Source and central level recovery of nutrients from urine and wastewater: A state-of-art on nutrients mapping and potential technological solutions

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ABSTRACT

Wastewater treatment facilities typically transform nitrogen compounds into innocuous nitrogen. Phosphorous is partially recovered in biosolids while the potassium cycle in the treatment process is still not well mapped. This paper provides a mapping of the nutrients from household sources (urine, faeces and greywater) to wastewater in the traditional wastewater treatment plants. Following this mapping, the article critically reviews the nutrients recovery technologies, including their applicability at source (for urine) and centralised (for wastewater) levels, benchmarking, social acceptance, and techno-commercial assessments. Additionally, the paper has identified the current technological challenges and opportunities as well as suggested a several future recommendations for developing commercially viable nutrient recovery technologies from urine at source level and wastewater at wastewater treatment plants (centralised level). The high energy requirement, high cost and safety issues are some of the common challenges for most of the nutrient recovery technologies. However, anaerobic membrane bioreactor integrated with biochar or zeolite (for wastewater) and biosorption (for both urine and wastewater) can be highly attractive from those perspectives.

1. Introduction

The demand for a constant supply of fertiliser has augmented in the past few decades because of the industrialisation of farming [1]. Globally, the demand for nitrogen (N), phosphate (P) and potassium (K) fertilisers increases every year by 1.5% [2]. It is forecasted that there will be a shortfall of at least one of the NPK fertilisers in most regions of the world by 2022 [3]. The limited availability of phosphate and the high production price of nitrogen has led scientists to consider alternative routes for providing these essential agriculture nutrients.

Notably, nitrogen is a renewable resource; however, the production of nitrogen fertiliser using the Haber-Bosch process is significantly energy-intensive. Moreover, industrial production of nitrogen fertiliser is dependent on the limited supply of natural gas and given its high and fluctuating price, and it is necessary to consider other production methods. Potassium, so far, has not created a substantial economic impact on farming compared to nitrogen and phosphorous as it has more than 330 years of reserves at its current consumption rate [4]. In contrast, phosphorus is a finite resource and is considered a strategic commodity as there is no substitute for this element in the agricultural production [5]. Australia and the rest of the world are currently dependent on phosphorus from finite phosphate rock ores. These ores are concentrated only in a few countries and becoming costlier day by day. Phosphorus scarcity is likely to threaten cost-effective global food

production in the future. Australia has naturally phosphorus-deficient soils [6] and is substantially dependent on imported sources of phosphorus to maintain agricultural productivity. It means that the declining availability of phosphorus will inevitably threaten food production and its value as an export industry for Australia. Therefore, achieving a substantial improvement in the phosphorus balance of global agriculture, particularly in Australia, is essential for potential food production and environmental benefits. In this scenario, an integrated approach that recycles nutrients from multiple sources and finds innovative ways to recover them is highly merited.

Nutrients discharged in the form of human waste (such as urine, faeces and grey water) generated from household and industrial activities ends up in the wastewater which, if not treated to high standards, becomes a substantial source of nutrient release into natural water bodies such as rivers, lakes, and lagoons. The discharge of excessive nutrients due to their ineffective removal/recovery from the wastewater treatment plants (WWTPs) can promote algal growths and the proliferation of other aquatic plants, leading to eutrophication [7,8]. Studies noted that the build-up of phosphorus in the water distribution system could lead to gradual precipitation of struvite, causing blockages and equipment scaling [9,10]. To tackle the environmental pollution triggered by wastewater discharge, the European Union (EU) has issued discharge limits for total nitrogen and phosphorus from WWTPs (15 mg/L for nitrogen and 2 mg/L for phosphorus) [11,12]. Also, in Australia, the South Australia Environmental Protection Authority (EPA) guideline suggests that the total nitrogen discharge from a septic system must not exceed 5 mg/L and this value is 0.5 mg/L for phosphorus [13]. To date, the majority WWTPs around the world do not recover nutrients but mainly remove them from wastewater and minimise the negative impact on the environment. However, the research and policy paradigms have shifted from removing nutrients to their recovery in recent times [14,15]. This is because industrial and domestic wastewater and sludge are considered to be an untapped source of these nutrients and energy [16–19]. It is notable that total phosphorus content in human waste (urine and faeces) can supply approximately 22% of global phosphorus demand [20].

