Nutrient recovery and recycling from wastewater in Ireland, with associated policy gaps and recommendations.

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1 Introduction

Nitrogen, phosphorous and potassium, together with carbon, are the nutrients at the basis of life, and they are essential to sustain cellular metabolism. For this reason, they are widely used as fertilisers in agriculture and food production. According to the Food and Agriculture Organisation (FAO), in 2018 the world demand for nitrogen, phosphorous and potassium as fertilisers was 106 MT, 46 MT and 38 MT respectively, but these quantities are projected to increase by 5, 7 and 8% by 2022 (FAO, 2018). Historically, fertilisers were only obtained from organic sources (e.g. manure), but at the beginning of the 20th century they became extensively available following the development of industrial processes for their large scale production. Nowadays, the majority of the ammonia utilised in agriculture is produced through the Haber-Bosch process, which combines atmospheric nitrogen with hydrogen obtained from natural gas (Rouwenhorst, Krzywda, et al., 2021). Despite several improvements over the year, the Haber-Bosch synthesis remains an energy intensive process, typically requiring 12.1 kWh per kg of NH_3-N (Pikaar, Matassa, et al., 2017). It is a process so prominent and widespread that it is responsible for roughly 2% of the global energy consumption (Desloover, Abate Woldeyohannis, et al., 2012), and originated roughly 80% of the nitrogen currently found in human tissues (Howarth, 2008). Similarly, potassium-containing salts (e.g. KCl) and mineral phosphates have become the main source of both potassium and phosphorous fertilisers as result of increasingly efficient processes to mine them and convert them into soluble forms that plants can use.

This has implications on different levels. For instance, more than 85% of the global phosphate reserves are located in five countries, and among them Morocco is projected to provide 80% of the world supply by 2100 (Cooper, Lombardi, et al., 2011). Countries such as Brazil or India are largely dependent on imports of P fertilisers, and so is the EU. European Countries import over 90% of the demand from US and China (*Consultative Communication on the Sustainable Use of Phosphorous*, 2013). For these reasons, it is not surprising that phosphate rock was added to the EU critical raw material list in 2014 (EU, 2014), as well as white phosphorous in 2017 (EU, 2017).While the global P reserves are quite large and will not be depleted in a short term, regional P scarcity is likely to become a growing issue. As there is no substitute for phosphorous in food production, this might jeopardise food security, especially considering that the world population is expected to reach 9.7 billion over the next 30 years (ONU, 2019) and that the pro-capita meat consumption is also increasing (animal derived food require 7-10 times P than crops - Metson, Bennett, et al., 2012).

Ireland is the 9th largest fertiliser consumer in the EU (European Environment Agency, 2015). In particular, according to O'Donnell, Egan, et al. (2021), 63% of the Irish agricultural land has agronomically insufficient levels of plant available phosphorous. Only one third of Ireland P requirements comes from indigenous sources (mainly cattle and pig slurry, poultry waste, dairy processing sludge and municipal wastewater sludge), and it is necessary to import 43 000 tonnes of P-based fertilisers every year. In their analysis, they also provide a comparison between the phosphorous requirements and the indigenous recycled P that is available in different regions of the republic of Ireland (Table 1).

Table 1 - Combined indigenous recycled P sources vs. regional P requirements estimated by P application for maintenance and to achieve optimum agronomical soil P-levels (O'Donnell, Egan, et al., 2021).

Region	Regional P requirement (t)	Indigenous recycled P (t)	Indigenous recycled P (%)
Dublin & mid-east	9756.2	3100.7	31.7
South-east	16714.8	4646.2	27.8
South-west	15264.6	6470.9	42.4
Mid-west	16395.2	3603.8	22
West	11719.1	2855.7	24.4
Midlands	10339	2810.1	27.2
Border	15404.4	5061.5	32.9

A change in phosphorous management is therefore necessary, not only in Ireland but in Europe as a whole, improving P recovering from waste streams and cost-effectively transporting the obtained fertilisers to regions with less P (Schoumans, Bouraoui, et al., 2015).

Another element of concern is linked to the use of natural gas as the main source of hydrogen for the ammonia production, which makes the fertilisers industry vulnerable to price volatility. As an example, in recent months a combination of reduced natural gas export from Russia, increased demand from China and India and reopening after COVID-19 restrictions, has resulted in a very sudden and significant increase in the natural gas price in Europe (Bond, Cornago, et al., 2021). This has not only affected electricity prices, but had impact on the cost of fertiliser: according to the Irish Farmer's Association, the price of UAN (Urea/Ammonium/Nitrate) increased 228% compared to 2020, costing as much as 860 €/ton (IFA, 2021). This trend is confirmed by the data collected from the world bank (Baffes and Koh, 2021), which are reported in Figure 1.

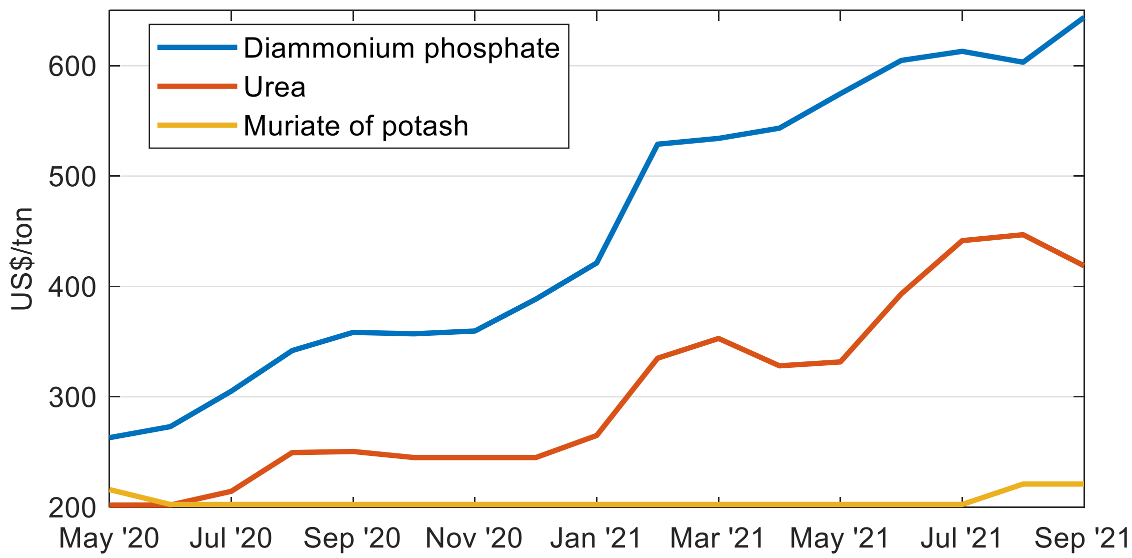

Figure 1 - Price of commonly used fertiliser products

A report from the Agriculture and Food development Authority suggests that increases in fertiliser prices in the order of 100% can be expected in 2022 (Teagasc, 2021).

Besides these economic issues, it is important to remember that the large amount of nutrients that is being introduced in the environment as a result of the industrial fertiliser availability is having an increasing impact on the global nitrogen and phosphorous cycles. For instance, it is estimated that as a consequence of human activities, the amount of nitrogen that enters the nitrogen cycle has doubled (Vitousek, Aber, et al., 1997), whereas the phosphorous mobilisation in land and water has tripled (Yuan, Jiang, et al., 2018).

Human alterations on phosphorous and nitrogen cycles also affect the ecosystems. High levels of N compounds can be toxic for certain organisms, can lead to soil and water acidification and even contribute to global warming when released into the atmosphere.

Excess nitrogen and phosphorous from agricultural usage can also permeate the soil and reach streams and lakes. This nutrient discharge in aquatic ecosystems is linked to algal blooms that pose a threat to the life of plants and animals and reduce the quality of the water they live in. In particular, the decomposition of dead algae can create anoxic zones where the dissolved oxygen concentration is not sufficient to support most organisms. Moreover, high nutrients concentrations (especially if there is a low nitrogen to phosphorous ratio) are ideal for the growth of cyanobacteria, which are known for their ability to produce toxins that are poisonous for both the wildlife and human beings. In the same way, nutrients are dissolved in the wastewater generated both by industries and households. Nitrogen is mainly present in wastewater in the form of urea or combined in proteins, which are decomposed to ammonia by the microbial activity. Under aerobic conditions, bacteria can then oxidise the ammonia to nitrites (NO_2^-) and nitrates (NO_3^-), which can be converted back into protein closing the cycle.

Municipal wastewater typically also contains 4-16 mg/l of phosphorous, which can be found in different forms. In general, orthophosphates (e.g. PO_4^{3-}, HPO_4^{2-}, $H_2PO_4^-$, H_3PO_4...) can be directly metabolised by the microorganisms, whereas polyphosphates require to be hydrolysed to orthophosphates first (Tchobanoglous, Burton, et al., 2003). Thus, the wastewater treatment sectors plays a critical role in preventing large amounts of nitrogen and phosphorous from reaching the waterbodies. Effective treatment is crucial for healthy ecosystems.

Despite its importance, however, the 2019 EPA report showed how in Ireland only 30.4% of the wastewater treatment plants (WWTPs) are designed to provide secondary treatment with some form of nutrient removal (EPA, 2019). In other words, more than 50% of the wastewater that is generated in urban areas is discharged with a level of treatment below European Union standards (EPA, 2020b). Moreover, according to the River Basin Management Plant (RBMP) that is being drafted for 2022-2027 (Government of Ireland, 2022), urban wastewater is the 4th largest pressure on water quality in Ireland affecting 208 water bodies, while, 188 waterbodies are directly impacted by domestic wastewater and septic tanks. More than one third of the Irish waterbodies are classified as "at risk", requiring restoration measures to achieve the objective set in the Water Framework directive (European Commission, 2000). There is also an additional 1256 waterbodies (26% of the total) which status is currently under review, which means that more information is required to determine whether they are at risk or not, or that the measure taken so far for their protection have not been enough to achieve the "not at risk" status.

On one hand, this presents significant challenges in terms of improving the currently insufficient infrastructure, but on the other it means that there are substantial opportunities to reshape the wastewater treatment sector. The inclusion of nutrient recovery processes in the layout of existing plants constitutes an excellent example of revamping interventions that addresses these issues.

Nutrient recovery is an effective way to reduce the amount of pollutants that is discharged by wastewater treatment plants (as well as the extent of the associated eutrophication), while producing additional fertilisers for the food industry and reducing the European dependency on their importations.

The ramifications of the adoption of nutrient recovery technologies are however not limited to food security and P and N cycles. On a much broader sense, a widespread implementation of nutrient recovery technologies could be beneficial for the energy system as a whole. Being responsible for 2-3% of the world's total electrical consumption, wastewater treatment plants are quite energy intensive (Emami, Sobhani, et al., 2018). In particular, nitrogen removal is responsible for a considerable fraction of the energy required for the plant operation, increasing the energy necessary for aeration, pumping and solids processing in conventional activated sludge plants by 30-50% depending on the C/N ratio of the influent (Tchobanoglous, Burton, et al., 2003). This results into up to 60% of the plant operating costs being a consequence of conventional nitrogen removal (Ledezma, Kuntke, et al., 2015). There is a growing awareness that converting ammonia to nitrogen gas is not a sustainable process, and evidences suggest that shifting the focus from nutrient removal to nutrient recovery can significantly reduce the WTTPs energy consumption (Daneshgar, Buttafava, et al., 2019), with reported savings as high as 27% .(Levlin and Hultman, 2003). This directly reduces the CO_2 emissions associated with the generation of the energy that is necessary for the plant operation. Recovering nutrients also reduces the need to manufacture them through the previously mentioned industrial processes, which also has benefits in terms of GHGs emissions. Finally, GHG are generated in the conventional nutrient removal processes themselves: anaerobic digestion for the biogas production generates CO_2 as a by-product, methane can leak from the pipelines and methane emissions can originate from sludge handling or primary sedimentation, and nitrification can be associated to significant NOx emissions (Mannina, Ekama, et al., 2016). Depending on the nitrification and denitrification operation, as much as 5% of the nitrogen load to the plant can be emitted as N_2O, whereas methane emissions can reach up to 0.0085 kg CH_4/kg COD (Soares, 2020). Given their significant global warming potential (roughly 20 and 300 $CO_{2\,\text{equivalent}}$ for methane and N_2O respectively), to manage these emissions is extremely important (Soares, 2020).

It is also noteworthy to mention that, due to the lower amount of energy required, the application of nutrient recovery technologies has the potential for reducing the wastewater treatment plants operational costs. For instance Dockhorn (2009) found that phosphate recovery allowed an European plant to save 2-3 €/kg P compared to the removal of the same quantity through conventional processes. Phosphorous recovery can also be associated with a 2-8% reduction in the sludge production, which in turns results into lower sludge management costs

Conventional wastewater infrastructures are designed and operated with the goal of reducing human exposure to "waste", removing it from people proximity and disposing it through a sewage network that transports it to a centralised WWTP. However, as the awareness of the importance of the effects of the human activities on the environment grew over the past years, the drawbacks and limitations associated with this traditional approach have become more and more evident. WWTPs are often both water and energy intensive, and the ultimate disposal of the treated waste in waterbodies and landfills contributes to an already high environmental burden (Cumming, 2009).

The development of new processes and technologies promoted a paradigm shift in the wastewater treatment sector: the nutrients dissolved in the wastewater are more and more being considered as a resource to be recovered, rather than merely as pollutants to be removed. Wastewater treatment plants are abandoning their traditional role and are increasingly becoming wastewater resource recovery facilities (WRRFs) capable of producing clean water and nutrients, as well as renewable energy (Regmi, Miller, et al., 2019; Hamiche, Stambouli, et al., 2016).

In the following sections of this report, the most prominent examples of nutrient recovery technologies will be described in detail, identifying cases studies, and reviewing the policies that are in place in Ireland and the EU.

