumulate P as polyphosphate which is then used as a "luxury" energy reserve. However, while physico-chemical methods provide reliable P extraction with a relatively simple operation, sustainability concerns limit their usefulness in relation to small-scale WWTPs, and they perhaps best serve as a "polishing" treatment option in combination with other treatments such as constructed wetlands. The extraction of P by EPBR may be variable, with fluctuations in performance dependant on operational factors, local environmental conditions, microbial diversity and the number of PAOs occurring in the system. More recent developments have seen EBPR techniques integrated into membrane bioreactors (MBRs), and the development of "novel technologies" such as active filter media, reverse osmosis and compressible media filtration. In general, however, "novel technologies" are still at the developmental stage and/or are associated with high operation and maintenance costs. A further alternative to WCPs is the use of decentralised technologies, such as package plants and constructed wetlands (CWs), with these generally being more focused towards localised small-scale treatment of waste-water. However, integrated constructed wetlands (ICWs) can be employed at sites with a population equivalent (PE) > 500 (e.g. Clonaslee ICW, Co. Laois, which has a PE of 1200, and Stoneyford ICW, Co. Antrim, which has a PE of 950). Constructed wetlands and ICWs, for example, are artificial engineered systems that make use of the natural processes associated with wetland vegetation, soils and their related microbial assemblages to contribute to the treatment of waste-water. While CWs and ICWs have the potential for removing both N and P, the rates of removal are very much dependant on the method employed, and are perhaps best utilised in combination with other systems such as the previously mentioned physico-chemical methods.

When planning treatment options for small-scale WWTWs consideration should be given to the costs involved and to any social and environmental concerns that may arise. The selected treatment options also need to effectively deliver on P discharge consents, while requiring low/minimal maintenance, given the potential isolation of most WWTWs. Additionally, future and climate proofing of selected treatment options should be considered, especially as WWTWs are significant sources of 'priority substances' (PSs) as listed under the European Union Water Framework Directive (WFD), 'contaminants of emerging concern' (CECs) such as steroids, hormones, pesticides, and pharmaceuticals, and micro-plastics. They are also relatively high energy consumers. The potential to reclaim P should also be considered.

This report considers the pros and cons of 'alternative treatments' for WWTW effluent within the context of WCP as a treatment technology – of which the latter (in isolation or combination) remains a valid solution for the small-scale considered here.

1. Introduction

Enrichment of freshwater ecosystems is a major environmental concern, with high nutrient inputs causing eutrophication, and potentially resulting in excessive algal growth and degradation of water courses (Smith and Schindler, 2009; Mainstone and Parr, 2002). The main nutrients of concern are phosphorus (P) and nitrogen (N) with P being potentially more important in freshwater systems. These P emissions occur either via diffuse sources such as from agricultural run-off, or from point sources such as waste-water treatment works (WWTWs) or industrial discharge (Bowes et al., 2010). Combined with this, an increasing population size is likely to result in higher food demands and therefore an increase in agricultural stressors, along with an increase in the number and pressures from WWTWs. To limit the environmental degradation associated with P, effluent standards are in place to restrict the concentrations of P (and other nutrients) entering waterbodies. The Urban Waste-water Treatment Directive (UWWTD) (OJEC, 1991) for example, requires total P concentrations in waste-water effluent to be ≤ 2 mg/l P for treatment works of 10,000 – 100,000 population equivalent (PE) and 1 mg/1 P for a PE > 100,000). While the implementation of the UWWTD, along with regulation of P in detergents, has seen significant declines in concentrations of orthophosphate in rivers and lakes through-out the EU, P from waste-water still remains as a key pressure (EC, 2019).

Treatment methods at WWTWs are generally classified as being either primary, secondary or tertiary processes. Primary treatment is the first main step in the WWTW process and is designed to remove large solid objects through either flotation, settling or screening, while smaller objects are either allowed settle out by gravity and collected in grit chambers and sediment basins, or in the case of suspended sediments, are removed by clarifiers (Carey and Migliaccio, 2009). The collected "sludge" is then transported off-site for further treatment. Following primary treatment, a secondary treatment process targets additional organic matter and dissolved nutrients. By itself, secondary treatment has the potential to remove up to 50 % of P from WWTW systems either through biomass accumulation and/or partitioning of solids (Environment Agency, 2012). In order to meet regulatory guidelines, a tertiary treatment step may additionally be required to remove suspended and dissolved materials (e.g. nutrients and metals) remaining following secondary treatment (Carey and Migliaccio, 2009). The amount of P remaining in the effluent after the tertiary process is dependent on the tertiary process method employed.

Willows have several characteristics that enable their use for environmental remediation. These include having high rates of evapotranspiration, being non-edible, having a high nitrogen absorption capacity, and being able to absorb certain metals (Lachapelle-T. et al., 2019). To this end, willow coppice plantations (WCPs) have been employed for a variety of environmental treatment options including, for example, landfill leachate treatment (Aronsson, et al., 2010), polluted groundwater (Yang et al., 2019), and heavy metal extraction (Mleczek et al., 2017; Cao et al., 2018;). Their use for treating municipal waste-water is also well established (Perttu and Kowalik, 1997; Wyrwicka and Urbaniak, 2018), not least because of the similarity between willow nutrient requirements and the typical nutrient characteristics of waste-waters, i.e. N, P and K proportional requirements for willow of 100:14:72 vs N, P and K proportional occurrence in waste-waters of 100:18:65 (Perttu, 1993). Lachapelle-T. et al. (2019) reported that WCPs enabled 98 % removal of total N and total P from waste-water following irrigation, although the same study reported significant increases in soil P and suggested an eventual soil P saturation with continued irrigation. In contrast to this however, recent soil analysis from a WCP site irrigated with WWTWs discharge at Bridgend, Co. Donegal, has indicated only low levels of P build up in the soil (AFBI, communication). Nevertheless, this potential for eventual soil P saturation, along with difficulty sourcing nearby land for growing WCPs, may necessitate the identification of alternative methods to the use of WCPs for the treatment of waste-water. Furthermore, due in part to the high evapotranspiration rates, concerns have been raised relating to WCPs disrupting natural hydrological regimes and exacerbating water shortage problems to the point where in some parts of the world they are considered invasive (Frédette et al., 2019).

Table 1 - Advantages and disadvantages of physico-chemical (section 2) and EBPR methods.

Configuration	Plant Size	Cost/PE/year	CAPEX/PE	OPEX/PE/year
	(PE)	€	€	€
FS	6	302	877	211
	20	160	604	99
	49	122	517	71
	50	166	588	106
	200	140	534	84
HF	6	291	738	206
	20	141	404	94
	49	101	302	65
	50	142	379	98
	200	115	328	76
MT	6	322	645	246
	20	179	347	134
	49	137	241	105
	50	200	304	158
	200	174	257	136

Alternative methods to WCPs for removing P from wastewater include physico-chemical based treatments, biological treatments and/or a combination of both, with these being categorised based on the method employed and the part of the process at which they occur. Additionally, novel technologies such as membrane bioreactors (MBRs) and reverse osmosis (RO) systems are continuously being developed, while decentralised technologies that include package plants and the use of natural systems such as constructed wetlands are also options (Environment Agency, 2012; Macintosh et al., 2019). However, while large-scale techniques for the removal of P from waste-water are well established, issues such as variable inflows with seasonal fluctuations, management and accessibility concerns, and wastewater composition limit this practice at smaller-scales (Bunce et al., 2018). When assessing options for the removal of P from WWTW effluent therefore, these points should be considered, along with capital, operational and maintenance costs and the efficiency of P removal. With this background, the aim of this review was to highlight alternative options to the use of WCPs with a particular emphasis on small-scale WWTWs (i.e. PE < 250) and the treatment of P prior to waste-water discharge.

2. Physico-chemical treatments

Physico-chemical methods for the removal of P may occur through precipitation, sorption and/or ion exchange. Each are considered.

2.1. Precipitation

The precipitation method is an additive process that involves adding (dosing) multi-valent metal salts such as ferric chloride ($FeCl_3$) or aluminium sulphate $Al_2(SO_4)_3$ to the waste-water, with this usually resulting in a net increase in the waste-water components (Tchobanoglous et al., 2003). These salts precipitate out the P to form solid residuals that are then either filtered out of the system or extracted by allowing to settle under gravity (Bunce et al., 2018).
Sample reactions of a) aluminium and b) iron salts with P are given below:

a) $Al_2(SO_4)_3 \bullet (14H_2O) + 2H_2PO_4^- + 4HCO_3^- \rightarrow 2AlPO4 + 4CO_2 + 3SO_4^{2-} + 18H_2O$
b) $FeCl_3 \bullet (6H_2O) + H_2PO_4^- + 2HCO_3^- \rightarrow FePO_4 + 3Cl^- + 2CO_2 + 8H_2O$

Precipitation with lime ($Ca(OH)_2$) is also possible. This occurs as the lime reacts with the natural bicarbonate alkalinity of the water to form calcium carbonate ($CaCO_3$), and "as the pH of the waste-water increases beyond 10, excess calcium ions react with the P to precipitate hydroxylapatite $Ca_{10}(P0_4)_6(OH)_2$" (Tchobanoglous et al., 2003). In recent years however, precipitation with lime has fallen out of favour due to chemical handling issues and increased sludge production (USEPA, 2009).