Nutrient recovery from human waste such as urine, faeces and greywater can be approached in two ways. (i) tackle them at the source level (i.e., at house, apartments, estate or suburb levels) or (ii) handle them at the central level (i.e. at the WWTPs). Urine contains the majority of wastewater nutrient load, and therefore, there is a significant interest to recover nutrients from urine at the source level. Both centralised and source separation options have their own benefits, bottlenecks and disadvantages [21,22]. Nutrient recovery from urine at the source level can significantly increase the life of WWTPs assets by minimising the need for nitrification and denitrification processes, resulting in lower energy demand for aerobic degradation of organic matters [23]. However, providing infrastructure at the source level for existing houses, apartments, the estate is a challenging task. For new houses, apartments, estate or suburb, this can be an attractive option; however, cost-effectiveness and suitable applications of nutrients are yet to be established. For central systems such as WWTPs, the nutrients are diluted to a greater extent, and there is a technological barrier to overcome for their recovery. Both areas require technology or process advancement/development, demonstrations and social research before they can be largely adopted into practice.

A large number of review articles on the nutrient's recovery from different sources of wastewater have been published in the current literature [24–35]. However, the published review papers mostly focussed either on a specific recovery technology [25,26] or a specific nutrient [24,26], highlighting fundamentals of the technologies [30], influential factors for nutrient recovery and associated challenges [27–29,31] with a very limited focus on benchmarking of nutrients (i.e., N, P and K) recovery technologies, their techno-economic and suitability analysis for WWTPs. To the best of the authors' knowledge, there is no review article which provides a systematic understanding of mapping of

Table 1
Amount of available nutrients and organic carbon in household wastewater [38].

	Urine		Faeces		Grey water[a]	
	g/person/day	%	g/person/day	%	g/person/day	%
Total nitrogen	11	82	1.5	11	1	7
Total phosphorus	1	61	0.5	30	0.15[b]	9
Potassium	2.7	74	0.43	12	0.5	14
Total organic carbon	5[c]	13	17	46	15	41

[a] Grey water means wastewater from kitchen, laundry and bathroom.
[b] Without phosphorus in detergents.
[c] Does not include carbon from urea.

Table 2
Concentration of key components in different urine.

Variable	Fresh urine	Hydrolysed urine	Stabilised urine[a]
Electrical conductivity (mS/cm)	160–270 [45]	26–41 [46]	
Osmolarity (mosmol/kg)	50–1200 [47]		
Chemical oxygen demand (mg/L)	6270–17,500 [48, 49]	4873–6174 [46]	5800 [50]
Total Nitrogen (mg/L)	4000–13,900 [51]	3846–6817 [46]	
Total Kjeldahl Nitrogen (mg/L)	5580–9220 [52]		
Urea (mg/L)	9300–23,300 [53]		5460 [50]
NH_4-N (mg/L)	125–600 [45]		
NH_3-N (mg/L)	200–730 [54]		419 [50]
Total Phosphorus (mg/L)	350–2500 [55]		
PO_4-P (mg/L)	205–760 [45,56]	85–178 [46]	<10 [50]
K (mg/L)	750–2610 [57]	1373–1604 [46]	348 [50]
Ca (mg/L)	32–230 [54]	5–7 [46]	1150 [50]
Mg (mg/L)	70–120 [54]	13–36 [46]	1 [50]
Creatine (mg/L)	0–890 [48]		
Creatinine (mg/L)	311–2150 [48]		
Uric acid (mg/L)	40–858 [58]		

[a] Urine stabilised using $Ca(OH)_2$.

nutrients and their fate in WWTPs together with their recovery using different processes and understanding the source and centralised level feasibility of each technology. Therefore, a state-of-the-art review focusing on the detailed mapping of nutrients and benchmarking of the recovery technologies in-line with their applicability at source and centralised level is essential to understand the scope for developing feasible nutrient recovery technology for WWTPs, which is the novelty of the current review. The current paper aims to critically review the literature on nutrient recovery from urine and wastewater, which include (i) mapping the nutrients from source to centralised levels, (ii) reviewing process/technologies critically for nutrients recovery including their advantages, disadvantages and their suitability, (iii) social acceptance studies, (iv) case studies on land application of urine and urine based fertilisers, (v) techno-economic analysis on currently available technologies/processes, and (vi) benchmarking and qualitative comparison of the reviewed technologies that can be applied at both source and central levels. The paper also discusses the opportunities and challenges for various nutrient recovery technologies and has identified key knowledge gaps and developed a set of future recommendations.