2 Technologies and processes for nutrient recovery – overview

Conventional processes for wastewater treatment are quite mature and widespread. Their adoption is crucial for the protection of the ecosystems, and they can achieve remarkable levels of pollutant removal from the treated influent. However, the current approach does not address the critical issues associated to fertiliser production and over-exploitation that were discussed in the previous section of this report. In common practice, the nitrogen that is fixed in organic compounds is converted back into gaseous N_2, and phosphates are often precipitated with elements such as iron, forming compounds that are not easily utilised by crops. It is thus necessary to resort to Haber-Bosch synthesis and to phosphorous mining to obtain the nutrients used in agriculture, consuming energy and introducing more alterations on N and P cycles. According to Cruz, Law, et al. (2019), around 20 million tons of ammonium are globally discharged in wastewater every year, which corresponds to roughly 19% of the production through Haber-Bosch synthesis, and which would require ~240000 GWh every year.

To recover these nutrients can be an effective way to produce wastewater-based fertilisers, contributing to food security while reducing the environmental footprint of wastewater treatment (e.g. less energy required, diminished sludge production…) and the natural resources consumption (Yan, Ye, et al., 2018).

In general, the nutrient recovery processes become more efficient and feasible when the nutrient concentration in the water increases. A variety of particularly nutrient rich wastewater streams have been considered for nutrient recovery, such as sewage, urine (Gao, Liang, et al., 2018) and landfill leachate (Zhang, Peng, et al., 2017).

However, the conventional wastewater collection systems are not designed to maintain high concentration of nutrients, in fact quite the opposite. Dilution starts in the very first step, flushing toilets. Moreover, it is very common for sewer system to receive and transport not only households and industrial flows, but also the meteoric water that reaches the ground through precipitation (e.g. rainfall, or snow melting) (Gernaey, Flores-Alsina, et al., 2011). Hence, nutrient recovery usually involves an initial step in which nutrients are concentrated (for instance through their accumulation in biomass), followed by their actual recovery, which is achieved through various methods.

An overview of the main processes that are available for nutrient recovery is reported in Table 2, together with a summary of the key challenges that are associated to their mainstream implementation

Table 2 - Overview of nutrient recovery methods and key challenges for mainstream implementation

Process	Key challenges
Struvite and hydroxyapatite formation	• High operating costs (mainly due to Mg/Ca sources), increases the final product costs • Purity of the final product • More effective on concentrated streams
Adsorption/Desorption processes	• Identification of suitable chemicals to minimise impurities in the final product • Economical feasibility depends on the nutrient concentration stream
Membrane systems	• Membrane fouling
Bioelectrochemical Systems	• Further research and development required for full-scale implementation
Stream segregation technologies	• Significant modifications of the existing infrastructure might be required • Behavioural modifications required

In this section of the report, the main processes that are available for nutrient recovery will be presented and discussed.

2.1 Biological processes

As for nitrogen and carbon, biological processes can be exploited for the recovery of phosphorous from wastewater. While microorganisms in conventional activated sludge systems can accumulate up to 2% on a dry biomass basis, polyphosphate-accumulating organisms (PAOs) are capable of taking up and storing P in their cells in quantities exceeding the growth needs. For instance, P content of up to 8% on a dry biomass basis have been reported for the PAOs used in enhanced biological phosphorus removal (EBPR) processes (Yang, Shi, et al., 2017). Although this kind of processes is becoming increasingly common and it is significantly contributing to reducing eutrophication, they do not address the disruption of the P cycle unless they are coupled with P recovery technologies. This synergy is a promising way to exploit the PAOs ability to concentrate phosphate (Blank, 2012).

The literature reports successful applications of EPBR processes for wastewater characterised by P concentration ranging from 20 to >100 mg P/l, obtaining 90-99% P removal efficiencies (Yuan, Pratt, et al., 2012).

Recent studies have shown that microalgae can also accumulate P (up to 3.3%), thus having the potential to be an alternative to PAOs (Yuan, Pratt, et al., 2012).

Biological processes to separate P and concentrate it in biomass are anyway only the first step for phosphorous recovery: it is also necessary to individuate ways to effectively release the stored P and make it available. The simplest method is the direct application of the P enriched biomass on cultivated fields. However, this low cost approach presents two main issues.

- The biomass can contain traces of heavy metals such as chromium, copper, lead and cadmium, which can accumulate in the soil and eventually pose a threat for health. Additional risk can be associated with micropollutants (pharmaceutical products, detergents, hormones, microplastics…). These contaminants can permeate the soil and reach groundwater, or become part of the crops (Chavoshani, Hashemi, et al., 2020).
- It is important to monitor the N/P ratio of the biomass. According to Tchobanoglous, Burton, et al. (2003), the sludge obtained from EPBR processes is characterised by an N/P ratio close to 1, but most plants require a higher nitrogen content (N/P ~ 3-5), which might limit the sludge application.

Settling the sludge containing polyphosphate-accumulating organisms in anaerobic conditions causes them to release the stored phosphorous. The supernatant is then a P-rich stream that can then undergo chemical or thermo-chemical processes, increasing the maximum achievable phosphorous recovery. Phostrip process (Figure 2) is an early example of application of this concept (Levin and Sala, 1987). A portion of the recycled activated sludge is sent to a stripping tank, where it remains under anaerobic conditions. In this way, the sludge releases the accumulated phosphates, which are then removed from the stripping tank in the supernatant.

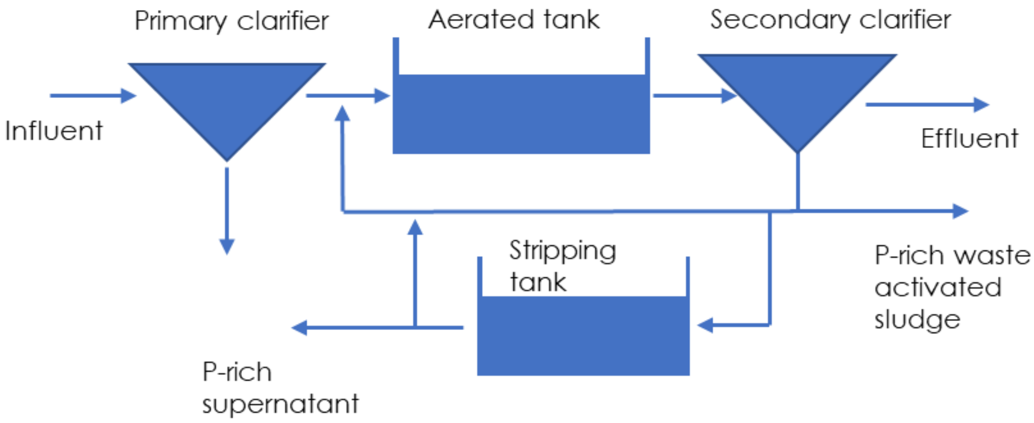

Figure 2 - Phostrip process conceptual flow diagram

This configuration allows for P recovery from both the wastage P-rich activated sludge and the stripping tank supernatant. However, a system based only on biological processes does not generally meet the discharge limitations, and it is therefore often operated in combination with chemical precipitation or thermal treatments. P-rich streams can be used both for phosphate recovery (e.g. forming apatite), or combined with high ammonium concentration flows such as the ones obtained from dewatering digested sludge, and dosed with magnesium for struvite precipitation (Levlin and Hultman, 2003).

The portions of phosphorous that is bounded in nucleic acids within the biomass is highly resistant to biological release, but can be made available through thermochemical approaches (Gifford, Liu, et al., 2015), which can also allow for the recovering of additional resources such as elements from the platinum group (Westerhoff, Lee, et al., 2015).

2.2 Chemical processes

2.2.1 Struvite and hydroxyapatite formation

The conventional methods for phosphorous removal from wastewater involve the dosing of chemicals (metal salts) such as aluminium sulphate, sodium aluminate, ferric chloride, ferric sulphate, ferrous sulphate and ferrous chloride (M.P.C.A, 2006). These metal salts react with the dissolved phosphate, forming solid precipitates that are then easily removed by solid separation processes (e.g. settling or filtration). They were chosen for their ability to form very stable P compounds, to make the separation process easier. However, this also means that the phosphorous they contain cannot be directly metabolised by organisms. An alternative approach utilises magnesium and calcium based materials, which combine with phosphates according to the reactions described in equation 1 and 2, and produce struvite ($MgNH_4PO_4 \cdot 6H_2O$) and hydroxyapatite ($Ca_5(OH)(PO_4)_3$) respectively.

$$Mg^{2+} + PO_4^{3-} + NH_4^+ + 6H_2O \rightarrow MgNH_4PO_4 \cdot 6H_2O \downarrow \qquad [1]$$

$$5Ca^{2+} + 3PO_4^{2-} + OH^- \rightarrow Ca_5(OH)(PO_4)_3 \downarrow \qquad [2]$$

Figure 3 - Hydroxiapatite crystal

Hydroxyapatite (Figure 3) is a valuable resource that can be used as raw material in the phosphate industry or as slow release P fertiliser (Shashvatt, Aris, et al., 2017). It is naturally found in human and animal bones and teeth, and processes to promote its exploitation in agriculture are being researched (Liu and Lal, 2014; Kottegoda, Khalil, et al., 2011).

Struvite crystals (Figure 4)are also a promising fertiliser that can supply both phosphorous and nitrogen. As shown in equation 1, the formation of struvite crystals requires stoichiometric amounts of phosphate, ammonium and magnesium. Unlike traditional ammonium-phosphate fertilisers, struvite is characterised by a low solubility in pH neutral solutions, which makes it a slow release fertiliser that does not "burn" roots, helping reducing nutrient run-off and its impact on nearby waterbodies (Shu, Schneider, et al., 2006).

Figure 4 - Struvite crustals

An overview of the properties of the two compounds can be found in Table 3

Table 3 - Hydroxyapatite and struvite properties.

	Hydroxyapatite	Struvite
Formula	$Ca_5(OH)(PO_4)_3$	$MgNH_4PO_4 \cdot 6H_2O$
Molecular mass	502.3 g/mol	245.4 g/mol
Density	3.2 g/cm^3	1.7 g/cm^3
Colour	White/gray/yellowish green	White/yellowish white/brownish white
Solubility K_{sp}	3.2×10^{-59}*	2.05×10^{-14}**

*(Prakash, Kumar, et al., 2006)
**(Bhuiyan, Mavinic, et al., 2007)

Current fundamental research has shown that struvite crystallisation is an effective solution for P recovery, although the yield of the process is largely dependent on a number of factors (Li, Huang, et al., 2019). In particular, temperature, pH, seed selection and the concentration of nutrients in the wastewater have a great influence on P removal. A study from Rittmann, Mayer, et al. (2011) indicates that struvite crystallisation requires phosphates concentration between 100 and 200 mg/l, which is considerably higher than the one typically found in municipal wastewater. One of the major obstacles to the application of struvite crystallisation technologies is their high operating cost, which is mainly due to the cost of the magnesium source. However, choosing cost-effective Mg sources can lower the struvite costs by 18-81% without necessarily complicate the process or requiring additional operations (Wang, Ye, et al., 2018). In coastal areas, seawater or bittern (the solution remaining after the crystallisation of salt from brine or seawater) can be successfully used as Mg sources. According to Maaß, Grundmann, et al. (2014), seawater usage is an attractive option despite the extra maintenance due to high salinity, which does not significantly affect the process. In some cases, however, less homogeneous (and thus less valuable) struvite was obtained when using seawater (Lahav, Telzhensky, et al., 2013).

The purity of the produced struvite is an important factor for its direct use in agriculture. Heavy metals or metalloids (e.g arsenic) contained in the wastewater can be incorporated in the struvite crystals during their precipitation, and cause soil contamination. However, the benefits associated with nutrient recovery are fuelling a growing interest on the struvite precipitation process, with a number of studies focused on different wastewater streams, from bakery production (Uysal, Demir, et al., 2014), to farming and slaughterhouse wastewater (Kabdaşli, Tünay, et al., 2009), landfill leachate (Huang, Xiao, et al., 2014), potato processing industry (Uysal and Kuru, 2013) and human urine (Nagy, Mikola, et al., 2019).

In conventional wastewater treatment plant, uncontrolled struvite precipitation can occur in pipes, centrifuges and pumps, damaging the equipment and resulting in extra costs for its cleaning/replacement. However, phosphorous recovery through struvite crystallisation mitigates these issues, and it is an excellent example of how changing the traditional processes allows to turn previous issues into valuable materials.

According to Melia, Cundy, et al. (2017), recovering ammonium and phosphate through struvite precipitation could significantly reduce the operational costs of a wastewater treatment plant in the Netherlands. It was also reported that the produced struvite could be sold to the fertiliser industry for 50-100 €/t.

2.2.2 Adsorption/desorption processes

Adsorption is considered another promising technique for nutrient recovery. It necessitates a simple design and operation, and it is characterised by low cost and high stability. Thanks to the high selectivity, it is a viable alternative for water contaminated with low level of phosphate (Bacelo, Pintor, et al., 2020). Metal- and biochar-based adsorbents are generally utilised to recover phosphate, but the use of low-cost recycled materials from bio-derived resources and waste is also being investigated for both economic and environmental reasons (e.g. egg shells - Köse and Kivanç, 2011).

The adsorption step has to be followed by desorption to recuperate P and regenerate the adsorbents. Different compounds have been tested for this purpose, and alkaline NaOH solutions are in general regarded as the most appropriate ones.

Ammonium ions dissolved in the wastewater can be converted to volatile ammonia when exposed to high temperature/pH. It is then possible to adsorb the stripped ammonia utilising acid solutions (e.g. sulphuric acid), forming ammonium sulphate (Ye, Ngo, et al., 2018).

To ensure that the nutrients recovered through adsorption/desorption processes can be directly used in agriculture, both the adsorbent and desorbent should be accurately selected to limit the unwanted pollutants quantities that are present as impurities in the finished product. The selection of the chemicals used in the process is also very important for the economics of the nutrient recovery. The economic feasibility of the process is also largely dependent on the nutrient concentration: for instance, according to De Vrieze, Smet, et al. (2016), recovering ammonia through stripping and adsorption is only a viable option if the total ammonium nitrogen concentration exceeds 1000-1500 mg/l.

2.3 Membrane systems

As the biological processes described in the previous sections of this report, membrane technologies also offer the opportunity to increase the nutrient content in wastewater. Membranes can also be used to separate nutrients from other unwanted substances, such as heavy metals and pathogens, with relatively low energy requirements (Ye, Ngo, et al., 2018).

Different approaches are available. It is possible to concentrate the nutrients suspended in the wastewater through microfiltration, ultrafiltration or nanofiltration, whereas technologies such as forward osmosis (FO), reverse osmosis (RO), membrane distillation (MD) and electrodialysis (ED) are used to recover P and N that are dissolved in the liquid phase.