Metal salts or lime are added either upstream of the primary treatment (pre-precipitation), at the secondary stage (co-precipitation) – (excluding lime as the activated sludge process requires a pH of below 9), or following the secondary stage in a tertiary process (post-precipitation), with multi-point additions typically removing more P than single point additions (Tchobanoglous et al., 2003; Environment Agency, 2012). Post-precipitation of P has some advantages over pre- or co-precipitation, as removal of too much P prior to the secondary stage may impact on the growth of micro-organisms present in the activated sludge which rely on P as a food source (USEPA, 2009). Additionally, during the secondary stage biological process, P entering the system as soluble orthophosphate, soluble polyphosphates, and organically bound P is converted to more simple orthophosphates that are easier to treat and result in lower effluent levels (USEPA, 2009). However, post-precipitation has the potential to have higher capital costs and require more space than adding chemicals at an earlier stage (e.g. primary stage) (Tchobanoglous et al., 2003; USEPA, 2009).

Chemical precipitation is one of the most widely used methods in the UK for the removal of P from wastewater due in part to its cost effectiveness and its reliability (Environment Agency, 2012). However, it is not without its problems and may be unsuitable for use with small-scale WWTWs. For example, chemical precipitation produces a large amount of sludge and this in turn requires treatment with a large amount of secondary chemicals (Macintosh et al., 2019). Furthermore, once chemically-bonded, extracting the P for reuse is difficult and is not cost efficient for small-scale systems, while issues relating to chemical storage and usage, sludge transport, and variability in the pH of the influent, again limit small-scale potential (Bunce et al., 2018).

2.2. Absorption

A second physico-chemical method utilises absorptive media to filter and accumulate inorganic P from waste-water by passing the effluent through the filter media which contains reactive components such as calcium or iron (Bunce et al., 2018). Typically, the reactive materials used come from either natural (e.g. limestone, orange peel, sawdust), industrial (e.g. fly-ash or steel slag) or artificial (e.g. Filtralite^{TM}) sources (Bunce et al. 2018; Macintosh et al., 2019). Reviews of variations in absorptive methods along with the advantages and dis-advantages of different absorptive materials are provided by Bacelo et al. (2020) and Loganathan et al. (2014) and summarised here. While absorptive media has not been used extensively in WWTWs, due in part to the requirement to regenerate the absorptive media following saturation, it is gaining interest because of the potential to simultaneously remove and recover P from waste-water (Loganathan et al., 2014; Bacelo et al., 2020). For certain absorptive media (e.g. bio-derived materials or materials based on non-toxic metals such as Ca and Mg) recovered P in a ready-to-use-form may be sold directly as a raw material or as fertiliser, with this also limiting environmental impacts associated with P recovery (Macintosh et al., 2019). Interest in the use of adsorption technology is also growing because it is relatively simple to apply, it has a high selectivity, the use of natural or industrial by-product materials make it a cost-effective option, and retrofitting is possible (Loganathan et al., 2014; Macintosh et al., 2019; Bacelo et al., 2020). However, the amount of P removed is dependent on the mineral content of the media, and as such this may reduce over time as more P is accumulated, while correction of pH values relating to the influence of certain media, may incur additional excessive costs (Bunce et al. 2018). Nevertheless, in combined systems, absorptive media is also particularly useful as a bed material in combination with constructed wetlands (see also the ‘Novel technologies’ section).

2.3. Ion exchange

Ion exchange is useful for removing P from waste-water due to P being primarily anionic, and may be “selected” from waste-water through the use of a metal cation used in combination with P-selective nano-particles such as ferric oxide (Loganathan et al., 2014; Martin et al., 2009). Due to similarities in processes, some studies (e.g. Loganathan et al., 2014) group ion exchange in the absorption category. However, while ion exchange has been used extensively for purification and separation processes such as desalinisation and deionisation of water (Awual and Jyo, 2011), its use for P extraction or use with WWTWs is not as well established as compared with other physico-chemical methods (Bunce et al. 2018). Additionally, it may be unsuitable for small-scale WWTWs due to chemical requirements and associated costs (Bunce et al., 2018). Nevertheless, research aimed at improving P selectivity with respect to competing ions (Zhao and Sengupta, 1998; Awual and Jyo, 2011), and applying it to alternative waste-water management approaches such as urine separation (Sendrowski and Boyer, 2013) may lead to reduced costs and future potential.

Bunce et al. (2018) summarises physico-chemical methods as providing reliable P extraction with a relatively simple operation, but sustainability concerns limit their usefulness in relation to small-scale WWTWs, and they perhaps best serve as a "polishing" treatment option in combination with other treatments such as constructed wetlands. The ability of treated effluent to comply with Environmental Quality Standard regulations also needs to be considered. For example, as iron and aluminium are classified as a "specific pollutant" to surface waters and toxic to fish, respectively, (along with both being classified as a non-hazardous pollutant for groundwater), the Environment Agency (UK) has set limits on the amount of iron and aluminium allowable in discharge water following chemical dosing treatment (Environment Agency, 2018). Limits on pH levels are also applicable. Furthermore, the chemical treatment of wastewater is required to comply with the European/British standards (e.g. BS EN 12255) that specifies, for example, the criteria for chemical storage and design requirements.

3. Biological treatments

Phosphorus may be removed from waste-water effluent by biological processes such as Enhanced Biological Phosphorus Removal (EBPR). This process relies upon polyphosphate-accumulating organisms (PAOs), primarily the bacteria *Candidatus Accumulibacter phosphatis* (Accumulibacter), that accumulate P as polyphosphate which is then used as a “luxury” energy reserve (Oehmen et al., 2007; Nielsen et al., 2012). Other organisms, such as *Acintobacteria* sp., *Pseudomonas* sp., *Paracoccus* sp., and some *Enterobacter* sp., are suspected as behaving as PAOs, although to date most research has focused on Accumulibacter (Bunce et al., 2018). Additionally, some PAOs have the potential to accumulate nitrate (Denitrifying PAOs), which potentially reduce energy demands by conducting two cleaning services (P and N) in combination (Oehmen et al., 2007; López-Vázquez et al., 2008; Bunce et al., 2018). Conventional activated sludge (CAS) systems, which are composed of an aeration tank where biological degradation takes place, along with a sedimentation/settling tank where waste sludge and treated waste-water are separated, have traditionally employed EBPR techniques (Bunce et al. 2018). In an EBPR reactor system set-up, an anaerobic tank is placed ahead of the activated-sludge aeration tank, with this lay-out providing PAOs with a competitive advantage over other bacteria (Tchobanoglous et al., 2003) (see Figure 1). Phosphorus is removed from the system via the removal of PAOs in the waste activated sludge (Oehmen et al., 2007), although a portion of the waste activated sludge (with PAOs) is returned to the system to reseed the anaerobic reactor.

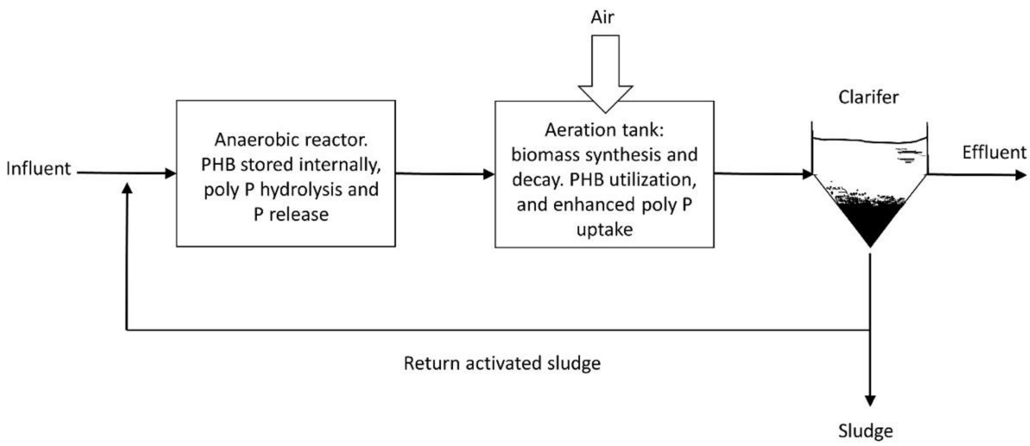

Figure 1 - Biological P removal (from Tchobanoglous et al., 2003).