2. Source of nutrients in human waste

Although human urine accounts only 1% of wastewater entering the treatment plants, human urine is by far the largest contributor of nutrients to wastewater [36–38], as shown in Table 1. Several studies reported that approximately 80% of nitrogen, 50% of phosphorus, and
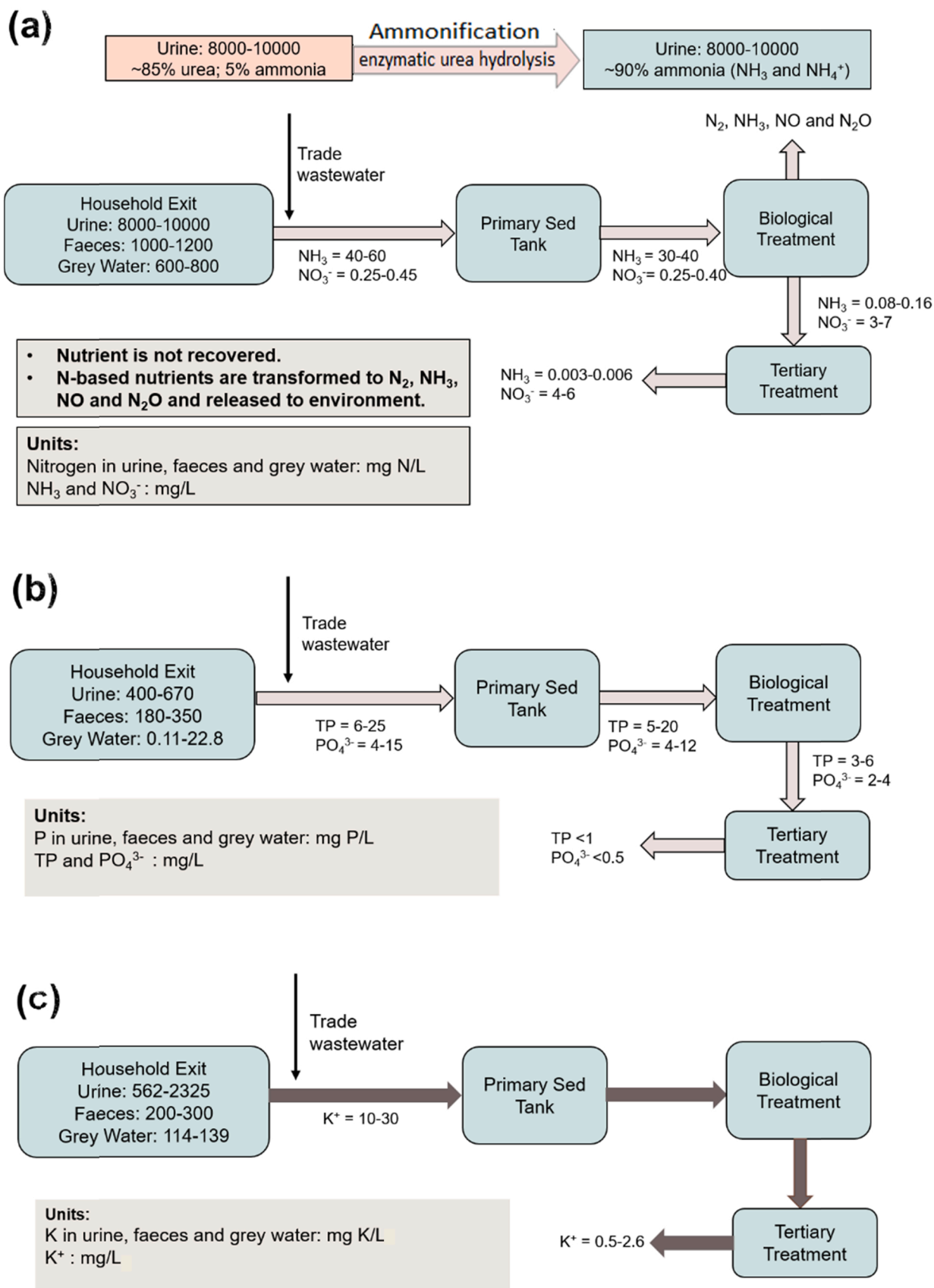
Fig. 1. Mapping of (a) nitrogen, (b) phosphorus and (c) potassium from household to wastewater. (Prepared from Refs. [59,71,73,74,76–78].