In forward osmosis, a semipermeable membrane separates the feed side (wastewater) from the draw side, where a high concentration solution is flowing. The concentration gradient across the membrane creates an osmotic pressure, causing the water molecules to flow from the feed side to the draw side. In the process, the concentration in the feed stream increases, whereas the draw solution is diluted. In principle, forward osmosis does not require additional energy inputs. However, as the concentration of the draw solution decreases, so does the osmotic pressure gradient across the membrane, which ultimately decreases the process driving force.

On the other hand, reverse osmosis exploits an artificial pressure gradient across the membrane. In this configuration, there is no need for a high concentrated solution flowing in the draw side, thus eliminating the issues related to a diminishing driving force. The process does however require hydraulic pressure to be applied on the feed side (with the associated high energy costs), to overcome the osmotic pressure and to force water molecule through the membrane to the draw side. This results into an increased concentration in the membrane feed side.

Osmosis systems find application in both phosphorous and nitrogen recovery. However, it is generally possible to achieve higher phosphate concentrations due to the larger hydrated radius of phosphates. The membrane surface can also become negatively charged in alkaline environments, rejecting negatively charged phosphate ions. Conversely, positively charged ammonium ions are electrostatically attracted by the membrane, which means that the liquid phase pH plays a crucial role in the process efficiency (Ye, Ngo, et al., 2020).

Membrane distillation is a different process configuration, which can be used to recover nutrients from the wastewater following an approach that is similar to stripping and adsorption systems. Heating up the feed solution (from which nutrients need to be recovered) results in the volatilisation of some of the compounds dissolved in the liquid phase. A semipermeable membrane then allows the volatilised nutrients to reach the draw solution. When membrane distillation is used to recover ammonia, the draw solution is generally acid (e.g. sulphuric acid), as they react with the ammonia to produce ammonium salts (Qu, Sun, et al., 2013).

Finally, membranes can also find application in electrodialysis processes. A voltage gradient is applied to the feed solution, which causes positively charged ammonium ions and negatively charged phosphate ions to migrate towards different sections of the cell. A cation exchange membrane ensures that the ammonium ions remain in the cathode chamber, whereas the phosphate ions are retained in the anode chamber by an anion-exchange membrane. In this way, P and N can be concentrated before being sent to further recovery. This approach has been raising an increasing interest over the past 20 years, with studies being performed over industrial and municipal wastewater, excess sludge sidestream, separately collected urine and animal farming waste (Gurreri, Tamburini, et al., 2020).

One of the biggest challenges for the application of membrane systems to the nutrient recovery from wastewater is membrane fouling, which can deteriorate the membrane performance over time, increasing the energy requirements and negatively affecting the overall economic feasibility of the process (Quist-Jensen, Macedonio, et al., 2015). It is however possible to address this issue. Chemical cleaning of the membrane, reversing the electrode polarity or reducing the current intensity are the commonly utilised ways to lessen membrane fouling in electrodialysis processes (Mondor, Ippersiel, et al., 2009). Fouling is also observed in forward osmosis processes, which means that extensive membrane cleaning is required to the operation of this system configuration as well (Deng, van Linden, et al., 2021).

2.4 Bioelectrochemical systems

Bioelectrochemical systems are another promising approach to recover nutrients (as well as energy) from wastewater. They exploit microorganisms to carry out oxidation and reduction reactions, which eliminates the need for costly precious metals as catalysts. Microbial fuel cells (MFCs) were originally developed as a way to generate electricity from the wastewater, whereas microbial electrolysis cells (MECs) use electricity to generate biofuels (Modin and Gustavsson, 2014). They can however be repurposed to the recovery of nutrients (Fischer, Bastian, et al., 2011). Near the cell cathode, water is consumed and hydroxide ions (OH^-) are generated as byproduct. This also results into an increased pH in the cathode surroundings and, provided that a suitable amount of magnesium and ammonium are present, it results into favourable conditions for the direct recovery of phosphorous in the form of struvite. Recovery efficiencies up to 40 (Taddeo and Lepisto, 2015) and 90% (Tao, Zhou, et al., 2015) were reported for single chamber MFC and double chamber MFC respectively. However, as more struvite is formed, the accumulation of crystals near the cathode can hinder mass transfer and reduce the recovery efficiency (Santoro, Ieropoulos, et al., 2013).

The electrons flow in microbial electrolysis cells, combined with a suitable catalyst, allows the recovery of valuable products such as hydrogen and methane, but also struvite. A study from Cusick, Ullery, et al. (2014) reported a 70-85% reduction in the amount of phosphates dissolved in digestate supernatant obtained exploiting microbial electrolysis cells, associated to an energy consumption of 0.2-0.3 Wh/l, which is lower than energy required by other struvite recovery technologies. As discussed for MFCs, microbial electrolysis cells are also prone to electrode fouling issues, which can negatively affect their recovery efficiency.

While bioelectrochemical systems represent a promising technology that can contribute to nutrient recovery, they still require further research and development before they can successfully be applied at full scale for this purpose (Siciliano, Limonti, et al., 2020).

2.5 Stream segregation technologies

The different processes and technologies that were discussed in the previous sections of this report are addressing the recovery of nutrients in the context of the conventional existing infrastructure for wastewater collection, transportation and treatment. This means that a significant emphasis and effort are placed on methods that are capable of concentrating the nutrients dispersed in the wastewater, before the actual recovery starts to be economically feasible.

There is however a radically different approach which requires to move past the conventional combined sewers, in favour of a system that not only allows for dividing wastewater and meteoric water, but also for separate collection and individual treatment of various wastewater fractions. In the context of municipal wastewater, to separate the different streams (e.g. urine, faecal matters…) at the source would have clear benefits on the nutrient recovery: for instance, human urine contributes for less than 1% of the total wastewater volumetric flowrate, but it contains more than 80% of the total nitrogen, and more than half of the total phosphorous and potassium (Table 4)(Larsen, Lienert, et al., 2004; Vinnerås and Jönsson, 2002). Furthermore, to isolate the faecal matter (which contains most of the pathogens) would prevent them from contaminating the wastewater, reducing the risk for water-borne diseases (Ashbolt, 2004).

Table 4 - Typical nutrient concentrations in human urine (Samuel, 2021).

Parameter	Typical values (g/l)
Urea	10-20
Sodium	2.8 – 6.5
Potassium	0.8 – 3.7
Phosphate (PO_4^{3-})	0.7-2.4
Phosphate-P	0.2 – 0.8
Ammonia	0.47 – 0.5
Ammonia-N	~0.35

Other benefits associated with the use of human excreta as a source of nutrients are their low heavy metals content (Karak and Bhattacharyya, 2011). Heavy metals contamination in fertilisers obtained from recovered resources is mostly due to the treatment of industrial effluent, and even the use of mineral fertiliser has been linked with high levels of heavy metals in livestock feed and crops (Atafar, Mesdaghinia, et al., 2010).

This is not a new idea, and dedicated toilets (urine diverting toilets – UDT) are already available in various models and at competitive prices. Urine and faeces can be collected in a front and rear bowl respectively. Both compartments can be flushed, but configurations in which only one of them, or none of them are flushed are also possible. The applicability of diverting toilets in lieu of the conventional approach is well documented in a number of installations, particularly in South Africa, where they serve 450 000 inhabitants, or in Liberia, where they improved sanitation for 100 000 people (Simha and Ganesapillai, 2017). Pilot projects were also implemented in industrialised countries such as Denmark (Magid, Eilersen, et al., 2006) Germany (Winker, Vinnerås, et al., 2008), Sweden (Seneca, 2022), as well as India (Langergraber and Muellegger, 2005) and China (Zhou, Liu, et al., 2010). In Ireland, one example of stream segregation can be found in the Ballymum Rediscovery Centre (Co. Dublin), where composting toilets were introduced in the 2016 refurbishments, together with the urinal waste water collection and use for plant nutrition within the internal comfrey wall.

However, it is important to consider that it is easier to adopt such a novel approach in scenarios where it is not necessary to improve the existing conventional infrastructure, but rather to build new infrastructure to make up for the lack of adequate sanitation systems. To retrofit existing infrastructure at residential scale is not economically feasible, but it might be possible to concentrate the efforts on public buildings such as cinemas, malls, stadiums or universities (in these settings, urinals are already commonly installed in the men's restrooms, which makes it easier to implement source segregation).

From the technological point of view, the utilisation of source separated waste as fertilisers has also been investigated (Bonvin, Etter, et al., 2015; Ganesapillai, Simha, et al., 2016): the nutrients are generally plant available, and they have demonstrated a beneficial effect on the crop productivity that is comparable with the one resulting from the application of mineral fertilisers. The urea recovered from separately collected urine has a natural tendency to hydrolyse forming ammonia, and its use for struvite precipitation and vacuum stripping is a sustainable alternative to conventional industrial processes both for the synthesis of ammonia and the production of phosphorous fertilisers from phosphate rocks (Tao, Bayrakdar, et al., 2019).

There are, however, some issues that need to be addressed. For instance, it is necessary to regulate the application of human urine in soil to prevent the volatilisation of ammonia (which is a greenhouse gas), and to avoid the uncontrolled increase in the soil pH (Heinonen-Tanski, Sjöblom, et al., 2007). Secondly, large volumes of urine are required to achieve the same fertilising effect as industrial fertilisers, which poses a challenge on the logistic associated with the urine collection and transportation. Thirdly, in some cases (Höglund, Ashbolt, et al., 2002), the collected urine was found to be contaminated by pathogens, and the long term effect of pharmaceuticals are not fully understood (Larsen, Lienert, et al., 2004). Despite these legitimate concerns about micropollutants and pharmaceuticals however, the quantities that are generally found in human urine remain far less than wastewater or farm manure (Winker, Tettenborn, et al., 2008).

Finally, the availability of technologies that allow for an effective stream segregation and recovery of the nutrients from human excreta is not the only factor standing in the way of a large scale implementation of this approach. A socio-economic and behavioural shift is required, and it is the end users that are ultimately responsible for the adoption of this new idea. To date, only a small number of sociological studies on the matter have been published. Lienert and Larsen (2010) sampled people from seven European countries, and found that more than two thirds of the participants liked the idea, and would have no issues with buying urine-fertilised food. Another study from Ishii and Boyer (2016) highlights that, while 84% of the participants were in favour of the installation of source segregation systems in their residence, this number declined significantly in case they had to pay for it or contribute to the purchase themselves.

2.5.1 Survey on public acceptance of urine diverting toilets and urine based fertilisers:

As part of this report, a survey was carried out by a UCD master student under the supervision of Dr. Dereli, aimed at investigating the public acceptance and the main concerns associated with urine diverting toilets and urine based fertilisers. 171 people from both Ireland and the UK (70% male, 29% female, 0.5% other and 0.5% preferred not to say) took part in the survey, as well as 20 people working in the Irish agriculture sector (Samuel, 2021).

The participants were asked about their knowledge of UDTs, as well as about their willingness to use them. Information about these topics was then provided, allowing to assess if an increased awareness can facilitate the adoption of stream segregation systems. Only 5% of the participant had prior knowledge of UDTs and their design and function, and only 3% knew about the advantages of source separation to recover resources and avoid to release them into freshwater bodies.

When asked for a preference about the type of toilet to use if given the option between a standard toilet and a UDT, no clear preference emerged from the survey. However, after being informed about the benefits of source separation, 69% of the participants indicated UDTs as their choice, whereas only 9% selected standard toilets (Figure 5).

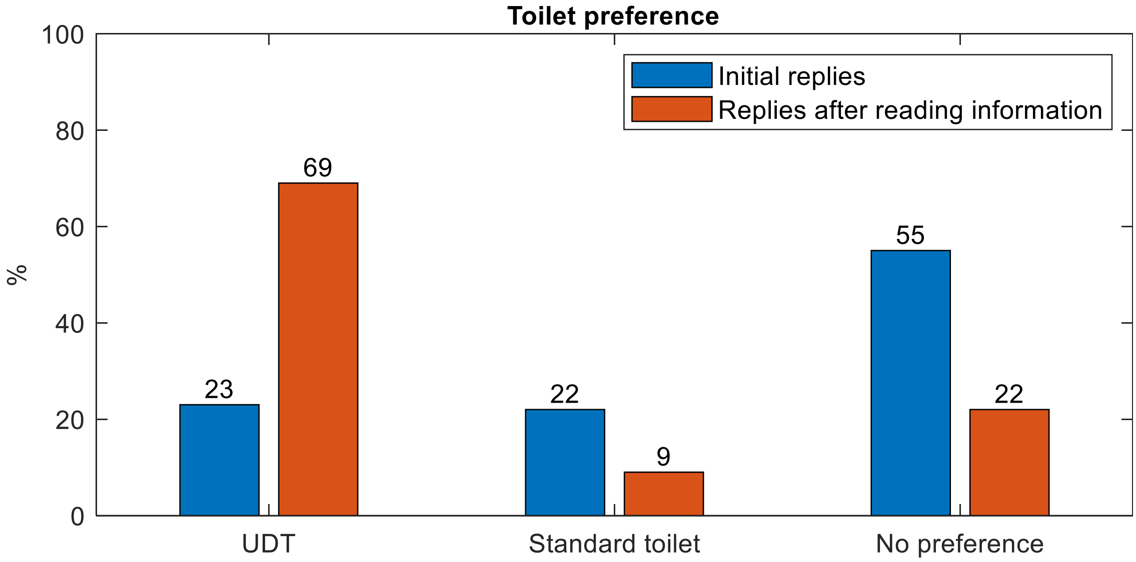

Figure 5 - Toilet preference before and after reading information about source segregation

After an explanation and description of UDTs, 76% of the participants thought they were a good idea, and the concerns associated with the adoption of UDTs are summarised in Figure 6, with increased necessity for cleaning considered as the main one.