In comparison to chemical production, EBPR has reduced chemical costs and produces less sludge (Environment Agency, 2012). However, P extraction results may be variable, with fluctuations in performance dependant on operational factors, local environmental conditions (e.g. heavy rainfall), competition from other non-P consuming micro-organisms, excessive nitrate loading and P-starvation (e.g. if pre-precipitation is employed) (Oehmen et al., 2007; USEPA, 2009). These factors, along with the requirement for some, if minimal, operator supervision, may limit the usefulness of EBPR systems in rural small-scale systems (Brown and Shilton, 2014). However, recent developments have seen EBPR techniques integrated into membrane bioreactors (MBRs), granular sludge reactors and sequencing batch biofilm reactors (SBRs), with these developments potentially being more attractive for small-scale WWTW operations due to improved P removal rates and a reduced physical footprint (Bunce et al., 2018).

A summary of advantages and disadvantages of physico-chemical (section 2) and EBPR methods are presented in Table 1

Table 1 - Advantages and disadvantages of the physico-chemical methods – chemical precipitation, absorption and ion-exchange; and the biological method Enhanced Biological Phosphorus Removal (EBPR) for removing P from waste-water (from Loganathan et al., 2014; Bacelo et al., 2020).

Method		Advantages	Disadvantages
Physico-chemical methods			
	Chemical Precipitation - iron, aluminium salts, lime	Widely used, reliable and well-established; Cost effective; Consistently high P-removal performance	Large amount of sludge produced; Secondary chemicals; Costs of chemicals, chemical storage and feeding system; P-reuse difficult; pH influence on stream-water; Chemicals used may affect subsequent biological treatment
	Absorption	Easy operation and low costs; Possible use of low-cost adsorbents; Selectivity and effectiveness for low concentrations; Fast adsorption rate; Possible phosphate recovery; Useful in combination with other methods, e.g. CWs	P-removal performance media dependant; pH influence on stream-water
	Ion exchange	Effective even at low P concentration; Flexible; Simplicity of design; Ease of operation; Potentially no waste production	Less widely used than other chemical methods; May be costly, needs pre-treatment, sorbent regeneration; can use low-cost sorbent (including certain waste materials); Low selectivity against competing ions; Chemical usage and costs may restrict usage
Biological method			
	Enhanced Biological Phosphorus Removal (EBPR)	Possible phosphate recovery for fertilizer use etc.; Chemicals not required; Produces less sludge; Simultaneous removal of both excess P and N	Operator supervision required; Competition from other biological organisms within the system may limit the overall performance - Uncontrolled microbial growth; Sensitivity to inhibiting substances, e.g. ammonia, pH, p-stravation; Low effectiveness, especially at low P concentration; Low operational cost but infrastructural investment required; Highly skilled operation (strict anaerobic and aerobic conditions required).

4. Novel technologies

Recent advances in methods for the removal of P from waste-water include the use of algal biofilm systems, membrane bioreactors and developments in the use of active filter media (Bunce et al., 2018). Additionally, the Environment Agency (2012) discusses the use of reverse osmosis (RO), Blue PROTM process, fuzzy filters ®, Hydrotech Discfilter ® and Virtec's Bauxsol ™. These methods are reviewed here.

4.1. Algal biofilms

Several studies have highlighted the usefulness of algae to remove nutrients from waste-water (e.g. Martínez et al., 2000; Shi et al., 2007; Boelee et al., 2011), although its large-scale application to date has been limited (Pittman et al., 2011). However, its potential use as a biofuel with waste-water acting as a sustainable nutrient supply has increased interest in its use (Christenson and Sims, 2011; Pittman et al., 2011; Kesaano and Sims, 2014). Uptake of P is required for algal growth, with P typically making up 1 % of algal dry weight (Brown and Shilton, 2014). As with the bacteria Accumulibacter however, in certain conditions microalgae, primarily the species *Scenedesmus* sp. and *Chlorella* sp., remove P from waste-water. This is similarly accumulated as "luxury" P in excess of normal growth requirements, and stored as polyphosphate granules to be used as a growth reserve during periods where P is lacking in the environment (Brown and Shilton, 2014). The microalgae uptake of P is either as orthophosphate or organic P that is then converted to orthophosphate via the phosphatase enzyme (Bunce et al., 2018). Systems that use algae include algal biofilm systems (Wei et al., 2008; Boelee et al., 2011) and algal membrane bio-reactors (MBRs) (Kumar et al., 2020).

Algal systems are potentially suitable for small-scale treatment works due to their resilience to changes in environmental conditions and the recoverable biomass (Bunce et al., 2018). Additionally, in comparison to conventional chemical systems, algal systems have lower associated costs and require less technological investment, while being more environmentally acceptable and sustainable due in part to reduced by-products such as sludge (Pittman et al., 2011). However, algal systems face challenges particularly regarding achieving optimum algal growth rates, and as of yet, many methods are only at the lab or pilot scale (Kesaano and Sims, 2014), although advances are progressing rapidly (Kumar et al., 2020).

4.2. Membrane bioreactors

Membrane bioreactors (MBRs) are designed to combine the activated sludge process with a solid-liquid membrane separation process that halts the movement of bacterial flocs and all suspended solids across the membrane, but that allows clean "treated" water through (Le-Clech et al., 2006; Environment Agency, 2012). The MBRs may be set up either as an external/side-stream system, whereby the membrane is outside of the activated sludge tank, or as an internal/submerged system where the membrane is within the activated sludge tank (see Figure 2), with the latter being the more commercially significant option (Judd, 2006). Membrane bioreactors are increasingly being employed in waste-water systems to the point where they may no long be considered a novelty, due in part to MBRs having a small footprint and reactor requirements, the effluent is of high quality and there is less sludge produced compared to other WWTW systems (Le-Clech et al., 2006; Judd, 2006). While they have traditionally been employed for the separation of bacterial flocs, they are also now being increasingly utilised for algal separation and recovery (Kumar et al., 2020). MBR systems may be employed at small-scale WWTWs, however, problems with fouling and an associated high maintenance cost (Le-Clech et al., 2006; Meng et al., 2009), along with high initial capital costs (Environment Agency, 2012), may restrict their current suitability, although this may change with the advancement of technology and potentially reduced costs (Bunce et al., 2018).

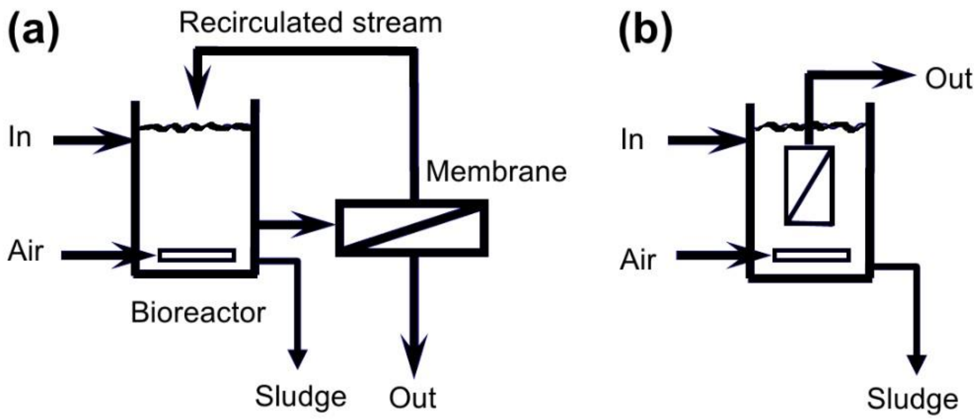

Figure 2 - Configurations of membrane bioreactor: (a) sidestream and (b) immersed (Judd, 2006).

4.3. Active filter media – including Blue PRO™ and Bauxsol™

As previously mentioned, there is a gaining interest in the use of absorbents and active filter media, especially because of the potential for cheap P removal and recovery, and as such novelties in this area are continuously been reported/published (Bacelo et al., 2020). These novelties include, for example, the use of biochar derived from sugar beet tailings (Yao et al., 2011), wood, corn and rice husk biochar (Kizito et al., 2017), and biochars from wood chips, manure and orange peel that have been modified or dosed with, for example, Ca, Fe, Mg or Al (Chen et al., 2011; Micháleková-Richveisová et al., 2017; Novais et al., 2018). However, many of these methods are only at the laboratory/trial stage (see Bacelo et al., 2020). Examples of commercially available active filter systems include Blue PRO™ and Bauxsol ™.

The Blue PRO™ reactive filtration system is a tertiary process used to remove P from waste-water by combining "co-precipitation and adsorption to a reactive filter media in an upflow sand filter" (USEPA, 2013). Phosphorus is removed by co-precipitation and adsorption as it passes through the sand filter media as the sand is coated with reactive hydrous ferric oxide coating (USEPA, 2013). Following adsorption the iron and P are abraded from the sand (USEPA, 2013). The media does not need to be changed as the ferric oxide coating is continually formed on the sand media, abraded and regenerated (Environment Agency, 2012). High P removal is possible with the Blue PRO™ system, there is a high capacity to retrofit, and capital, operational and maintenance costs are considered moderate (Environment Agency, 2012). It is considered most suitable for small to medium WWTW plants and in Ireland and the UK is available via Evergreen Water Solutions[1].