60% of potassium in wastewater generally come from urine [39–41]. In addition to nutrients, Rocha et al. identified a total of 294 metabolites present in human urine [42]. However, the concentrations of these metabolites are usually very low [43]. The composition of urine significantly varies depending on the method of urine collection and treatment as presented in Table 2. In principle, the recovery of nutrients is more feasible if urine is separated from the rest of the liquid waste streams in households such as kitchen and shower wastewater. It is notable that the separation of urine from the existing sewer network system is complex as it requires a lot of upgrading for the existing sewer network setup. For example, a dual pipe plumbing system may be one option, and this option is expensive or difficult to install in established properties. Therefore, this approach would be nearly impractical in established buildings and sewer network for source level nutrient recovery from urine. However, Flanagan and Randall recently constructed a novel fertiliser producing waterless urinal without any connection to the conventional sewage line [44]. The urine was collected into a tank below the urinal, where the urine mixes with pre-dosed $Ca(OH)_2$ powder and produces solid fertiliser, containing mostly calcium phosphate. Considering the laboratory data, it was estimated that 1000 male urinals could produce a daily profit of US$ 85. The nutrient recovery from urine at the source level for newly developed estates or suburbs can also be a pragmatic approach for a sustainable solution. The urine separation at source level can reduces nutrients load in wastewater stream at WWTPs and thereby reduce the cost and energy demand for nitrification and denitrification process as well as increase life of assets and operations. Apart from these, nutrient recovery at source level from urine can make additional revenue. However, management of odour as well as pathogens and pharmaceuticals in urine are some of the major aspects, which needs to be considered.

3. Nutrient mapping from household to wastewater

There are mainly four streams that form domestic wastewater: (i) urine stream, (ii) faces, (iii) greywater, and (iv) toilet flushing water [59]. Greywater comes from household sinks, showers, tubs and washing machines [60]. In addition to these four domestic wastewater streams, trade wastewater enters to the wastewater system. Toilet flushing water usually does not contain any nitrogen, phosphorus and potassium nutrients. However, all other streams contain these nutrients and create their load to WWTP.

The influent nitrogen load that enters to the WWTPs is mainly from urine (Fig. 1(a)). Urine contains urea that converts to ammonium nitrogen (NH_4-N) due to the presence of bacteria in urine [61]. These bacteria release urease enzyme that hydrolyse urea and forms ammonia (NH_3) gas, which causes a substantial amount of nitrogen loss to the atmosphere as NH_3. However, stabilisation of urine via alkalinisation and acidification can prevent the losses of NH_3 nitrogen. Both strong and weak acids, including sulphuric acid, acetic acid and citric acid can be used to lower the pH of fresh urine below 4, leading to the prevention of urea hydrolysis during the storage of urine [62–64]. On the contrary, the acidification of hydrolysed stored urine can maintain nitrogen in the form of NH_4 rather than NH_3 [63]. It is recommended that fresh urine requires approximately 60 mEq H^+/L, whereas hydrolysed stored urine requires about 10 times more acid for the acidification to store for 100 days [64,65]. Alkalinisation is another method of urine stabilisation, where various bases such as $Ca(OH)_2$, CaO or $Mg(OH)_2$ etc. have used to increase the pH of urine [50,66]. Alkalinisation of fresh urine with a pH above 10 inhibits the release of urease enzyme [67,68], resulting in the reduction of enzymatic hydrolysis, while the alkalinisation of hydrolysed urine enhances the release of NH_3. Randall et al. recommended to use approximately 10 g of $Ca(OH)_2$ in 1 L of fresh urine to maintain a good level of stabilisation [50].

The dominant form of nitrogen in the wastewater system is NH_4-N. Studies noted that 60–82% of total nitrogen in the influent is actually NH_4-N [69,70]. The average concentration of NH_4-N in the influent of

the WWTP is approximately 50 mg/L [59,71]. In the primary sedimentation tank, a portion of sludge is separated, and thus, the concentration of NH_4-N drops by 30% in the primary effluent, whereas the concentration of nitrate-nitrogen (NO_3-N) remains almost the same (0.37 mg/L in influent vs 0.34 mg/L in the primary effluent). In the biological treatment, the NH_4-N converts to N_2, NO, N_2O gases as well as NO_3-N. This is likely due to biological nitrification, assimilation and/or ammonia nitrogen (NH_3-N) loss that is caused by increased pH and the uncovered surface of the pond, promoting ammonia volatilisation [72]. Biological nitrification is unlikely in the anaerobic pond; however, NH_3-N removed by gaseous ammonia stripping or its assimilation in algal biomass can occur in anaerobic pond. The conversion of NH_4-N to NO_3-N clearly shows that ammonia is effectively reduced in the biological treatment processes. The NH_4-N reduces from 35 mg/L to 0.12 mg/L, whereas the concentration of NO_3-N increases from 0.34 mg/L to 5 mg/L after secondary treatment [59,71]. In most of the cases, tertiary treatment is ineffective to remove these nitrogen species.