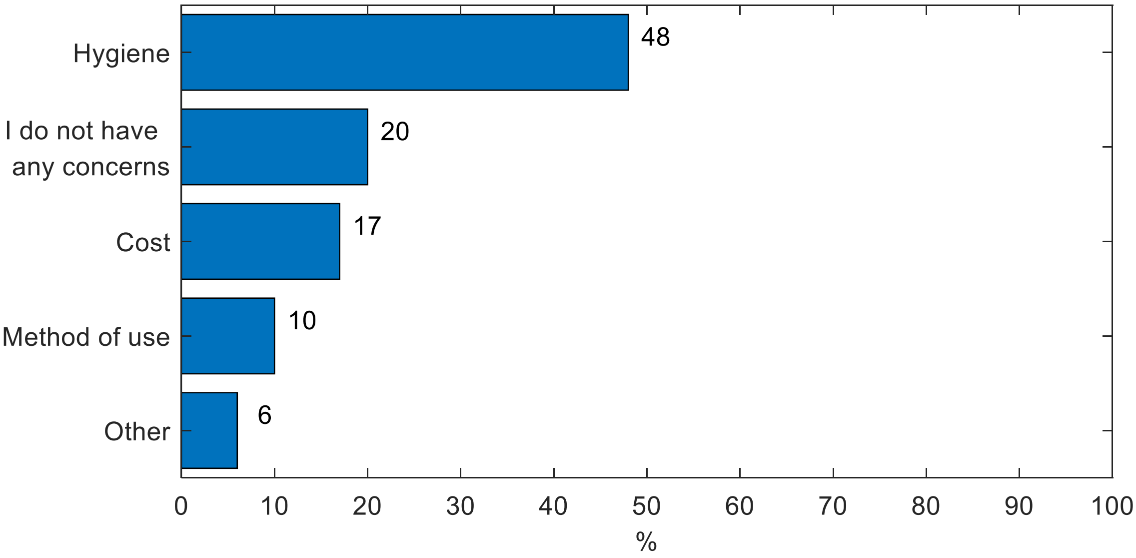

Figure 6 - Main concerns associated with the use of UTDs

While 85% of the participants were in favour of UDTs installation in the workplace, acceptance dropped to 58% for the place of residence, indicating that some perplexities still remain.

The survey also focused on the public acceptance of natural fertilisers, particularly on urine based fertilisers (UBFs) and animal waste. When asked whether they would purchase food produced using animal waste as fertiliser, 56% of the participants said they would (Figure 7), but only 29% stated that they would buy food obtained with UBFs.

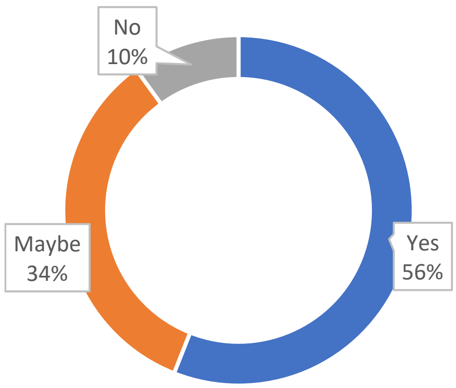

Figure 7 - Acceptance of animal waste as fertiliser

Once again, the survey results showed how providing people with more information can be a powerful tool to encourage the adoption of source segregation and urine based fertilisers. Figure 8 summarises the results in terms of willingness to buy food obtained from urine based fertilisers before and after having received informative materials about the benefits of this technology. The percentage of participants in favour of UBFs food increased to 44%, whereas the share of people against it went from 34% to 17%.

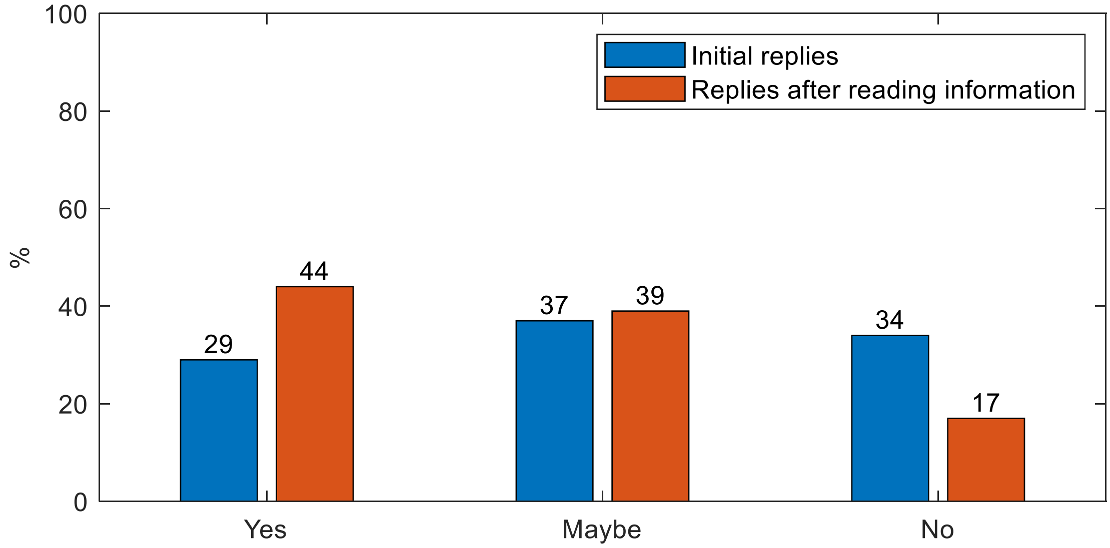

Figure 8 - Acceptance of urine-based fertilisers, comparison between the initial replies and the ones obtained after reading information material

The participants in the survey were also asked what their main concerns with the use of UBFs in food production were. 38% of the respondents indicated that they had no particular concerns, followed by hygiene (33%), odour (16%).

Within the same study, a similar survey was conducted targeting people working on the agriculture industry in Ireland, focusing on the acceptance of UBFs and the attitude towards UBFs and synthetic fertilisers. When asked about what typology of fertilisers they would prefer to use, 58% of the participants indicated a preference for natural fertilisers, whereas 42% preferred synthetic fertilisers. None of the 20 participants selected UBFs as their preference.

A summary of the main reasons for the choice of synthetic or natural fertilisers that emerged from the survey is reported in Table 5.

Table 5 - Main reasons for preference of natural and synthetic fertilisers

Synthetic fertilisers	Meets requirements Uniformity and availability Proven Most established, you know what you are getting Optimal plant growth and nutrient uptake
Natural fertilisers	Tried and tested Reliable Most sustainable Most traditionally used Prefer the organic nature of it

58% of the participants were aware of the potential for soil contamination that is associated with the use of synthetic fertilisers, and 67% had knowledge of the fact that non-renewable sources are necessary to produce this kind of fertilisers. When specifically asked about urine-based fertilisers, only 17% of the participants were aware of struvite as an option, with an additional 42% that was partially aware, which was not surprising given its limited application in Ireland. When compared to natural and synthetic fertilisers, the initial struvite acceptance was substantially less: only 27% of the participants would definitely use it, with an additional 18% that would probably accept it. After having read the informative material, 80% of the participants stated that they would probably or definitely accept to use it; furthermore, no respondent declared that they would not accept the use of struvite (Figure 9).

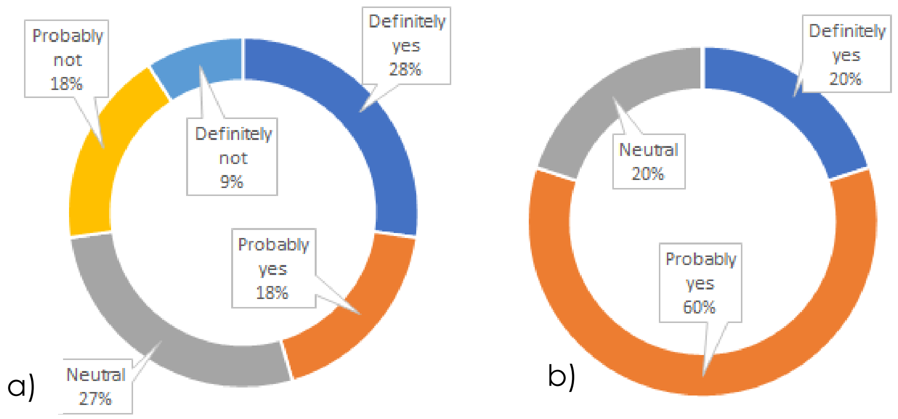

Figure 9 - Acceptance of UBFs utilisation in agriculture, before and after having read informative material (a and b, respectively).

Similar results were obtained when the survey participants were asked about whether they would consider purchasing UBFs, with 80% indicating that they would probably or definitely buy struvite fertilisers after reading the information.

It is also very interesting to observe how the main concerns associated with the utilisation of UBFs varied before and after providing the participants with information about struvite (Figure 10).

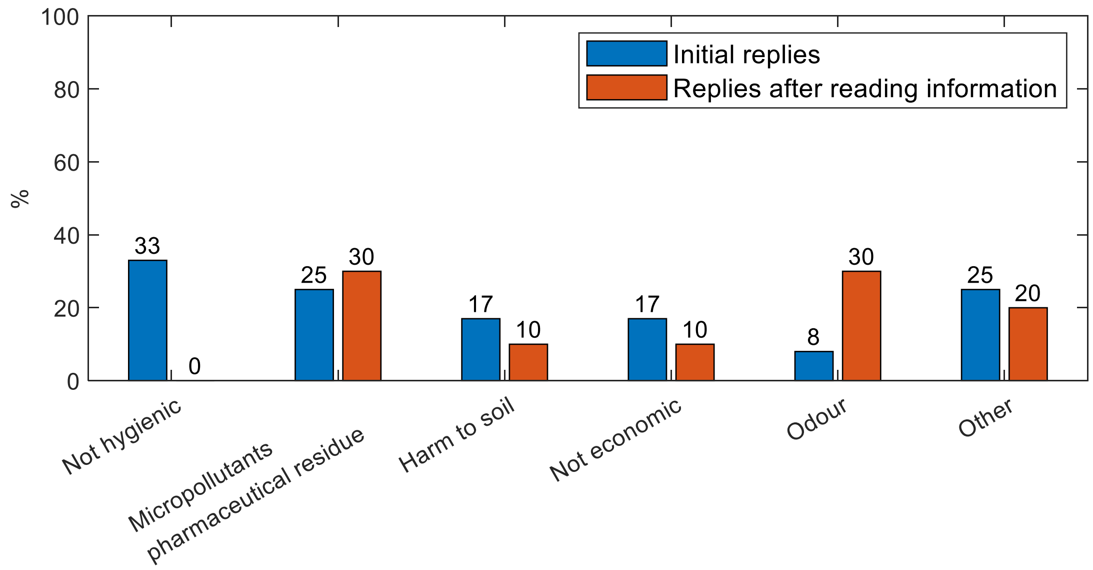

Figure 10 - Main concerns associated with the utilisation of UBFs in agriculture, before and after having read information documents

Once the benefits associated with the utilisation of struvite were highlighted, the importance of concerns in terms of hygiene and cost decreased significantly. However, the risks linked to the presence of micropollutants or pharmaceutical residue in the UBFs remained quite important, and concerns about odours became more significant, rising from 8 to 30%. One drawback of fertilising the soil using struvite can be its lower nitrogen content, which might require to combine struvite and other fertilisers. However, 80% of the participants stated they were comfortable with utilising a combination of fertilisers.

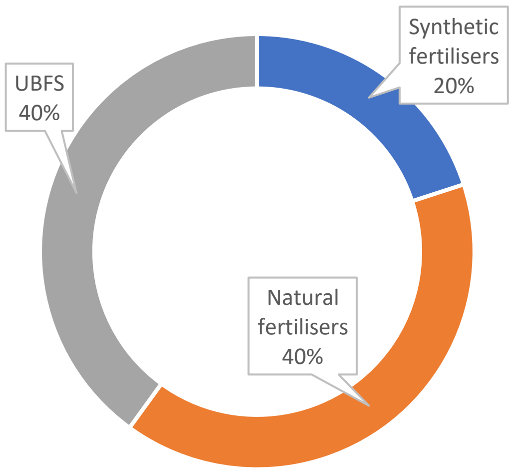

Figure 11 - Preference of fertiliser type after having read information document.

Finally, when asked for their preference of fertiliser type after having read the information document, 40% of the participants stated that they would consider struvite as their fertiliser of choice (Figure 11). However, it remains noteworthy to highlight that the sample size (20 people) might not be sufficient to ensure that the results are representative of the overall population, and further interviews might be necessary to improve the reliability of the survey.

In summary, the results of these surveys indicate that the willingness to adopt new, more sustainable practices depends on the public awareness of such technologies. This suggests that actions aimed at promoting and increasing knowledge and awareness can prove beneficial for a more widespread acceptance of nutrient recovery processes.

3 Commercial processes and case studies

As mentioned in the previous sections of this report, processes and technologies for the recovery of nutrients are widely available, relatively mature and well developed. While the principles of operation remain fairly similar, there is a variety of different configurations.

The Veas wastewater treatment plant (Oslo, Norway) recovers ammonium through stripping and adsorption, producing ammonium nitrate. The ammonium recovery efficiency is kept at around 88%, as to increase it would result in a significant increase in the energy consumption, which would not be proportionate to the increased yield of the process (Ye, Ngo, et al., 2018). Hadlocon, Manuzon, et al. (2014) describe a spray scrubber for the ammonia recovery from poultry facilities, which was capable of achieving high ammonia removal efficiencies (71-81%) and producing ammonium sulphate comparable to commercial grade nitrogen fertilisers. Moreover, a preliminary economic analysis indicated that the break-even time for the investment was one year.

Another example of full-scale nitrogen recovery is operated in Italy, in two plants treating 50 m^3/d and 100 m^3/d of digested cattle manure respectively. Through a series of physical/chemical treatments (N-Free® process) it is possible to produce up to 1.8 m^3 ammonium sulphate from every 100 m^3 of processed digestate. (Ledda, Schievano, et al., 2013).

In Japan, 16 full-scale plants are operated, producing both struvite and calcium phosphate through Gifu or PHOSNIX processes, and there is a strong collaboration between the steel, agriculture and chemical industries (Shaddel, Bakhtiary-Davijany, et al., 2019; Nättorp, Kabbe, et al., 2019).

Beckinghausen, Odlare, et al. (2020) illustrate that despite the large number of different options for nitrogen recovery, many of them still require a detailed energy and economic analysis before they can be successfully scaled up. However, a significantly larger number of commercial processes is available to combine nitrogen and phosphorous recovery in the form of struvite. A non-exhaustive summary of commercially available recovery processes is reported in Table 6.