Bauxsol ™ is a red mud produced during the refinement of alumina from bauxite that has been geochemically and physically modified and trademarked by Virotec International Ltd (Akhurst et al., 2006). Natural Bauxsol is high in Ca^{2+}, Mg^{2+} and Al^{3+} ions and therefore it is useful to adsorb and remove P from solutions (Akhurst et al., 2006; Despland et al., 2011, 2014). Virotec's Bauxsol ™ products have the potential to remove TP to less than 2 mg/l, there is a high capacity to retrofit, capital costs are moderate, and operational and maintenance costs are considered low to moderate (Environment Agency, 2012). Bauxsol ™ in pellet form has the potential to be utilised in construction wetlands for P removal (Despland et al., 2014).

Alternative absorption methods that are capable of being utilised in construction wetlands are also being assessed, e.g. Li et al. (2019).

4.4. Reverse Osmosis (RO), Compressible Media Filtration (CMF) and Hydrotech Discfilter®

Reverse Osmosis (RO) systems rely on pressure being applied to a solution that is on one side of a selective membrane, so that the solution moves across the membrane to a second solution of lower solute concentration and as it moves across the membrane, filtering of the first solution takes place (Tchobanoglous et al., 2003: Environment Agency, 2012). While it is possible to achieve high P removal and there is high capacity for retrofitting at a tertiary stage, the capital costs and operational and maintenance costs are high (Environment Agency, 2012).

Compressible Media Filtration (CMF) (e.g. Fuzzy Filters ®) is a tertiary process that uses highly porous filter media that is compressible so that the media porosity may be adjusted to suit the characteristics of the influent (Schreiber, 2020). Similar products include the WWETCO FlexFilter™ and Bio-FlexFilter™ (USEPA, 2013). According to the developer (Schreiber) P removal of below 0.1 mg/l is possible, it has a high retrofitting capacity and low operating costs (Schreiber, 2020), although the Environment Agency (2012) classify the operating and maintenance costs as high with and associated high pumping energy cost.

Hydrotech Discfilter® is a compact design filter system that micro-screens waste-water at the secondary or tertiary stage, removing small particles and P (Veolia, no year; Environment Agency, 2012). The design, based on woven cloth filters mounted on multiple disks, provides a filter area two to three times greater than conventional methods (Veolia, no year). Annual average total P removal of 0.3 mg/l is possible with the Hydrotech Discfilter ® system, there is a high capacity to retrofit, however operational and maintenance costs are considered high, again related to high pumping energy costs (Environment Agency, 2012).

[1] http://www.evergreenengineering.ie/bluepro.php

5. Decentralised technologies

Decentralised technologies are technologies that are more focused towards localised small-scale treatment of waste-water, specifically as they have reduced infrastructure and are associated with more minimal environmental impacts (Environment Agency, 2012). They include package plants, modified package sand filters and natural systems such as constructed wetlands and integrated constructed wetlands and are reviewed here.

5.1. Package plants

Packaged plants, in comparison to traditional on-site constructed WWTW, are complete units that are pre-manufactured and shipped to a location for direct installation (Fletcher et al., 2007). The main processes associated with these packaged plants are "submerged aerated filter (SAF), conventional activated sludge (CAS), rotating biological contactor (RBCs), sequencing batch reactor (SBR), trickling filter (TF) and biological activated filter (BAF)" (Fletcher et al., 2007). For example with a SBR set-up, which is based on an activated sludge process where all major steps occur in sequential order, the components of the plant such as the batch tank, the aerator, mixer, the decanter device, pumps are all included in the package (USEPA, 2009). Phosphorus removal is typically low for these plants, e.g. 10-15 % in trickling filter package plants (Environment Agency, 2012), which may therefore necessitate the addition of a further polishing step (e.g. chemical precipitation). Additionally, in comparison to bespoke on site designs, designs for package plants are generally limited due to manufacturing costs and mass production requirements (Fletcher et al., 2007). However, capital and operational costs are generally considered low, with these costs primarily depending on the size and shape of package plant employed, i.e. population equivalent (PE) value (Environment Agency, 2012). Capital costs for package plants include the tank, installation, membranes, pumps, air blowers and diffusers, screens and timer switches, along with customer training as required, while operational costs are associated with power consumption, maintenance, de-sludging and chemicals (Fletcher et al., 2007). Details of the annual cost per person (note that these will require updating) and the absolute costs and power requirements for three package plant system designs include: 1) Membrane-aerated flat sheet immersible MBR (FS); 2) Membrane-aerated hollow fibre immersible MBR (HF); and 3) Pumped multi-tubed submerged MBR or (MT); for different plant sizes, are presented in Tables 2 and 3. The average per capita capital and annual electricity costs for single house and small decentralised packaged treatment systems in Ireland are presented in Table 4.

Table 2 - Annual cost per person at three different plant sizes for 1) Membrane-aerated flat sheet immersible MBR (FS); 2) Membrane-aerated hollow fibre immersible MBR (HF); and 3) Pumped multi-tubed submerged MBR or (MT) (Fletcher et al., 2007).

Configuration	Plant Size	Cost/PE/year	CAPEX/PE	OPEX/PE/year
	(PE)	€	€	€
FS	6	302	877	211
	20	160	604	99
	49	122	517	71
	50	166	588	106
	200	140	534	84
HF	6	291	738	206
	20	141	404	94
	49	101	302	65
	50	142	379	98
	200	115	328	76
MT	6	322	645	246
	20	179	347	134
	49	137	241	105
	50	200	304	158
	200	174	257	136

Table 3 - Absolute costs and power requirements for: 1) Membrane-aerated flat sheet immersible MBR (FS); 2) Membrane-aerated hollow fibre immersible MBR (HF); and 3) Pumped multi-tubed submerged MBR or (MT) (Fletcher et al., 2007).

Plant size (PE)	Plant type	Power cost	Plant Capital cost
		€	€
6	FS iMBR	178	5262
	HF iMBR	143	4431
	MT sMBR	383	3870
20	FS iMBR	592	12086
	HF iMBR	477	8088
	MT sMBR	1276	6933
50	FS iMBR	2221	29378
	HF iMBR	1788	18947
	MT sMBR	4787	15211
100	FS iMBR	4442	55425
	HF iMBR	3578	34307
	MT sMBR	9576	27054
200	FS iMBR	8884	106860
	HF iMBR	7161	65625
	MT sMBR	19152	51332

Table 4 - Average per capita capital and annual electricity costs for single house and small decentralised packaged treatment systems in Ireland (Dubber and Gill, 2014).

Treatment systems	Single house system costs [1] [€/ca]		Small decentralised system costs [2] [€/ca]	
	Capital	**Operational**	**Capital**	**Operational**
MBR	1800–2000	50–70	600–1200	<30
MBBR	1500	20–30	600–800	<10
Filter media	840–1200	0–5	350–700	0–5
SBR	620–900	4–7	300–500	4–7
SAF	475–840	20–30	150–250	<18
CAS	540–600	20–30	250–450	<15
RBC	n/a	16	420–600	<5

MBR = Membrane Bioreactor; MBBR = Moving Bed Bioreactor; SBR = Sequencing Batch Reactor; SAF = Submerged Aerated Filter; CAS = Conventional Activated Sludge; RBC = Rotating Biological Contactor; 1—based on a single house system serving 3 to 6 inhabitants; 2—serving small communities ≥ 20 PE.

In Northern Ireland, for example, RBC package plants are regularly being used by Northern Ireland Water to replace or improve the wastewater treatment processes (Jacopa, 2018), with this primarily occurring under the Rural Wastewater Investment Programme (2008-2021). This program, which was set up to address and refurbish some of Northern Irelands 900 small-scale WWTWs is predicted to have seen an investment of £47 million by 2021, having completed 160 projects by September 2019 (NI Water, 2019). In this instance, costs were reduced in part due to the use of a standardised RBC package plant design, where units differed only by size and in response to variation in populations between locations (WIJ, 2010).

5.2. Modified package sand filters

Sand filters may be used to provide advanced secondary treatment of waste-water and typically consist of aerobic beds of sands or other granular materials contained within an impermeable structure (e.g. lined with PVC) with an underlying drainage system (USEPA, 2009). Partially treated waste-water is passed over the surface of the sand and, as it percolates through to the drainage system, it is filtered to remove pollutants (USEPA, 2009). A pre-treatment step is required to remove solids, etc., to prevent the filter material from clogging. Filtering within sand filters works either by microorganism bio-slimes that grow on the sand particle surfaces accumulating waste materials, or by chemical adsorption processes, with this being dependant on the type of material employed (USEPA, 2009). Sands rich in Fe or high-Al muds for example are useful for P removal (Environment Agency, 2012). Modifications of sand filter systems include for example recirculating sand filters (RSF) whereby the effluent that has percolated through the filter media is recycled back through the media for further cleansing. Sand filter systems are suitable for small WWTWs because of low capital and operation and maintenance costs, but there is a requirement to maintain aerobic conditions or risk the reduction of Fe (III) by bacteria and the potential subsequent release of stored

P (Environment Agency, 2012). Sand filter systems are particularly suitable in combination with constructed wetlands (see below).

5.3 Constructed Wetlands and Integrated Constructed Wetlands

A further alternative to WCPs is the use of constructed wetlands (CWs). These are artificially engineered systems that make use of the natural processes associated with wetland vegetation, soils and their related microbial assemblages to contribute to the treatment of waste-water (Scholz et al., 2007a; Vymazal, 2007). Their construction is primarily categorised into either surface or subsurface flow systems (Dotro et al., 2017), although their construction may be further classified based on the type of macrophytic growth occurring (free-floating, floating leaved, emergent or submerged plants) and in the case of subsurface systems, whether the flow of water moves horizontally or vertically (Vymazal, 2007; Gorito et al., 2017). Thus CWs may be categorised as "surface flow constructed wetlands (SFCWs), horizontal subsurface flow constructed wetlands (HSSF-CWs) and vertical subsurface flow constructed wetlands (VSSF-CWs)" (Gorito et al., 2017) (see Figure 3), with these potentially being hybridised to provide the most effective end-result combination.

Surface flow constructed wetlands (SFCWs) (or free water surface (FWS)) are designed to maintain a shallow horizontal flow of waste-water above a soil/media surface, and are similar in design to natural wetlands (USEPA, 1993; Healy et al., 2007). Pollution removal mechanisms in SFCWs vary depending on the wetland layout, but include physical sedimentation, plant uptake and chemical, microbial and photo degradation (Dotro et al., 2017). Typical plants associated with SFCWs include the emergent taxa *Typha*, *Phragmites* and *Scirpus*, submerged taxa such as *Potamogeon* and *Elodea* and floating taxa such as *Eichornia*, *Lemna* (Dotro et al., 2017; see also DEHLG (2010) for a list of recommended taxa in Ireland). However, SFCWs require a large area and may be more important for their aesthetic and habitat values rather than their potential to improve water quality (Dotro et al., 2017 – but see integrated construction wetlands below).

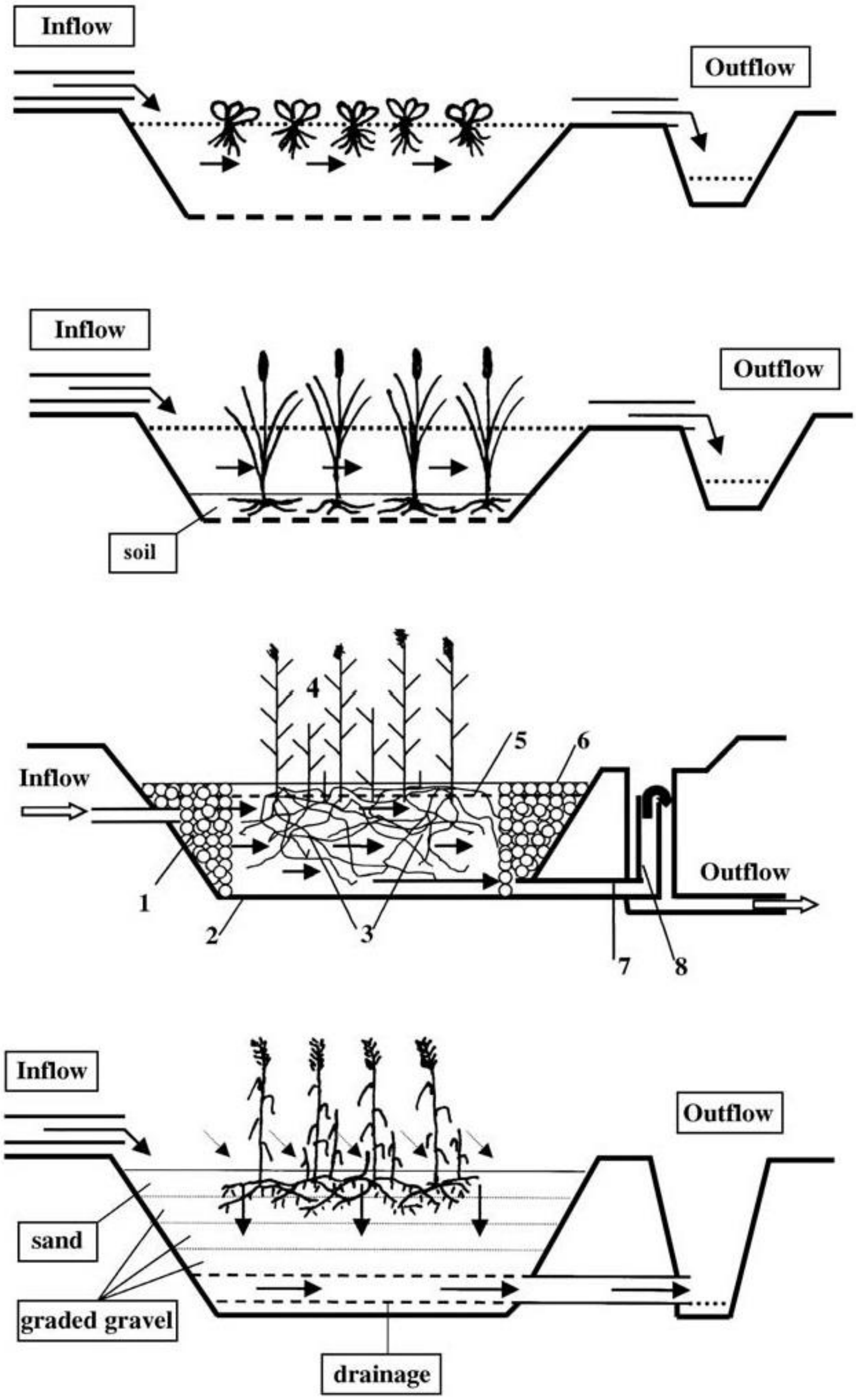

Figure 3 - Constructed wetlands for wastewater treatment (from top to bottom): CW with free-floating plants (FFP); CW with free water surface and emergent macrophytes (FWS); CW with horizontal sub-surface flow (HSSF, HF); CW with vertical sub-surface flow (VSSF, VF) (Vymazal, 2007).

In contrast to surface flow, subsurface flow CWs are designed to create a flow system that passes the waste-water through a permeable medium (typically soil, sand or gravel) that filters/treats the waste-water as it moves through (USEPA, 1993; Healy et al., 2007). In HSSF-CWs systems the waste-water flow is designed to pass horizontally through the filter media, whereas in VSSF-CWs the waste-water passes vertically through the filter media (see Figure 3). Keeping the waste-water below the media surface helps to reduce odours and other associated problems (USEPA, 1993), although the use of a dense plant cover and maintaining constant flow in SFCWs may also facilitate this (Carty et al., 2008).

Integrated constructed wetlands (ICWs) are a series of unlined interconnected SFCWs (e.g. Figure 4) that simultaneously offer water cleaning services while helping to enhance the ecological status and aquatic biodiversity of their associated habitat (Becerra-Jurado et al., 2014; Cooper et al., 2020). The design of an ICW typically consists of a chain of 4-7 similarly sized shallow ponds, with the first pond in the series being fed/pumped waste-water for treatment (DEHLG, 2010; VESI, 2018). This first pond, which acts as a settlement pond and is characterised by high emergent vegetation cover, is where primary organic matter decomposition and sediment accumulation occurs (Becerra-Jurado et al., 2014). Discharge from the shallowest first pond (due in part to sediment accumulation) flows by pipework and/or gravity to subsequent deeper ponds (and as a consequence the vegetation types occurring change - i.e. the last pond in the series is characterised by submerged vegetation and open water areas), and into the associated stream or river (Becerra-Jurado et al. 2014; VESI, 2018).