Table 6 - Commercially available processes for the recovery of phosphorous: an overview (Melia, Cundy, et al., 2017; Siciliano, Limonti, et al., 2020).

Process	Details
AirPrex process	• Currently operational at several WWTPs in Germany and the Netherlands. • Struvite is crystallised directly in the digested sludge stream adding $MgCl_2$, stripping CO_2 to increase pH.
ANPHOS®	• Batch process: CO_2 is stripped in a first aerated reactor, and $MgCl_2$ is added in a second reactor to promote struvite formation. • 80-90% of the phosphorous can be recovered. • There is also a reduction in the influent COD content, which decreases DO consumption in subsequent treatments.
DHV Crystalactor®	• The sludge centrate/supernatant is fed to a reactor, where quartz sand is added as seed material to promote precipitation
NuReSys® process	• Air is used to strip CO_2 from the side stream. • $MgCL_2$ is fed to a stirred crystalliser tank (stirrer speed controls crystal size) • NaOH is used to regulate pH.
Ostara Pearl® process	• Sludge side stream is fed to a fluidised bed crystalliser. • $MgCl_2$ is used as source of magnesium, and NaOH to control pH. • There is a recycle of the crystalliser effluent, that is fed back to crystallisation
Phosnix® process	• There is a cylindrical reaction zone with a conical bottom section. • $Mg(OH)_2$ is used as source of Mg, and NaOH for pH regulation. • Aeration is used to strip CO_2. • Struvite settles at the bottom
PHOSPHAQ process	• Aeration is used to provide mixing, DO for the biological treatment and to strip CO_2 increasing pH. • MgO is the source of magnesium for struvite formation.
FIX-Phos	• Calcium silicate hydrate (CSH) particles are added to the anaerobic digester. • CSH adsorbs P, limiting unwanted struvite formation • Ca-P is then separated from the digested sludge
P-Roc®	• Similar to Crystalactor, but avoids pre-treatment such as CO_2 stripping and pH control. • Final products showed P content 11-13%, similar to phosphate rocks.
PHOXNAN	• Combination of low-pressure wet oxidation with two membrane filtration steps. • The final product is H_3PO_4.
Aqua Reci	• It uses supercritical water oxidation, and a base selectively dissolves P. • Phosphorous is precipitated with Ca.
Mephrec	• High temperatures (2000 °C) to melt the sludge • Oxygen is added, to destroy organic pollutants • Detoxified mineral P is obtained
SEABORNE	• Nutrients recovery from digested sludge pretreated with sulphuric acid. • Solid phase is incinerated • Gas liquid precipitation is used to recover heavy metals from the liquid phase using H_2 rich biogas. • Struvite recovery dosing $Mg(OH)_2$, and the residual ammonium is recovered as ammonium sulphate using a scrubber.

Among the commercially available processes listed above, the Pearl process developed by Ostara is particularly relevant for nutrient recovery in Ireland. This technology recovers phosphorous and ammonia from nutrient-rich streams, and converts them to struvite. It is currently operational at 22 WWTPs (Table 7), both in North America and Europe (Siciliano, Limonti, et al., 2020), with the earliest implementation commissioned in 2009 in Portland, USA and the largest facility capable of producing 30 tonnes of struvite per day (Gysin, Lycke, et al., 2018). Up to 22% of the total phosphorous can be recovered from urban sidestream wastewater, and up to 95% and 15% of P and NH_3-N, respectively, from digestion supernatants.

Table 7 - Examples of full scale implementations of Pearl® process (Gysin, Lycke, et al., 2018).

Plant location	Plant PE	Start of operation
Slough - UK	250 000	2012
Amersfoort - NL	500 000	2016
Madrid - ES	1 200 000	2016
Chicago - USA	2 300 000	2016
Portland - USA	500 000	2009

This process was selected as part of the larger upgrade plan for the Ringsend wastewater treatment plant in Dublin, which is responsible for the treatment of over 40% of Ireland wastewater. Ringsend plant was designed for 1.64 million PE, but is currently overloaded, receiving wastewater for approx. 1.98 million PE (with even larger peak flows). As part of the retrofitting plan, a reactor for struvite crystallisation is scheduled to be installed in the plant layout. This crystalliser is designed for the production of 14 tonnes of struvite per day (Ostara Inc., 2021).

4 Related policies

The number of full scale operational nutrient recovery facilities has been steadily increasing over the past decade, particularly in areas such as North America or Japan, although the EU is the region with the highest number of installed units (Shaddel, Bakhtiary-Davijany, et al., 2019). Strong, nationwide collaborative programs between industry, academia and government are in place in Japan, where strategies for integrated P – recovery have been developed since the 1980s (Nättorp, Kabbe, et al., 2019). Struvite production is also in place in North America, where more than 15 full scale plants are active, and dedicated policies and regulatory incentives are being discussed in Canada (D. Roy, R. Grosshans, M. Puzyreva, 2018).

Technology availability is not, however, the only factor that determines the adoption of nutrient recovery on large scale: a key driver for this paradigm shift is the implementation of environmental regulations that promote nutrient recovery even in those cases where the direct costs may be against it.

4.1 European policies

At European level, the Water Framework directive (European Commission, 2000) is the main legislation concerning water quality, setting the relevant standards, implementing discharge controls and fostering the minimisation of the anthropogenic pressures on water ecosystems. Although its implementation has been a major challenge, the Water Framework Directive sparked the harmonisation of monitoring methods across the EU. It also promoted the change from simply pollution control to protecting the ecosystems integrity, and significantly contributed to an increased knowledge of the ecology of European surface waters (Hering, Borja, et al., 2010). Monitoring rivers quality is important for several goals, from biodiversity analyses to research, but in the context of the Water Framework Directive the main objective is the identification of restoration and protection measures, which are then implemented through River Basin Management Plants (RBMPs).

However, despite the clear merits that this legislation has in terms of ecosystems protection, still a significant share of the EU surface water does not reach good ecological status (60% of the total, according to a 2018 report from the European Environment Agency). A more interdisciplinary approach which combines both natural and social research might be the key to address the complex environmental problems that are involved in water management (Voulvoulis, Arpon, et al., 2017).

The Wastewater Directive (European Commission, 1991) is the main regulation concerning the quality of treated municipal wastewater that is discharged in waterbodies across the EU. It defines the minimum treatment standards that wastewater treatment plants need to meet, based on the population equivalent, the category of the receiving waterbody and its risk of eutrophication. Countries such as Austria, Denmark, Lithuania or the Netherlands chose to consider all the waterbodies as at risk of eutrophication, whereas others (e.g. Italy, Germany, Ireland and the UK) only identified some areas as sensitive. On top of the European Directive, Member States can also decide to impose stricter national limitations. In this regard, an interesting example is offered by Denmark, where the introduction of a discharge tax on BOD_5, total nitrogen and total phosphorous is promoting the reduction of nutrients in the treated wastewater (Preisner, Neverova-Dziopak, et al., 2020). This approach follows the so called "polluter pays principle", and sets prices that the WWTPs must pay based on the amount of pollutants discharged (Table 8).

Table 8 - Tax rates for discharged pollutants in Denmark (Preisner, Neverova-Dziopak, et al., 2020)

Parameter	**Tax rate**
BOD_5	2.47 €/kg
Total nitrogen	4.44 €/kg
Total phosphorous	24.46 €/kg

The technical aspects of nutrient recovery processes have been the focus of the most part of the studies that are published on the topic (de Boer, Romeo-Hall, et al., 2018). However, the availability of processes and technologies that allow for the recovery and reuse of nutrients is not sufficient for their widespread adoption. There is often a delay between the development of a new process and the definition of the regulations that are applied to it. Moreover, despite the efforts to uniform regulations across the different countries within the European Union, disparities between member states can still hinder the application of novel processes. Different regulations applied in different countries mean that a technology that has successfully been installed in one country cannot necessarily be sold in another, or that struvite that is being produced in one state has to be sold as a fertiliser somewhere else.

This is particularly evident in the context of the utilisation of struvite recovered from wastewater as a fertiliser. According to a report from the European Sustainable Phosphorous Platform, in 2017 the use of struvite recovered from wastewater is only authorised in some EU states, such as Denmark. In the Netherlands, certain recovered phosphate (struvite, dicalcium phosphate and magnesium phosphate) are authorised as fertilisers, but they remain considered as waste. While this status does not prevent their application in agriculture, it is an obstacle to cross border trade and to use as a raw material for fertiliser production. It also means that recycling and recovering companies are labelled as waste management and have to follow far stricter rules than fertiliser companies that work with phosphate rocks (Hukari, Hermann, et al., 2016). Extra permits and installations are required for a WWTP to obtain the status of "fertiliser producer", which takes extra time and money. For these reasons, WWTPs often prefer to sell the recovered phosphorous as a waste rather than converting it into fertilisers (de Boer, Romeo-Hall, et al., 2018). There is however an additional requirement on the pathogens content of the recovered struvite, which has to be disinfected.

National or regional authorities have also granted a number of case-by-case authorisations, although these only apply to specific products, from specific waste streams, sites or processes. For instance, fertilisers produced from NuReSys process have been authorised in Belgium, and the use of struvite from Ostara processes is permitted in some plants in the UK and Germany (European Sustainable Phosphorous Platform, 2017). Moreover, phosphorous recycling legislation has been approved at national level in Switzerland, where it is mandatory to recover phosphorous from all sewage sludge and slaughterhouse waste. It is under examination in Germany, where it might target larger wastewater treatment plants, as well as in Austria and Sweden (European Sustainable Phosphorous Platform, 2022).

The lack of uniformity in the regulations is an obstacle to the cross-border trade of fertilisers produced from recovered nutrients: fertilisers can be registered as a waste in a country and as a product in one other, and waste and products are subject to different directives and regulations. Moreover, the registration of a substance as fertiliser can require up to 7 years, which hinders innovation (Hukari, Hermann, et al., 2016). On the other hand, supplying mineral P or nitrogen fixated via Haber-Bosch synthesis is generally easier than investing on recovered nutrients.

There is no comprehensive and binding soil conservation legislation at European level, but there are other directives that incorporate some soil conservation aspects, such as the Pesticide Directive and the Sewage Sludge Directive. In particular, the Sewage Sludge Directive (European Commission, 1986) sets limit values for concentration of heavy metals for the use of the sludge as fertilisers. Many Member States did however set stricter national limitations (e.g. Italy), or, like Switzerland and Germany, completely banned the direct use of sewage sludge (Garske, Stubenrauch, et al., 2020). There are also a number of regulations that address the deterioration of EU waterbodies, such as the EU Nitrates Directive, which mandates the implementation of good agricultural practices (e.g. allowing for the application of fertilisers only in certain time periods, and defining procedures for their utilisation). An equivalent legislation for P is still lacking at European level, although some countries such as the Netherlands and Ireland have established laws for phosphorous as well, where Ireland have included phosphorus in their Nitrate Action Programme to prevent further deterioration of water quality (European Commission, 2011).

Nutrient recovery in the EU is also affected by the EU Fertilising Product Regulation (European Commission, 2019), which aims to promote the use of fertilisers according to the circular economy model. This regulation will apply as of 16th July 2022, and will replace and update the current legislation (European Parliament, 2003), which does not include fertilisers produced from recovered or organic materials. Furthermore, this new framework eliminates the preference for inorganic fertilisers, incentivising the use of organic or recovered materials for the large-scale fertilisers production. Despite the acknowledgement in European and national legislation for the opportunities of nutrient recovery, there is however no dedicated support for recycled phosphorous fertilisers (Garske, Stubenrauch, et al., 2020). The updated regulation also establishes thresholds for inorganic macronutrients and heavy metals that fertilising products need to meet (Table 9).

Table 9 - Contaminant thresholds in organo-mineral fertilisers.

Element	Threshold values
Cd	60 mg/kg P_2O_5
Cr^{6+}	2 $mg/kd_{dry\ matter}$
Hg	1 $mg/kd_{dry\ matter}$
Ni	100 $mg/kd_{dry\ matter}$
Pb	120 $mg/kd_{dry\ matter}$
As	40 $mg/kd_{dry\ matter}$
$C_2H_5N_3O_2$	12 $mg/kd_{dry\ matter}$
ClO_4^-	50 $mg/kd_{dry\ matter}$
Cu	600 $mg/kd_{dry\ matter}$
Zn	1500 $mg/kg_{dry\ matter}$

In case the organic carbon content of the fertilisers is exceeding 1% of the mass, there are also additional requirements for the pathogens content that have to be met.

It is however noteworthy that more ambitious limitations, particularly in terms of Cadmium content, could have played an important role in increasing the competitiveness of alternative phosphorous fertilisers. Unlike mineral P fertilisers, which often have high levels of Cd that can ultimately contaminate soil, water and food, recycled P is generally characterised by lower Cadmium contents (Garske, Stubenrauch, et al., 2020).

Taxation on mineral P fertilisers, or limitations on the quantity that can be utilised could help tackle the issues related to the dependency on phosphate rocks import (Garske and Ekardt, 2021a). Taxes on the utilisation of phosphorous as fertilisers have been experimented in EU countries such as the Netherlands, Sweden, Finland, Denmark, Austria and Norway, aiming at decrease phosphorous usage on farms (Eckermann, Golde, et al., 2015).

Some significant measures for the recovery of nutrients and the conservation of soils and ecosystems can also be found on the EU Organic Farming Regulation (European Commission, 2007). In particular, this legislation sets limitations to the utilisation of mineral fertilisers as well as the ratio between livestock and land, and mandates sustainable cultivation practices such as crop rotations. This promotes the conservation of the soil organic matter and biodiversity, and an extension of the same rules to a larger number of farms could help supporting sustainable nutrients management (especially P) (Garske, Stubenrauch, et al., 2020).

Nutrient recovery should also be considered as a means to meeting the United Nations Sustainable Development Goals. Of the seventeen Sustainable Development Goals (SDGs), multiple SDGs relate to managing nutrients in wastewater. SDG 6, which is to “ensure availability and sustainable management of water and sanitation for all,” contains targets that aim to improve water quality by reducing pollution, halve the amount of untreated wastewater released to the environment, and increase recycling and safe reuse of wastewater (UN, 2017). SDG 2 seeks to improve food security and SDG 12 seeks to sustainably manage natural resources.