In Ireland their use was initially developed and promoted by the National Parks and Wildlife Service as a means to combine water management with nature conservation (Scholz et al., 2007). They are now present through-out the country serving as treatment options for farmyard run-off, domestic waste-water effluent, industrial effluent from mining and food processing plants, and for treating landfill leachate (Carty et al., 2008; Harrington and McInnes, 2009; Becerra-Jurado et al., 2014; Hickey et al., 2018; VESI, 2018). Examples of ICWs developed in Ireland by the environmental company VESI Environmental Ltd. (https://www.vesienviro.com/) include Clonaslee ICW in Co. Laois (WWTWs PE 1200), Rossbeigh ICW, Co. Kerry (WWTWs PE 136), Dungarvan Landfill ICW, Co. Waterford, and in Northern Ireland Stoneyford ICW, Co. Antrim (WWTWs PE 950). The capital costs of three ICWs receiving discharge from WWTWs in Norfolk, UK, although not including land acquisition costs, and as assessed by van Biervliet et al. (2020) and Cooper et al. (2020) are presented in Tables 5 and 6, respectively.

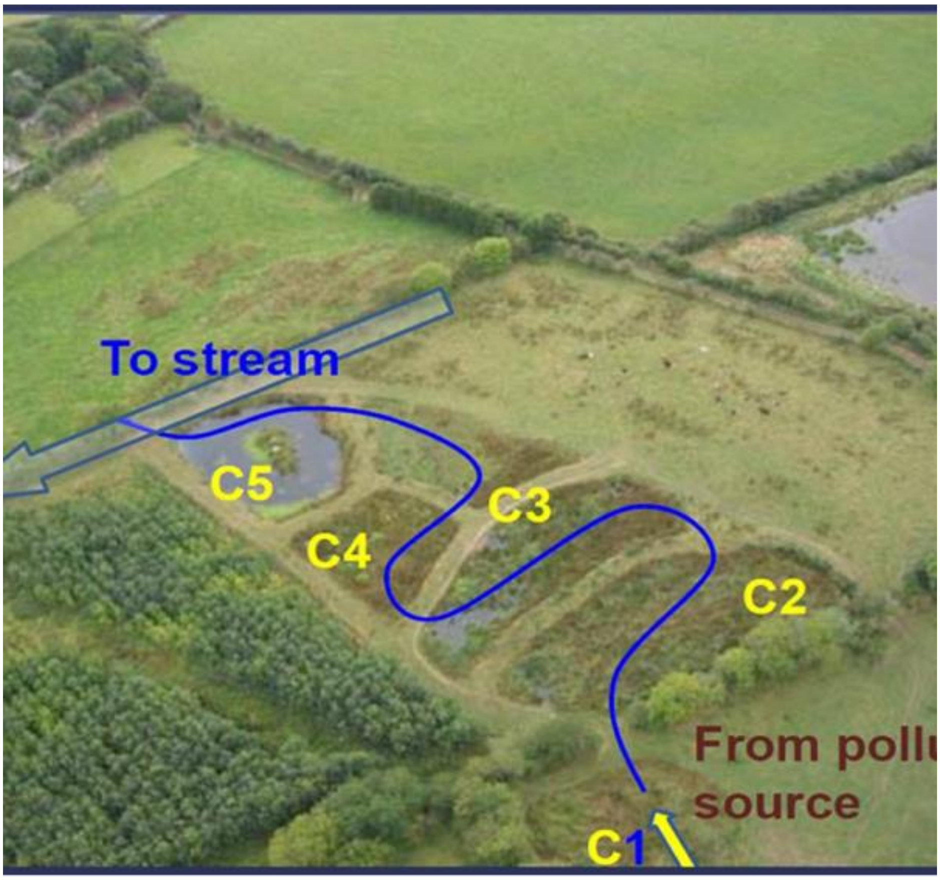

Figure 4 - Example of an integrated constructed wetland layout – Donegal County Council communication.

Table 5 - Capital costs of Frogshall integrated constructed wetland ICW (adapted from van Biervliet et al., 2020).

Item	Cost Including VAT (£)
Site Works	
Surveys, applications for permissions	£1,305
Earthworks, pipework and site supervision	£21,712
Wetland Plants	£7,004
Monitoring	
Water quality analysis	£6,900
Stream invertebrate analysis	£4,000
Population served	553
Total	£40,920

Table 6 - Capital costs for the construction of the River Ingol and River Mun ICW (Cooper et al., 2020).

Parameter	River Ingol	River Mun
Planning, design & management	£15,000	£1,305
Construction	£161,000	£21,712
Wetland planting	£18,000	£7,004
Population served	6238	772
Total cost	£194,000	£30,021
Cost per person	£31	£39

While CWs and ICWs are considered effective for reducing biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids and faecal coliforms, their potential to remove N and P is much more variable (Toet et al., 2005; Hickey et al., 2018). For example, (Healy and Cawley, 2002) assessed the nutrient processing capacity of a CW fed with tertiary effluent in Western Ireland and found the percentage reduction of total N and total P to be 51 % and 13 %, respectively. In contrast, other studies have reported much more favourably N and P removal rates (e.g. Scholz et al., 2007b; Dzakpasu et al., 2015; van Biervliet et al., 2020). For example, in a case study of thirteen ICWs used to treat farmyard dirty water from farms located in the Annestown-Dunhill catchment in Co. Waterford, Ireland, Scholz et al. (2007b) reported general P concentration reductions of greater than 90 %, and while there was one exception to this (30 % reduction in P), this was likely related to high loading rates. Similarly, van Biervliet et al. (2020) reported reductions of 78 % and 80 % respectively, in the concentrations of total and dissolved P occurring between the influent and effluent of an ICW receiving discharge from a small WWTWs in Norfolk. The same study however, reported only non-significant reductions in nutrient concentrations in the receiving stream.

Phosphorus removal in CWs and ICWs may occur through chemical precipitation, sedimentation, sorption and plant and microbial uptake (Vymazal 2007; Dotro et al., 2017). However, these processes are slow and limited unless, for example, an absorptive media such as modified sand or, for example, Bauxsol ™ is used as a bedding substrate, while P uptake by plants, which is stored in their biomass, is initially only removed if harvesting occurs (Dotro et al., 2017). Furthermore, P accumulated by algae and microbes is stored for only a short-term period, with up to 75 % eventually being released back into the water following dieback (Healy and Cawley, 2002). This loss of P efficiency may be reduced however by using appropriate designs and ensuring anaerobic conditions are maintained (Harrington and McInnes, 2009), although this latter point seems counter-intuitive. However, peat/soil accretion is a major long-term P sink in wetlands, and while the peat formation process is very slow and dependant on the levels of Fe, Ca, Al and organic material present in the substrate (Healy and Cawley, 2002; Vymazal, 2007), accumulations of 2-4 cm/yr are possible (Scholz et al., 2007; Harrington and McInnes, 2009). Additionally, desludging of, for example, initial cells in the ICW chain, which may typically be required every 10 to 15 years, may yield (and potentially return) between 20 - 50 % of the annual P requirement of a typical farm

(Scholz et al., 2007). Nitrogen removal from CWs may occur through, volatilization, denitrification, plant uptake, ammonia adsorption, anaerobic ammonia oxidation (ANAMMOX) and organic nitrogen burial, although the rate and occurrence of these processes is very much dependent on the type of constructed wetland employed – e.g. VSSF-CWs remove higher levels of ammonia-N than FWS and HSSF wetlands but lesser nitrates (Vymazal 2007).

The nutrient removal efficiency of CWs and ICWs is partially temperature dependant and may therefore vary in response to changing seasons (Healy and Cawley, 2002; Healy et al., 2007; Hickey et al., 2018). Similarly, the highest P uptake by plants occurs early in the growing season (Vymazal 2007). As previously mentioned, the design, size and loading rates are also important considerations (e.g. Scholz et al., 2007b). For example, in an assessment of the performance of 52 Irish municipal constructed wetlands, that included HSSF, Free-SFCW and Hybrid systems, Hickey et al. (2018) reported that ICWs achieved the lowest nutrient concentrations in effluents, although this was only if the design of the ICW followed specified guidelines (e.g. DEHLG, 2010). A further consideration relating to CWs and ICWs is their long-term P retention ability. Dzakpasu et al. (2015) found total P and dissolved P effluent concentrations increased only marginally over a four year period in an assessment of a WWTWs fed ICW located in Glaslough, County Monaghan. Similarly, Cooper et al. (2020) reported high P removal efficiency five years after the instalment of an ICW on the River Mun wetland in Norfolk, U.K. However, this is something that probably necessitates more intensive study, given the relatively short time frame in which these studies were carried out (i.e. four and five years), and the relatively recent wide-scale use of ICWs.