The European Commission adopted the new circular economy action plan (CEAP) in March 2020. It is one of the main building blocks of the European Green Deal, Europe’s new agenda for sustainable growth. The EU’s transition to a circular economy will reduce pressure on natural resources and will create sustainable growth and jobs. It is also a prerequisite to achieve the EU’s 2050 climate neutrality targets and to halt biodiversity loss. Measures that will be introduced under the new action plan aim to make sustainable products the norm in the EU.

While European policies play a key role in the adoption and implementation of nutrient recovery, it is necessary to consider that these remain global issues that should be tackled with global perspective. This is obvious for greenhouse gases such as N_2O, which is dispersed evenly in the atmosphere regardless the emission point. However, failing to adopt more ambitious environmental standards worldwide can result into a shift of the pollutant sources to regions where less strict regulations are applied (Bodirsky, Popp, et al., 2014).

4.2 Irish policies

The most recent EPA report (EPA, 2020a) paints a grim picture of the status of the Irish waterbodies. Despite the actions taken to reverse the trend, the overall water quality in Ireland is declining, and there is a growing number of waterbodies in poor ecological health. Nutrient pollution coming from agriculture has been individuated as the primary driver for this quality decline, with a call for tighter measures to be taken targeting fertiliser and slurry spreading and other nutrient losses. One third of rivers and lakes, and one quarter of estuaries already have too much nutrient in their waters. It is notable that the European Water Framework directive explicitly prohibits declines in ecological status. Of particular concern is the continuing decline in high status waterbodies, which has implications for the survival of protected species.

With the second National River Basin Management Plan, released in 2018, 1460 waterbodies were individuated as at risk of significant damage, and different measures were defined to promote their protection. In particular, the Local Authority Waters Programme was established to undertake water catchment assessments and develop actions plans. Another programme ASSAP (Agricultural Sustainability Support and Advisory Programme) was created with the goal of advising farmers on the most appropriate measures to tackle the water quality issues. The agricultural sector was also targeted with the Nitrate Action Programme, which was designed to prevent pollution of waters through agricultural sources setting limits for nitrogen and phosphorous application rates, prohibiting the application of organic and chemical fertilisers during more environmentally vulnerable periods and instituting set-back distances from waters. Roughly 3500 farm inspections were carried out every year under these regulations.

Actions were also defined to improve the collection and treatment of urban wastewater, financing 255 wastewater treatment projects and 41 collection system projects with €1.7 billion between 2017 and 2021. Quite significant is the high failure rate that emerged from septic tank inspections, which reached 48% in 2016 and remained at 50% between 2018 and 2021 (EPA, 2021). It is estimated that 1.4 million people is served by septic tank systems, which pose a substantial pressure on 11% of the waterbodies classified as "at risk". Since 2013, a programme has been in place to support remedial work on septic tanks that failed the inspection.

Significant progress is still required however to reach the environmental objectives set by the Water Framework Directive, and the new River Basin Management Plant 2022-2027 will have to face considerable challenges. Through the collaboration between all the public entities involved in the water sector, 527 areas were selected for interventions of water quality improvements or protections. Restoration works are planned for 427 areas that are currently not meeting the environmental objectives, whereas 85 areas were identified for protection measures to maintain their water quality. Furthermore, initiatives aimed at promoting the active involvement of a larger number of parties are scheduled.

A review of the wastewater discharge license is scheduled to be carried out by EPA, as well as update of the Nutrient Sensitive Areas that were designated under the Urban Wastewater Treatment Directive. Under the EPA, National Inspection Plan for Domestic Wastewater Treatment Systems, a minimum of 1000 inspections must be carried out annually. To mitigate the environmental impact of domestic wastewater discharges in rural communities, an EPA project launched in May 2021 is investigating the potential for zero discharge solutions, and will help inform future policies.

The RBMP also sets ambitious targets in terms of nutrients losses, such as a 50% reduction of the agricultural nitrate losses from high-risk areas, and the reduction of point source pollution from farms. Measures to limit the phosphate and sediment losses to surface waters are included in the plan. The existing controls on N and P from agriculture are set to remain in place, but additional measures such as the establishment of a chemical fertilisers register for the farmers, a stricter control on the use of chemical fertilisers and enhanced enforcement programmes are proposed.

With respect to the urban wastewater, Irish Water has planned an investment of €1.022 billion over 83 wastewater treatment plants and 10 collection networks. In terms of nutrient recovery, it is noteworthy that a struvite crystallisation system is under construction in the Ringsend wastewater treatment plant, as discussed in Section 3 of the present report.

With the publication of the Food Vision 2030 Strategy by the Irish Department of Agriculture, Food and the Marine, there has been a push for more sustainable practices in Ireland (Department of Agriculture, 2021). Among the targets set for 2030, a 50% reduction of the nitrous oxide emissions associated to chemical fertilisers as well as nutrient losses from agriculture can benefit from a more widespread implementation of nutrient recovery processes.

The importance of nutrient recovery has been highlighted by the Water Forum in a number of their recent policy submissions, which advocate for the inclusion of the agri-food sector in the circular economy. For instance, in the context of the Food Vision 2030, An Fóram recommended that an additional action should be added to Misison 1, Goal 6, to include the recycling of non-toxic phosphorous and nitrogen from municipal wastewater for use as fertiliser, which would also support the reduction of nutrients from municipal sewage treatment systems (An Fóram Uisce, 2021a). Additional recommendations were also extended on the Nitrate Action Programme, for instance suggesting a collaboration between Government departments and the lead partners of the ReNu2Farm project (ReNu2Farm, 2021) at Cork Institute of Technology to further develop best practices in how nutrient recovery and reuse can be implemented in Ireland (An Fóram Uisce, 2021b).

Analogous recommendations are also brought forward in the final report of the Phos4You project, an 11.02 million € study involving Cork Institute for Technology among the partner organisations. In their final report (Ploteau, Altoff, et al., 2021), it was advised to investigate a mandatory nutrient recovery and new legislation for Ireland, as it would improve downstream water quality and be consistent with the trend for management of P recovery in a European context. The importance of ensuring uniformity of the developed policies with the European legislation and practices was also highlighted.

Despite these efforts, more could be done in terms of recovery. As highlighted in a study from O'Donnell, Egan, et al. (2021), up to 1480 tonnes of P are lost through effluent from municipal wastewater treatment plants, and using existing P recovery technologies it could be possible to recover and reuse as P-rich fertilisers approximately 816 tonnes of phosphorous. Other indigenous waste streams from which P should be recovered are for instance dairy processing sludge, pig, cattle and chicken manure and dairy processing effluent. It is estimated that in total, 25548 tonnes of P could be obtained from these streams every year, with more than 19000 t that can potentially be recovered from cattle slurry alone. Prioritising recovery and redistribution of the indigenous P sources can help reduce the phosphorous deficit between soil requirements and the yearly importation.

5 Conclusions and recommendations

The current approach to the fertiliser production has significant negative effects on the natural nitrogen and phosphorous cycles. Increasingly larger quantities of these nutrients are introduced in the environment through municipal, agricultural and industrial sources, jeopardising the environment with the associated eutrophication and greenhouse gases emissions.

More sustainable solutions are required to address these issues while ensuring long-term soil fertility and food security. This is the main driving force that is leading to the development and adoption of nutrient recovery processes, and wastewater treatment can play a critical role in this transition. The current infrastructure is often failing to achieve sufficient levels of treatment, and while this presents significant challenges, it also offers substantial opportunities to reshape the entire sector. New technologies and processes are becoming available, and they are contributing to the paradigm shift from the nutrient removal perspective that was the main approach in wastewater treatment to nutrient recovery, particularly in terms of nitrogen and phosphorous. In this way, it is possible to both effectively reduce the amount of nutrients discharged in the environment, and to produce fertilisers that contribute reducing the European dependency on import.

Among the different possible systems, the recovery of N and P in the form of struvite has been particularly successful, with a number of full-scale installations in different parts of the world, from North America, to Japan, to Europe. In Ireland, nutrient recovery through struvite crystallisation is being installed in Ringsend.

Nutrient recovery technologies alone are however not sufficient. In some cases, it is necessary to ensure that they are accepted by the public. For instance, approaches such as stream segregation do not only require retrofitting of the infrastructure, but also behavioural modifications. In this regard, information provided to the people is critical to promote a wide acceptance, as highlighted by surveys conducted both among the general public and workers in the agricultural sector. Once the benefits of novel and sustainable approaches were explained, the acceptance of new technologies or processes can increase.

Regulations that favour the implementation of nutrient recovery technologies are also crucial, and can help promote their diffusion in many different ways. There is a huge untapped potential for nutrient recovery from various source materials, but they might be attributed different statuses in different countries. In this respect, it is essential that the legislation is uniformed and homogeneous across the EU, for instance by ensuring that materials are granted the status of products, which removes some obstacles to their trade across national borders.

Cadmium and uranium often contaminate mineral phosphorous fertilisers, and the updated European fertilising product regulation failed to impose lower limits. More ambitious limitations for cadmium and uranium in fertilising products can help promoting the adoption of recovered P based products, while enhancing environmental protection (Garske and Ekardt, 2021b).

A more sustainable nutrient management in Europe requires not only regulations, but also economic instruments. Regardless for the technology that is selected for the nutrient recovery, it is also necessary to account for the economical aspect of the process, assessing the associated costs and benefits (Yang, Shi, et al., 2017). It is equally important to find ways to factor in the indirect costs associated with the current approaches, for instance fostering the research in terms of good and reliable life cycle assessments of the impact of traditional fertilisers.

In order for alternative approaches to become attractive, one significant challenge is for them to compete with the availability and low cost of Haber-Bosch nitrogen and mineral phosphorous (Cruz, Law, et al., 2019). In this area, to offer incentives for the production or purchase of fertilisers obtained from recovered sources, or to impose penalties on the ones produced from conventional sources can have a significant impact. Economic policy instruments are a very promising way to promote the use of recycled nutrient based fertilisers. It could also be possible to envision a scheme analogue to the EU emission trading system, where the amount of extra nutrients that are allowed in a given area is limited, and recover and recycle are encouraged (Garske, Stubenrauch, et al., 2020). Subsidies had substantial benefits for the adoption of anaerobic digesters and for the production of biogas, and they can play a similar role in the context of nutrient recovery. Moreover, this sort of policies and economic instruments can encourage the private sector to undertake nutrient recovery as well, which has the potential to significantly develop the market. Modifications in the way the discharge limitations are designed can also be considered. For instance, the transition from the current approach, which is based on thresholds that should not be exceeded, to a system where the "polluter pays principle" is applied might prove beneficial in incentivising nutrient recovery from waste streams.

Nutrient recovery and re-use can also gain significant benefit from an increased allocation of resources for research and innovation, particularly if they are focused on farms.

It is important to remember that wastewater treatment traditionally involves a set of expertise and know-how that are quite different from those required to effectively market and sell the fertilising products obtained from recovered nutrients. The utilities that are operating the wastewater treatment plants do not necessarily have the capacity to deal with these new aspects, and dedicated training might be a tool to encourage the transition to water resource recovery.

Finally, a continuous monitoring and assessment of the effects of the implementation of new regulations and policies remains crucial, as they have the potential to trigger the shift of environmental problems from one sector to another. For instance, an increase in crops cultivation for bioenergy purposes might result into a more extensive application of industrial fertilisers, and it is also possible that without homogeneous legislation the burden for the ecosystems are merely passed on to countries with less restrictive environmental protection laws (Garske, Stubenrauch, et al., 2020).

The development of comprehensive and all-encompassing approaches is fundamental to ensure that the issues that involve how nutrients are dealt with over their entire life cycle are not simply shifted from one area to another. Both technological and policy related solutions are necessary to this goal, and can play an important role towards decreasing the quantity of nutrients that enter the natural P and N cycles.

6 Bibliography

An Fóram Uisce (2021a) *SUBMISSION TO THE DEPARTMENT OF AGRICULTURE, FOOD AND THE MARINE IN RELATION TO THE PUBLIC CONSULTATION PROCESS ON THE AGRI-FOOD STRATEGY 2030*,

An Fóram Uisce (2021b) *Submission to the Department of Housing, Local Government and Heritage on the Public Consultation on Stage 2 of the Nitrates Action Programme*,

Ashbolt, N. J. (2004) Microbial contamination of drinking water and disease outcomes in developing regions. Toxicology, **198**(1–3), 229–238.

Atafar, Z., Mesdaghinia, A., Nouri, J., Homaee, M., Yunesian, M., Ahmadimoghaddam, M., and Mahvi, A. H. (2010) Effect of fertilizer application on soil heavy metal concentration. Environmental Monitoring and Assessment, **160**(1–4), 83–89.

Bacelo, H., Pintor, A. M. A., Santos, S. C. R., Boaventura, R. A. R., and Botelho, C. M. S. (2020) Performance and prospects of different adsorbents for phosphorus uptake and recovery from water. Chemical Engineering Journal, **381**(August 2019), 122566. [online] https://doi.org/10.1016/j.cej.2019.122566.

Baffes, J. and Koh, W. C. (2021) Fertilizer prices expected to stay high over the remainder of 2021. [online] https://blogs.worldbank.org/opendata/fertilizer-prices-expected-stay-high-over-remainder-2021 (Accessed January 4, 2022).

Beckinghausen, A., Odlare, M., Thorin, E., and Schwede, S. (2020) From removal to recovery: An evaluation of nitrogen recovery techniques from wastewater. Applied Energy, **263**(February), 114616. [online] https://doi.org/10.1016/j.apenergy.2020.114616.

Bhuiyan, M. I. H., Mavinic, D. S., and Beckie, R. D. (2007) A solubility and thermodynamic study of struvite. Environmental Technology, **28**(9), 1015–1026. [online] https://doi.org/10.1080/09593332808618857.

Blank, L. M. (2012) The cell and P: From cellular function to biotechnological application. Current Opinion in Biotechnology, **23**(6), 846–851. [online] http://dx.doi.org/10.1016/j.copbio.2012.08.002.