The number of constructed wetlands in operation in Ireland to treat municipal, industrial and agricultural wastewater is estimated to be over 140 (EPA Catchments Unit, 2018). A database (although incomplete) containing information on over 100 of these Irish construction wetland sites is available at http://wetlands.nuigalway.ie (link broken in 2023). In the UK, information on more than 900 CW sites is maintained on a database hosted by the Constructed Wetland Association (https://www.constructedwetland.co.uk/ - although membership may be required to access this database) (Cooper, 2007). The Environmental Protection Agency in Ireland (EPA, 2010) provide guidelines for the design of small scale CWs, such as the area required, the loading rates and the length/width ratio, although these are specific to single household treatment systems (PE ≤ 10). Guidelines for a PE of up to 50 are provided in CEN/TR 12566 part 5 (BS 12566, 2008). Guidance for the assessment, design and construction of ICW systems in Ireland is provided by the Department of the Environment, Heritage and Local Government (see DEHLG, 2010), and parallels similar guidance on developing ICWs in Scotland and Northern Ireland (see Carty et al., 2007) and in Finland (see Puustinen et al., 2007). The Irish guidelines stipulate, for example, that the aria required for a domestic wastewater ICW should follow the equation:

Area Required (m^2) = (Population Equivalent x 20 to 40**) x 1.25*

Where:
*Supporting infrastructure = Area taken up by embankments and associated access.
** 20 m^2/P.E. for wastewater free of storm water increasing to 40 m^2/P.E. when storm water is also included.

Constructed wetlands and ICWs are considered suitable for small WWTWs because they are a low-cost sustainable treatment method that have low maintenance and operation costs (Environment agency, 2012; van Biervliet et al., 2020; Cooper et al., 2020) and subtly fit into the environment (e.g. Figure 4). Additionally, in comparison to the previously mentioned methods, constructed wetlands also have the potential to serve as and restore wildlife habits, provide flood relief, and they serve as public amenity and educational areas (Becerra-Jurado et al., 2014; Mackenzie and McIlwraith, 2015; van Biervliet et al., 2020) (e.g. Dunhill Integrated Constructed Wetland in Co. Waterford https://www.water.ie/wastewater/wetlands/dunhill/). In the Mississippi delta, for example, increased vegetative growth as a result of the input of secondary treated effluent has resulted in an accretion of organic soil which in turn has off-set subsidence and potential impacts from sea-level rise (Day et al., 2006). In Ireland and the UK studies by Becerra-Jurado et al. (2014) and van Biervliet et al. (2020) have highlighted their benefits for aquatic invertebrate taxa and bird species, respectively. However, the potential poor nutrient performance may see CWs best serve as a polishing step following secondary or tertiary treatment, although this may be addressed if, for example, CWs designs are modified to incorporate other P removal methods, such as the use of a reactive media as a substrate layer or plant harvesting, or in the case of ICWs, guidelines such as those outlined by Carty et al. (2007) and DEHLG (2010) are adhered to (e.g. Hickey et al., 2018).

5.3.1. Zero-discharge wetland systems

Modifications to CWs include for example “zero-discharge wetland systems” where, instead of having a discharge, the effluent is designed to be removed from the system by plant (e.g. *Salix sp*. – Willow) evapotranspiration (Dotro et al., 2017). These systems essentially combine CWs with WCPs. Examples of a zero-discharge wetland system designs for single households (PE = 5) employed in Denmark and Ireland are presented in Figures 5 and 6, respectively.

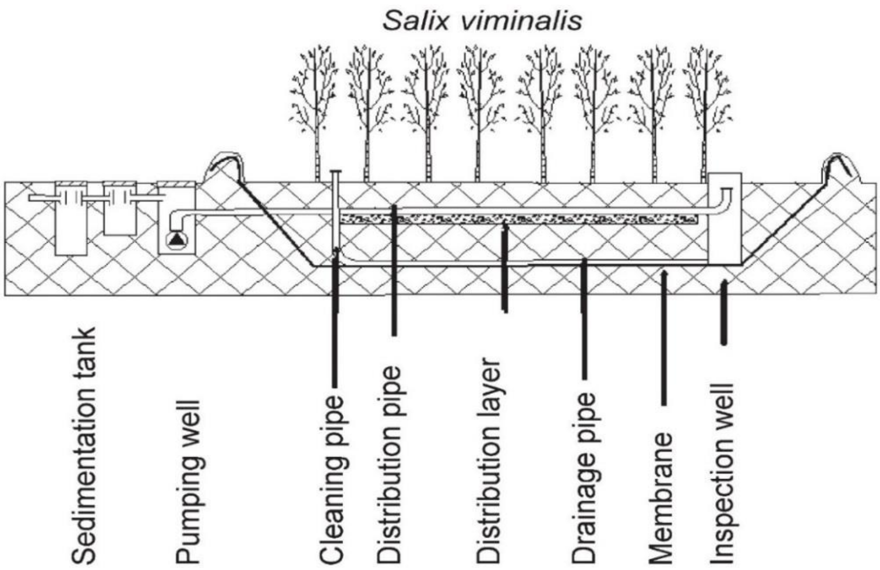

Figure 5 - Sketch of a willow system with no outflow (evaporative system) (Brix and Arias, 2005).

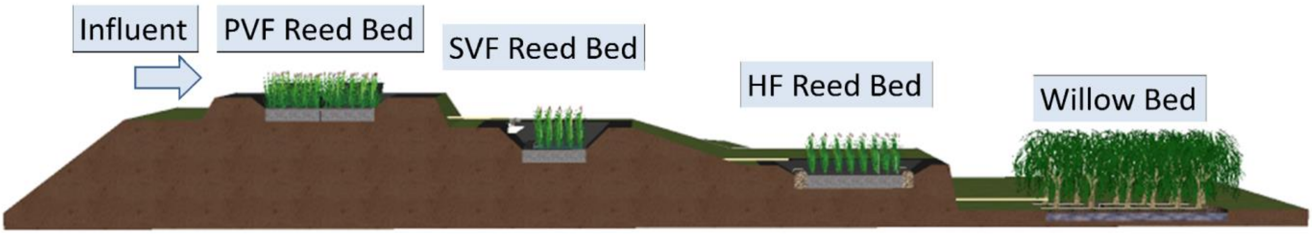

Figure 6 - Longitudinal section of a zero-discharge system design at Lynches Lane, Co. Dublin Ireland (O'Hogain et al., 2011).

5.4. Alternative plant species

While willow (*Salix* sp.) are perhaps the plants of choice where plant harvesting is employed, alternative plant options are also available. Rockwood et al. (2004) for example, lists poplar (*Populus spp*.), willow (*Salix* spp.), and black locust (*Robinia pseudoacacia*) as the main temperate region short rotation woody coppice plants used for phytoremediation. In a mesocosm study, Grebenshchykova et al. (2019) assessed the potential of five woody taxa (*Salix interior, Salix miyabeana, Sambucus canadensis, Myrica gale, Acer saccharinum*) to be used as "treatment wetland" plants. The growth characteristics, evapotranspiration rates and pollutant removal efficiency of these woody plants were compared with that of four herbaceous taxa (*Typha angustifolia, Phragmites australis australis, Phragmites australis americanus, Phalaris arundinacea*) that are typically associated with treatment wetlands. The study found that, following the addition of a synthetic sewage solution, all plants, with the exception of *A. saccharinum*, displayed healthy grow rates, although this was higher for the herbaceous plants than for the woody taxa. Similarly, the herbaceous taxa were shown to remove higher levels of total P and total Kjeldahl N. However, the study was only conducted over a single growing period, and the discrepancies between the herbaceous and woody taxa may be explained by herbaceous plants establishing themselves more quickly in comparison to the relatively "immature" woody taxa (Grebenshchykova et al., 2019). Despite their slower growth rates, the study highlighted the potential to use woody plants such as willow (Salix) as wetland vegetation, along with the potential for herbaceous taxa such as *Typha* to accumulate nutrients. Additionally, recent research has highlighted the potential for plants commonly associated with wetlands (e.g. *Typha latifolia*) to be useful as bioethanol (Rebaque et al., 2017). Where alternative plants are used, the plant should be a species with a high growth rate and with a resilience to disturbances such as coppicing, along with being tolerable of the high nutrients present in the effluent (Grebenshchykova et al., 2019).

A summary of decentralised P removal methods is presented in Table 7.

Table 7 - Summary of decentralised methods (adapted from Environment Agency, 2012).