Bodirsky, B. L., Popp, A., Lotze-Campen, H., Dietrich, J. P., Rolinski, S., Weindl, I., Schmitz, C., Müller, C., Bonsch, M., Humpenöder, F., Biewald, A., and Stevanovic, M. (2014) Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution. Nature Communications, **5**(May).

de Boer, M. A., Romeo-Hall, A. G., Rooimans, T. M., and Slootweg, J. C. (2018) An assessment of the drivers and barriers for the deployment of urban phosphorus recovery technologies: A case study of the Netherlands. Sustainability (Switzerland), **10**(6), 1–19.

Bond, I., Cornago, E., and Meyers, Z. (2021) *Why have Europe's energy prices spiked and what can the EU do about them?*, Bonvin, C., Etter, B., Udert, K. M., Frossard, E., Nanzer, S., Tamburini, F., and Oberson, A. (2015) Plant uptake of phosphorus and nitrogen recycled from synthetic source-separated urine. Ambio, **44**(2), 217–227.

Chavoshani, A., Hashemi, M., Amin, M. M., and Ameta, S. (2020) *Micropollutants and Challenges emerging in the aquatic environments and treatment processes*,

Consultative Communication on the Sustainable Use of Phosphorous (2013)

Cooper, J., Lombardi, R., Boardman, D., and Carliell-Marquet, C. (2011) The future distribution and production of global phosphate rock reserves. Resources, Conservation and Recycling, **57**(January), 78–86. [online] http://dx.doi.org/10.1016/j.resconrec.2011.09.009.

Cruz, H., Law, Y. Y., Guest, J. S., Rabaey, K., Batstone, D., Laycock, B., Verstraete, W., and Pikaar, I. (2019) Mainstream ammonium recovery to advance sustainable urban wastewater management. Environmental Science and Technology, **53**(19).

Cumming, O. (2009) The sanitation imperative: A strategic response to a development crisis. Desalination, **248**(1–3), 8–13. [online] http://dx.doi.org/10.1016/j.desal.2008.05.031.

Cusick, R. D., Ullery, M. L., Dempsey, B. A., and Logan, B. E. (2014) Electrochemical struvite precipitation from digestate with a fluidized bed cathode microbial electrolysis cell. Water Research, **54**, 297–306. [online] http://dx.doi.org/10.1016/j.watres.2014.01.051.

D. Roy, R. Grosshans, M. Puzyreva, and M. S. M. P. and M. S. (2018) *Nutrient Recovery and Reuse in Canada: Foundations for a national framework*, [online] https://iisd.org/event/national-nutrient-reuse-and-recovery-forum.

Daneshgar, S., Buttafava, A., Callegari, A., and Capodaglio, A. G. (2019) Economic and energetic assessment of different phosphorus recovery options from aerobic sludge. Journal of Cleaner Production, **223**, 729–738. [online] https://doi.org/10.1016/j.jclepro.2019.03.195.

Deng, Z., van Linden, N., Guillen, E., Spanjers, H., and van Lier, J. B. (2021) Recovery and applications of ammoniacal nitrogen from nitrogen-loaded residual streams: A review. Journal of Environmental Management, **295**, 113096. [online] https://doi.org/10.1016/j.jenvman.2021.113096.

Department of Agriculture, F. and the M. (2021) *Food Vision 2030*,

Desloover, J., Abate Woldeyohannis, A., Verstraete, W., Boon, N., and Rabaey, K. (2012) Electrochemical resource recovery from digestate to prevent ammonia toxicity during anaerobic digestion. Environmental Science and Technology, **46**(21), 12209–12216.

Dockhorn, T. (2009) "About the economy of phosphorus recovery" in International Conference on Nutrient Recovery from Wastewater Streams, 2009.

Eckermann, F., Golde, M., Herczeg, M., Mazzanti, M., Zoboli, R., and Speck, S. (2015) *Material resource taxation-an analysis for selected material resources*,

Emami, N., Sobhani, R., and Rosso, D. (2018) Diurnal variations of the energy intensity and associated greenhouse gas emissions for activated sludge processes. Water Science & Technology, (November), 1838–1850.

EPA (2020a) *Ireland's Environment*,

EPA (2021) Review of the National Inspection Plan for Domestic Waste Water Treatment Systems 2018-2021.

EPA (2019) *Urban Waste Water Treatment in 2019*,

EPA (2020b) *Urban Wastewater Treatment in 2020*,

EU (2017) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the 2017 List of Critical Raw Materials for the EU. Review of the List of Critical Raw Materials for the EU*, Brussels.

EU (2014) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Review of the List of Critical Raw Materials for the EU*, Brussels.

European Commission (1986) *Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture*, Brussels, European commission.

European Commission (1991) *Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment*, Brussels.

European Commission (2007) *Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91*, Brussels.

European Commission (2000) *Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for Community action in the field of water policy.*, Brussels.

European Commission (2019) *Regulation (EU) 2019/1009 — laying down rules on the making available on the market of EU fertilising products*, Brussels.

European Commission (2011) *REPORT FROM THE COMMISSION TO THE COUNCIL AND THE EUROPEAN PARLIAMENT on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports fo*, Brussels.

European Environment Agency (2015) Estimated consumption of manufactured fertilisers. [online] https://www.eea.europa.eu/data-and-maps/daviz/estimated-consumption-of-manufactured-fertilisers#tab-chart_1 (Accessed January 10, 2022).

European Environment Agency (2018) *European Waters - assessment of status and pressures 2018. EEA report no 7/2018*, European Parliament (2003) *Regulation (EC) no 2003/2003 of the European Parliament and of the Council of 13 October 2003 Relating to Fertilisers*, Brussels.

European Sustainable Phosphorous Platform (2022) ESPP regulatory activities. [online] https://phosphorusplatform.eu/activities/regulatory-activities (Accessed January 8, 2022).

European Sustainable Phosphorous Platform (2017) *Scope newsletter*,

FAO (2018) *World fertilizer trends and outlook to 2022*,

Fischer, F., Bastian, C., Happe, M., Mabillard, E., and Schmidt, N. (2011) Microbial fuel cell enables phosphate recovery from digested sewage sludge as struvite. Bioresource Technology, **102**(10), 5824–5830. [online] http://dx.doi.org/10.1016/j.biortech.2011.02.089.

Ganesapillai, M., Simha, P., Beknalkar, S. S., and Sekhar, D. M. R. (2016) Low-grade rock phosphate enriched human urine as novel fertilizer for sustaining and improving agricultural productivity of Cicer arietinum. Sustainable Production and Consumption, **6**(December 2015), 62–66. [online] http://dx.doi.org/10.1016/j.spc.2016.01.005.

Gao, Y., Liang, B., Chen, H., and Yin, P. (2018) An experimental study on the recovery of potassium (K) and phosphorous (P) from synthetic urine by crystallization of magnesium potassium phosphate. Chemical Engineering Journal, **337**(September 2017), 19–29. [online] https://doi.org/10.1016/j.cej.2017.12.077.

Garske, B. and Ekardt, F. (2021a) Economic policy instruments for sustainable phosphorus management: taking into account climate and biodiversity targets. Environmental Sciences Europe, **33**(1). [online] https://doi.org/10.1186/s12302-021-00499-7.

Garske, B. and Ekardt, F. (2021b) Economic policy instruments for sustainable phosphorus management: taking into account climate and biodiversity targets. Environmental Sciences Europe, **33**(1).

Garske, B., Stubenrauch, J., and Ekardt, F. (2020) Sustainable phosphorus management in European agricultural and environmental law. Review of European, Comparative and International Environmental Law, **29**(1), 107–117.

Gernaey, K. V., Flores-Alsina, X., Rosen, C., Benedetti, L., and Jeppsson, U. (2011) Dynamic influent pollutant disturbance scenario generation using a phenomenological modelling approach. Environmental Modelling and Software, **26**(11), 1255–1267. [online] http://dx.doi.org/10.1016/j.envsoft.2011.06.001.

Gifford, M., Liu, J., Rittmann, B. E., Vannela, R., and Westerhoff, P. (2015) Phosphorus recovery from microbial biofuel residual using microwave peroxide digestion and anion exchange. Water Research, **70**, 130–137. [online] http://dx.doi.org/10.1016/j.watres.2014.11.052.

Government of Ireland (2022) *Draft River Basin Management Plan for Ireland 2022-2027*,

Gurreri, L., Tamburini, A., Cipollina, A., and Micale, G. (2020) Electrodialysis applications in wastewater treatment for environmental protection and resources recovery: A systematic review on progress and perspectives. Membranes, **10**(7), 1–93.

Gysin, A., Lycke, D., and Wirtel, S. (2018) Phosphorus: Polluter and Resource of the Future - Removal and Recovery from Wastewater. Water Intelligence Online, **17**, 9781780408361.

Hadlocon, L. J. S., Manuzon, R. B., and Zhao, L. (2014) Development and evaluation of a full-scale spray scrubber for ammonia recovery and production of nitrogen fertilizer at poultry facilities. Environmental Technology.

Hamiche, A. M., Stambouli, A. B., and Flazi, S. (2016) A review of the water-energy nexus. Renewable and Sustainable Energy Reviews, **65**, 319–331. [online] http://dx.doi.org/10.1016/j.rser.2016.07.020.

Heinonen-Tanski, H., Sjöblom, A., Fabritius, H., and Karinen, P. (2007) Pure human urine is a good fertiliser for cucumbers. Bioresource Technology, **98**(1), 214–217.

Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C. K., Heiskanen, A. S., Johnson, R. K., Moe, J., Pont, D., Solheim, A. L., and de Bund, W. van (2010) The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. Science of the Total Environment, **408**(19), 4007–4019. [online] http://dx.doi.org/10.1016/j.scitotenv.2010.05.031.

Höglund, C., Ashbolt, N., Stenström, T. A., and Svensson, L. (2002) Viral persistence in source-separated human urine. Advances in Environmental Research, **6**(3), 265–275.

Howarth, R. W. (2008) Coastal nitrogen pollution: A review of sources and trends globally and regionally. Harmful Algae, **8**(1), 14–20.

Huang, H., Xiao, D., Zhang, Q., and Ding, L. (2014) Removal of ammonia from landfill leachate by struvite precipitation with the use of low-cost phosphate and magnesium sources. Journal of Environmental Management, **145**, 191–198. [online] http://dx.doi.org/10.1016/j.jenvman.2014.06.021.

Hukari, S., Hermann, L., and Nättorp, A. (2016) From wastewater to fertilisers - Technical overview and critical review of European legislation governing phosphorus recycling. Science of the Total Environment, **542**, 1127–1135. [online] http://dx.doi.org/10.1016/j.scitotenv.2015.09.064.

IFA (2021) IFA President Lambasts Commission for Failing to Address Fertiliser Crisis. [online] https://www.ifa.ie/campaigns/ifa-president-lambasts-commission-for-failing-to-address-fertiliser-crisis/ (Accessed November 11, 2021).

Ishii, S. K. L. and Boyer, T. H. (2016) Student support and perceptions of urine source separation in a university community. Water Research, **100**, 146–156. [online] http://dx.doi.org/10.1016/j.watres.2016.05.004.

Kabdaşli, I., Tünay, O., and Ozcan, P. (2009) Application of struvite precipitation coupled with biological treatment to slaughterhouse wastewaters. Environmental Technology.

Karak, T. and Bhattacharyya, P. (2011) Human urine as a source of alternative natural fertilizer in agriculture: A flight of fancy or an achievable reality. Resources, Conservation and Recycling, **55**(4), 400–408. [online] http://dx.doi.org/10.1016/j.resconrec.2010.12.008.

Köse, T. E. and Kivanç, B. (2011) Adsorption of phosphate from aqueous solutions using calcined waste eggshell. Chemical Engineering Journal, **178**, 34–39.

Kottegoda, N., Khalil, M., Madusanka, N., and Karunaratne, V. (2011) A green slow-release fertilizer composition based on urea-modified hydroxyapatite nanoparticles encapsulated wood Author (s): Nilwala Kottegoda , Imalka Munaweera , Nadeesh Madusanka and Veranja Karunaratne Published by : Current Science Association Sta. Current Science, **101**(1), 73–78.

Lahav, O., Telzhensky, M., Zewuhn, A., Gendel, Y., Gerth, J., Calmano, W., and Birnhack, L. (2013) Struvite recovery from municipal-wastewater sludge centrifuge supernatant using seawater NF concentrate as a cheap Mg(II) source. Separation and Purification Technology, **108**, 103–110. [online] http://dx.doi.org/10.1016/j.seppur.2013.02.002.

Langergraber, G. and Muellegger, E. (2005) Ecological Sanitation - A way to solve global sanitation problems? Environment International, **31**(3), 433–444.

Larsen, T. A., Lienert, J., Joss, A., and Siegrist, H. (2004) How to avoid pharmaceuticals in the aquatic environment. Journal of Biotechnology, **113**(1–3), 295–304.

Ledda, C., Schievano, A., Salati, S., and Adani, F. (2013) Nitrogen and water recovery from animal slurries by a new integrated ultrafiltration, reverse osmosis and cold stripping process: A case study. Water Research, **47**(16), 6157–6166. [online] http://dx.doi.org/10.1016/j.watres.2013.07.037.

Ledezma, P., Kuntke, P., Buisman, C. J. N., Keller, J., and Freguia, S. (2015) Source-separated urine opens golden opportunities for microbial electrochemical technologies. Trends in Biotechnology, **33**(4), 214–220.

Levin, G. V. and Sala, U. Della (1987) Phostrip® Process — a Viable Answer To Eutrophication of Lakes and Coastal Sea Waters in Italy. Biological Phosphate Removal from Wastewaters, 249–259.

Levlin, E. and Hultman, B. (2003) Phosphorus recovery from phosphate rich side-streams in wastewater treatment plants. Polish Swedish seminar, Gdansk March, 47–56. [online] http://www2.lwr.kth.se/Forskningsprojekt/Polishproject/JPS10s47.pdf.

Li, B., Huang, H. M., Boiarkina, I., Yu, W., Huang, Y. F., Wang, G. Q., and Young, B. R. (2019) Phosphorus recovery through struvite crystallisation: Recent developments in the understanding of operational factors. Journal of Environmental Management, **248**(May), 109254. [online] https://doi.org/10.1016/j.jenvman.2019.07.025.

Lienert, J. and Larsen, T. A. (2010) High acceptance of urine source separation in seven European countries: A review. Environmental Science and Technology, **44**(2), 556–566.

Liu, R. and Lal, R. (2014) Synthetic apatite nanoparticles as a phosphorus fertilizer for soybean (Glycine max). Scientific Reports, **4**, 1–6.