	Packaged plants	Sand filters	Constructed wetlands	Willow Coppice Plantations
Achievable effluent P quality.	Phosphorus removal is typically low - 10 to 15 % in aerobic/anaerobic trickling filter package plants (USEPA, 2007)	Media dependant. If high-Fe sands and high-Al muds used possible to remove 50 to 95 % TP		Up to 98 % Total P removal possible (Lachapelle-T. et al., 2019).
Suitability for small-scale WWTWs	Yes. Most often used in trailer parks, highway rest zones and rural areas; limited maintaince etc. required	Yes - low capital, operation and maintenance costs - but requirement to maintain aerobic conditions.	Yes, but land requirements	Yes, but land requirements. Similar N:P:K characteristics, i.e. N:P:K requirements for willow of 100:14:72 vs N:P:K occurrence in waste-waters of 100:18:65.
Process limitations in terms of influent wastewater/sludge characteristics.	Poor P removal performance. May require additional polishing methods, e.g. chemical precipitation	Requirement to maintain aerobic conditions; Pre-treatment to remove solids - prevent clogging; processes temperature dependant	Land requirements; drought may lead to plant deaths; clogging of filter media possible - solids, bioslimes etc.	Land requirements. Possible eventual soil P saturation e.g. Lachapelle-T. et al. (2019).
Ability to retrofit to existing WwTW?	Yes, but tend to be mass-produced therefore limited bespoke design options	Yes, but land requirements	Yes, but land requirements	Yes, but land requirements
Information on scale of costs: Capital costs and Operations and maintenance (O&M) costs	Capital costs – low O&M costs – low See Tables 1-3	Capital costs – low O&M costs – low	Capital costs – low/moderate O&M costs – low	Capital costs – low/moderate O&M costs – low; Provides income (biofuel) for land-owner

Information on energy requirements: aeration energy (AE), pumping energy (PE), and mixing energy (ME)	AE – N/A PE – low or none ME – N/A See Tables 1-3	AE – N/A PE – low or none ME – N/A	AE – N/A PE – low or none ME – N/A	AE – N/A PE – low/moderate?? ME – N/A
Scale of use:	Widely used in UK US, (e.g. RBCs in N.I.)		Widely used (>140 CWs in Ireland; >900 in U.K.)	Used for several different phytoremediation processes, e.g. heavy-metal extraction
Barriers to implementation	Poor P removal performance. May require additional polishing methods, e.g. chemical precipitation	Land availability	Land availability	Land availability
Sustainability issues – any social, environmental or economic impacts not considered elsewhere		Aerobic conditions must be maintained in the bed, otherwise bacteria may reduce Fe (III) under anaerobic conditions if the pH values decrease releasing soluble P into the effluent. When the bed's capacity for phosphate has been exhausted, the bed needs to be replaced, but P mining possible.	Maintaining anaerobic conditions in wetlands may assist peat formation and P accumulation. Potential to mine out P from primary receiving cells of ICWs after 10-15 years. Provider of ecosystem and recreational services.	Potential for system to be energy neutral; bio-fuel

Table 7 – continued (summary of decentralised methods).

6. Discussion and conclusions

Willow coppice plantations manage to divert P rich effluent away from streams while simultaneously removing P through plant/litter accumulation. However, there may be issues with potential soil P saturation which to date has received little attention, along with sourcing and retaining long-term land use. When planning or retrofitting small-scale WWTWs, consideration should be given to the costs involved and to any social and environmental concerns that may arise. Additionally, selected treatment options need to effectively deliver on P discharge consents, and there is a requirement for systems to operate with a low/minimal level of maintenance given the potential isolation of most WWTWs (Bunce et al., 2018). The review here highlights the main alternatives to the use of WCPs, with the pros and cons of each method presented and their suitability to remove P.

While physico-chemical methods are a reliable and well-established method for providing consistently high P-removal performance, they are associated with the production of a large amount of sludge and the necessity for additional chemicals, chemical storage and secondary environmental impacts such as altering stream pH and chemical toxicity (Bunce et al., 2018). Biological systems (e.g. EBPR) on the other hand, have minimal reliance on chemicals and produce less sludge, but are prone to fluctuations in P extraction performance (Oehmen et al., 2007; USEPA, 2009). Both of these systems are associated with additional maintenance and operational costs, such as operator supervision in the case of biological systems, and chemical feeding, storage and removal, in the case of physico-chemical methods. In contrast, decentralised technologies such as package plants, CWs and WCP systems require minimal supervision, and are associated with low/moderate maintenance and operational costs. However, their P removal performance is typically lower than that of physcio-chemical systems and much more variable (e.g. Healy and Cawley, 2002; Healy et al., 2007; Hickey et al., 2018). Additionally, in the case of WCPs and ICWs there are associated land use costs.

The review highlights that, for small-scale WWTWs, no single method is definitive in its suitability for the treatment of waste-water, and the best case scenarios are likely to see a combination of methods; for example the use of chemical polishing with package plants, or absorptive media with construction wetlands. When taking account of holistic environmental considerations however, WCPs and ICWs are a much more appealing option. Willow coppice plantations for example, provide a sustainable bio-fuel source, while ICWs help improve and benefit ecological habitats, provide ecosystem services such as recreation and educational amenities and have the potential to provide flood relief (Becerra-Jurado et al., 2014; Mackenzie and McIlwraith, 2015; van Biervliet et al., 2020). The zero-discharge wetlands system, that combines CWs with WCPs may also prove useful. However, its main use to-date (at least in Ireland and Denmark) has been for treating waste-water from single-dwelling households (Brix and Arias, 2005; O'Hogain et al., 2011), although the restored Churchtown Landfill site in Lifford, Co. Donegal, that combines ICWs with WCPs, is an example of its potential for larger scale use (e.g., https://supergen-bioenergy.net/newspdf/Biomass-Feedstock_Johnston.pdf).

Other factors to consider when planning small WWTWs relate to the volume of discharge water relative to the receiving water, as for example, the stream order may influence the stream dilution ability and with subsequent impacts on water quality impact. The cumulative effects of multiple WWTWs occurring on a stream is another important consideration. Carey and Migliaccio (2009) for example, cites the South Plate River Basin in Colorado as having over 100 municipal WWTWs discharging into it, with the discharge at times accounting for 100 % of the streamflow.

Future-proofing and climate-proofing WWTWs is also important. Outside of the afore mentioned nutrient pollutants, WWTWs are significant sources for other pollutants such as priority substances (PSs – e.g. benzene, lead, mercury) listed under the European Union Water Framework Directive (WFD) along with pollutants classified as "contaminants of emerging concern" (CECs) such as steroids, hormones, pesticides, and pharmaceuticals (Gorito et al., 2017) and micro-plastics. A recent evaluation report of the Urban Waste-water Treatment Directive highlighted the inadequacy of current treatment works to deal effectively with CECs and micro-plastics (EC, 2019). Any future revisions of the UWWT Directive are therefore likely to see a more stringent policy relating to emissions containing PSs and CECs. With this in mind, applications that are capable of removing multiple stressors should perhaps be prioritised. Sun et al. (2019) for example, reported good results when assessing the potential for a poly-3-hydroxybutyrate-co-3-hydroxyvalerate (PHBV) based solid-phase denitrification (SPD) systems to simultaneously remove both N and the pharmaceutical products ibuprofen and triclosan from waste-water effluent. Research at Ulster University assessed the use of membrane bio-reactors (MBRs) to remove trace organic chemical contaminants such as steroidal hormones, pesticides, and pharmaceuticals (Trinh et al., 2012a) and endocrine disrupting chemicals (Trinh, et al., 2012b) from waste-water, and the use of a sawdust derived bio-adsorbent to remove anti-microbial pollutants (Tretsiakova-McNally et al., 2019). Willow has been widely used for phytoremediation purposes, and has been shown to be effective in removing PSs such as lead and cadmium (Mleczek et al., 2017; El-Mahrouk et al., 2019) and CECs such as pesticides (Sun et al., 2013; Lafleur et al., 2016), and so again may be useful in this instance. Given that WWTWs are responsible for approximately 0.8% of the total EU energy consumption (EC, 2019), revisions to the UWWT Directive are also likely to require WWTWs to be more energy neutral. Again, WCPs may be applicable here due to the harvested plant being used as a bio-fuel in a circular economy, while ICWs are additionally a low energy option.

Phosphorus has many uses, such as the production of detergents and in food processing, although its main use is in the manufacture of fertilizers (Bacelo et al., 2020). The primary source of P is from mining "phosphate rock" deposits, with the main global reserves concentrated in China, Morocco and the USA (Cooper et al., 2011). However, these supplies are limited, with estimates of between 50 -300 years before these natural deposits are exhausted (Cooper et al., 2011; Van Vuuren et al., 2010). Given the likelihood of a global population increase and therefore an increase in food production and subsequent P fertilizer requirements, potential methods for the recycling/recovery of P should be sought. While recovery of P from WWTW effluent is one option, the present options from for example physico-chemical systems are restrictive based on costs (especially for small-scale

WWTWs) (Bunce et al., 2018). This may however need to be re-visited especially where more efficient and alternative methods are developed, as highlighted in the Novel technologies section (4) of this report. In contrast, recovery of P from biological processes such as EBPR (Bacelo et al., 2020) and from ICW systems (Scholz et al., 2007a) is a much more achievable option, and this should be factored into any decision making process.

In conclusion, WCPs remains a desirable option for the treatment of waste-water effluent at small-scale WWTWs, or at the very least be combined with other systems such as ICWs. While alternative methods exist, their usefulness is limited either through capital, operational and maintenance costs or through poor/variable P removal performance (e.g. CWs). Willow Coppice Plantations have the potential to remove P, allow the treatment of effluent to be an energy neutral process and keep the land they occupy in production, providing a source of income (harvest – biofuel) to landowners.

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