M.P.C.A (2006) *Phosphorus Treatment and Removal Technologies*, [online] http://www.pca.state.mn.us/index.php/view-document.html?gid=7095.

Maaß, O., Grundmann, P., and Von Bock Und Polach, C. (2014) Added-value from innovative value chains by establishing nutrient cycles via struvite. Resources, Conservation and Recycling, **87**, 126–136. [online] http://dx.doi.org/10.1016/j.resconrec.2014.03.012.

Magid, J., Eilersen, A. M., Wrisberg, S., and Henze, M. (2006) Possibilities and barriers for recirculation of nutrients and organic matter from urban to rural areas: A technical theoretical framework applied to the medium-sized town Hillerød, Denmark. Ecological Engineering, **28**(1), 44–54.

Mannina, G., Ekama, G., Caniani, D., Cosenza, A., Esposito, G., Gori, R., Garrido-Baserba, M., Rosso, D., and Olsson, G. (2016) Greenhouse gases from wastewater treatment - A review of modelling tools. Science of the Total Environment, **551**–**552**, 254–270. [online] http://dx.doi.org/10.1016/j.scitotenv.2016.01.163.

Melia, P. M., Cundy, A. B., Sohi, S. P., Hooda, P. S., and Busquets, R. (2017) Trends in the recovery of phosphorus in bioavailable forms from wastewater. Chemosphere, **186**, 381–395. [online] http://dx.doi.org/10.1016/j.chemosphere.2017.07.089.

Metson, G. S., Bennett, E. M., and Elser, J. J. (2012) The role of diet in phosphorus demand. Environmental Research Letters, **7**(4).

Modin, O. and Gustavsson, D. (2014) Opportunities for microbial electrochemistry in municipal wastewater treatment--an overview. Water Science & Technology, 2014.

Mondor, M., Ippersiel, D., Lamarche, F., and Masse, L. (2009) Fouling characterization of electrodialysis membranes used for the recovery and concentration of ammonia from swine manure. Bioresource Technology, **100**(2), 566–571. [online] http://dx.doi.org/10.1016/j.biortech.2008.06.072.

Nagy, J., Mikola, A., Pradhan, S. K., and Zseni, A. (2019) The utilization of struvite produced from human urine in agriculture as a natural fertilizer: A review. Periodica Polytechnica Chemical Engineering, **63**(3), 478–484.

Nättorp, A., Kabbe, C., Matsubae, K., and Ohtake, H. (2019) "Development of Phosphorus Recycling in Europe and Japan BT - Phosphorus Recovery and Recycling" in H. Ohtake and S. Tsuneda (eds.), Singapore, Springer Singapore, 3–27. [online] https://doi.org/10.1007/978-981-10-8031-9_1.

O'Donnell, C., Egan, A., Harrington, J., Barnett, D., Forrestal, P., and Power, N. (2021) An overview on deficit and requirements of the Irish national soil phosphorus balance. Science of the Total Environment, **785**.

ONU (2019) *World population prospects 2019*, Ostara Inc. (2021) Ostara and Murphy partner to deliver part of Ringsend Wastewater Treatment Plant Upgrade Project for Irish Water.

Pikaar, I., Matassa, S., Rabaey, K., Bodirsky, B. L., Popp, A., Herrero, M., and Verstraete, W. (2017) Microbes and the Next Nitrogen Revolution. Environmental Science and Technology, **51**(13), 7297–7303.

Ploteau, M.-E., Altoff, A., Nafo, I., and Teichgraber, B. (2021) *Final Report of the Phos4You partnership: deploying phosphorous recycling from wastewater in North-West Europe.*,

Prakash, K. H., Kumar, R., Ooi, C. P., Cheang, P., and Khor, K. A. (2006) Apparent solubility of hydroxyapatite in aqueous medium and its influence on the morphology of nanocrystallites with precipitation temperature. Langmuir, **22**(26), 11002–11008.

Preisner, M., Neverova-Dziopak, E., and Kowalewski, Z. (2020) An Analytical Review of Different Approaches to Wastewater Discharge Standards with Particular Emphasis on Nutrients. Environmental Management, **66**(4), 694–708.

Qu, D., Sun, D., Wang, H., and Yun, Y. (2013) Experimental study of ammonia removal from water by modified direct contact membrane distillation. Desalination, **326**, 135–140. [online] http://dx.doi.org/10.1016/j.desal.2013.07.021.

Quist-Jensen, C. A., Macedonio, F., and Drioli, E. (2015) Membrane technology for water production in agriculture: Desalination and wastewater reuse. Desalination, **364**, 17–32. [online] http://dx.doi.org/10.1016/j.desal.2015.03.001.

Regmi, P., Miller, M., Jimenez, J., Stewart, H., Johnson, B., Amerlinck, Y., Volcke, E. I. P., Arnell, M., García, P. J., Maere, T., Torfs, E., Vanrolleghem, P. A., Miletić, I., Rieger, L., Schraa, O., Samstag, R., Santoro, D., Snowling, S., and Takács, I. (2019) The future of WRRF modelling - Outlook and challenges. Water Science and Technology, **79**(1), 3–14.

ReNu2Farm Project summary (2021)

Rittmann, B. E., Mayer, B., Westerhoff, P., and Edwards, M. (2011) Capturing the lost phosphorus. Chemosphere, **84**(6), 846–853. [online] http://dx.doi.org/10.1016/j.chemosphere.2011.02.001.

Rouwenhorst, K. H. R., Krzywda, P. M., Benes, N. E., Mul, G., and Lefferts, L. (2021) *Ammonia Production Technologies*, Elsevier Inc. [online] http://dx.doi.org/10.1016/B978-0-12-820560-0.00004-7.

Samuel, M. (2021) Recovery of Struvite from Urine to be used as Slow Releasing Fertiliser in Agriculture: Feasibility Study and Implementation.

Santoro, C., Ieropoulos, I., Greenman, J., Cristiani, P., Vadas, T., Mackay, A., and Li, B. (2013) Power generation and contaminant removal in single chamber microbial fuel cells (SCMFCs) treating human urine. International Journal of Hydrogen Energy, **38**(26), 11543–11551.

Schoumans, O. F., Bouraoui, F., Kabbe, C., Oenema, O., and van Dijk, K. C. (2015) Phosphorus management in Europe in a changing world. Ambio, **44**(2), 180–192.

Seneca, J. (2022) The disruptive opportunity for mainstreaming urine recycling. The Source Magazine. [online] https://www.thesourcemagazine.org/the-disruptive-opportunity-for-mainstreaming-urine-recycling/.

Shaddel, S., Bakhtiary-Davijany, H., Kabbe, C., Dadgar, F., and Østerhus, S. W. (2019) Sustainable sewage sludge management: From current practices to emerging nutrient recovery technologies. Sustainability (Switzerland), **11**(12).

Shashvatt, U., Aris, H., and Blaney, L. (2017) *Evaluation of Animal Manure Composition for Protection of Sensitive Water Supplies Through Nutrient Recovery Processes*, Elsevier Inc. [online] http://dx.doi.org/10.1016/B978-0-12-809330-6.00013-1.

Shu, L., Schneider, P., Jegatheesan, V., and Johnson, J. (2006) An economic evaluation of phosphorus recovery as struvite from digester supernatant. Bioresource Technology, **97**(17), 2211–2216.

Siciliano, A., Limonti, C., Curcio, G. M., and Molinari, R. (2020) Advances in struvite precipitation technologies for nutrients removal and recovery from aqueous waste and wastewater. Sustainability (Switzerland), **12**(18).

Simha, P. and Ganesapillai, M. (2017) Ecological Sanitation and nutrient recovery from human urine: How far have we come? A review. Sustainable Environment Research, **27**(3), 107–116. [online] http://dx.doi.org/10.1016/j.serj.2016.12.001.

Soares, A. (2020) Wastewater treatment in 2050: Challenges ahead and future vision in a European context. Environmental Science and Ecotechnology, **2**, 10–13.

Taddeo, R. and Lepisto, R. (2015) Struvite precipitation in raw and co-digested swine slurries for nutrients recovery in batch reactors. Water Science & Technology.

Tao, Q., Zhou, S., Luo, J., and Yuan, J. (2015) Nutrient removal and electricity production from wastewater using microbial fuel cell technique. Desalination, **365**, 92–98. [online] http://dx.doi.org/10.1016/j.desal.2015.02.021.

Tao, W., Bayrakdar, A., Wang, Y., and Agyeman, F. (2019) Three-stage treatment for nitrogen and phosphorus recovery from human urine: Hydrolysis, precipitation and vacuum stripping. Journal of Environmental Management, **249**(August), 109435. [online] https://doi.org/10.1016/j.jenvman.2019.109435.

Tchobanoglous, G., Burton, F. L., and Stensel, H. D. (2003) *Wastewater Engineering: Treatment and Reuse. Metcalf & Eddy, Inc.*, [online] http://www.amazon.com/dp/007124140X.

Teagasc (2021) Projected higher costs to hit Tillage margins in 2022. [online] https://www.teagasc.ie/news--events/news/2021/projected-higher-costs-to.php (Accessed November 11, 2021).

Uysal, A., Demir, S., Sayilgan, E., Eraslan, F., and Kucukyumuk, Z. (2014) Optimization of struvite fertilizer formation from baker's yeast wastewater: growth and nutrition of maize and tomato plants. Environmental Science and Pollution Research.

Uysal, A. and Kuru, B. (2013) Magnesium ammonium phosphate production from wastewater through box-behnken design and its effect on nutrient element uptake in plants. Clean - Soil, Air, Water.

Vinnerås, B. and Jönsson, H. (2002) The performance and potential of faecal separation and urine diversion to recycle plant nutrients in household wastewater. Bioresource Technology, **84**(3), 275–282.

Vitousek, P. M., Aber, J. D., Howarth, R. W., Linkens, G. E., Matson, P. A., Schindler, D. W., Schlesinger, W. H., and Tilman, D. G. (1997) Human Alterations of the Global Nitrogen Cycle: Sources and Consequences. Ecological applications, **23**(1), 1–62.

Voulvoulis, N., Arpon, K. D., and Giakoumis, T. (2017) The EU Water Framework Directive: From great expectations to problems with implementation. Science of the Total Environment, **575**, 358–366. [online] http://dx.doi.org/10.1016/j.scitotenv.2016.09.228.

De Vrieze, J., Smet, D., Klok, J., Colsen, J., Angenent, L. T., and Vlaeminck, S. E. (2016) Thermophilic sludge digestion improves energy balance and nutrient recovery potential in full-scale municipal wastewater treatment plants. Bioresource Technology, **218**(2016), 1237–1245. [online] http://dx.doi.org/10.1016/j.biortech.2016.06.119.

Wang, J., Ye, X., Zhang, Z., Ye, Z. L., and Chen, S. (2018) Selection of cost-effective magnesium sources for fluidized struvite crystallization. Journal of Environmental Sciences (China), **70**, 144–153. [online] https://doi.org/10.1016/j.jes.2017.11.029.

Westerhoff, P., Lee, S., Yang, Y., Gordon, G. W., Hristovski, K., Halden, R. U., and Herckes, P. (2015) Characterization, Recovery Opportunities, and Valuation of Metals in Municipal Sludges from U.S. Wastewater Treatment Plants Nationwide. Environmental Science and Technology, **49**(16), 9479–9488.

Winker, M., Tettenborn, F., Faika, D., Gulyas, H., and Otterpohl, R. (2008) Comparison of analytical and theoretical pharmaceutical concentrations in human urine in Germany. Water Research, **42**(14), 3633–3640.

Winker, M., Vinnerås, B., Arnold, U., and Muskulos, A. (2008) New Fertilizers from Advanced Wastewater Treatment : Their potential Values and Risks. 13th RAMIRAN International Conference, (May 2014), 236–241.

Yan, T., Ye, Y., Ma, H., Zhang, Y., Guo, W., Du, B., Wei, Q., Wei, D., and Ngo, H. H. (2018) A critical review on membrane hybrid system for nutrient recovery from wastewater. Chemical Engineering Journal, **348**(December 2017), 143–156. [online] https://doi.org/10.1016/j.cej.2018.04.166.

Yang, Y., Shi, X., Ballent, W., and Mayer, B. K. (2017) Biological Phosphorus Recovery: Review of Current Progress and Future Needs. Water Environment Research, **89**(12), 2122–2135.

Ye, Y., Ngo, H. H., Guo, W., Chang, S. W., Nguyen, D. D., Zhang, X., Zhang, J., and Liang, S. (2020) Nutrient recovery from wastewater: From technology to economy. Bioresource Technology Reports, **11**(March), 100425. [online] https://doi.org/10.1016/j.biteb.2020.100425.

Ye, Y., Ngo, H. H., Guo, W., Liu, Y., Chang, S. W., Nguyen, D. D., Liang, H., and Wang, J. (2018) A critical review on ammonium recovery from wastewater for sustainable wastewater management. Bioresource Technology, **268**(June), 749–758. [online] https://doi.org/10.1016/j.biortech.2018.07.111.

Yuan, Z., Jiang, S., Sheng, H., Liu, Xin, Hua, H., Liu, Xuewei, and Zhang, Y. (2018) Human Perturbation of the Global Phosphorus Cycle: Changes and Consequences. Environmental Science and Technology, **52**(5), 2438–2450.

Yuan, Z., Pratt, S., and Batstone, D. J. (2012) Phosphorus recovery from wastewater through microbial processes. Current Opinion in Biotechnology, **23**(6), 878–883. [online] http://dx.doi.org/10.1016/j.copbio.2012.08.001.

Zhang, F. Z., Peng, Y., Miao, L., Wang, Z., Wang, S., and Li, B. (2017) A novel simultaneous partial nitrification Anammox and denitrification (SNAD) with intermittent aeration for cost-effective nitrogen removal from mature landfill leachate. Chemical Engineering Journal, **313**, 619–628. [online] http://dx.doi.org/10.1016/j.cej.2016.12.105.

Zhou, C., Liu, J., Wang, R., Yang, W., and Jin, J. (2010) Ecological-economic assessment of ecological sanitation development in the cities of Chinese Loess Plateau. Ecological Complexity, **7**(2), 162–169. [online] http://dx.doi.org/10.1016/j.ecocom.2009.10.001.