Like nitrogen, a noticeable amount of phosphorus comes from urine to the WWTPs (Fig. 1(b)). Municipal wastewater has a high nutrient content with a typical total phosphorus concentration varying from 6 to 25 mg/L [73,74]. Usually, 15–40% of total phosphorus in municipal wastewater can be found in the particulate form [73], and some of them could be removed in the sedimentation tank of the WWTPs. Biological process could remove more than 40% of total phosphorus [73,75]. Notably, phosphate can be released from particulate into dissolved form during anaerobic treatment of municipal wastewater. This counterbalances the loss of some phosphate molecules through mineralisation, thus resulting in a consistent total concentration of PO_4^{3-} after anaerobic treatment. In contrast, PO_4^{3-} concentration could mainly reduce in the aerobic pond. Finally, tertiary treatment could also remove phosphate from the treating water.

Potassium is ubiquitous in wastewaters. A study by Arienzo et al. [79] noted that the average potassium concentration in sewage water was found to be 15 mg/L in New Mexico, 12 mg/L in Melbourne (Australia) and 15–30 mg/L in Alberta (Canada). A value of 24 mg/L of potassium in sewage water was reported in Israel [79]. Thus, it seems that potassium concentration in effluents from domestic wastewater sources are between 12 and 30 mg/L (Fig. 1(c)). A slight removal of potassium can be obtained in the sedimentation tank. Unlike nitrogen and phosphorus nutrients and organic matter, the fate of potassium in the biological processes of WWTPs is not properly mapped. In fact, the levels of potassium have received less attention in the tertiary treatment and are not reduced during typical tertiary treatment processes. In some cases, the concentration of potassium increases due to water evaporation from wastewater treatment and storage ponds.

4. Nutrient recovery technologies and their suitability

4.1. Ammonia stripping

Ammonia stripping is a physicochemical process where ammonia is stripped from wastewater by air/steam. This technology is mainly applicable at centralised WWTPs. A pilot scale ammonia stripping plant was installed in a dormitory at the University of CanTho in South Vietnam and tested for the nitrogen recovery as NH_3 from source separated urine, collected from no-mix toilet [80]. The stripping feed pump was run at 10 and 80 L/h flow rate. Another pilot-scale ammonia stripping plant of capacity 158 m^3 of urine was built in the Netherlands under the Dutch "Betuwse Kunstmest" project [81]. Based on the stripping fluid, stripping processes may be categorised mainly into two types: (i) air stripping in which ammonia is transferred to air followed by ammonia absorption in acid (e.g., sulphuric acid (H_2SO_4) producing ammonium sulphate (NH_4SO_4), a compound that can be used as a fertiliser), and (ii) steam stripping in which ammonia is stripped at elevated temperature by steam and subsequently collected as a liquid condensate which can be transformed to ammonium sulphate if it is intended to use

as a fertiliser [82,83]. Air stripping is commonly applied for removing ammonia, particularly, where the wastewater contains relatively high ammonia concentration (NH_4-N > 2000 mg/L).

Several factors such as ammonium concentration of feed, hydraulic loading, air flow rate, packing, and pH and temperature affect the ammonia stripping process (the last two are considered the most important parameters) [84–86]. Air stripping of ammonia is a pH-dependent process, where ammonium nitrogen is converted to ammonia gas at a pH value of approximately 9.3. Guo et al. reported that the recovery efficiency of ammonia increased from 80% to 92%, when pH increased from 8 to 11 [84]. Guštin and Marinšek-Logar [87] noted that the removal efficiency of ammonia was found to be 27% at pH 8.5, but it rapidly rose to 92%, when pH changed to 11. Several studies established that a pH value between 11 and 12 is economically favourable for optimum recovery of ammonia [85,86]. Bonmati and Flotats [88] obtained 87% recovery of ammonia at 80 °C temperature and at a pH of 9.5. They also noted that air stripping is independent of pH at a temperature greater than 80 °C. Guštin and Marinšek-Logar explored the effects of pH, air flow rate and temperature on ammonia recovery in a bench-scaled ammonia stripping column [87]. They found that the ammonia recovery was 80% and 92% at 40 °C and 70 °C, respectively. They observed that the airflow rate has an immense effect on ammonia recovery. For example, the recovery of ammonia was approximately 50% at the air to liquid (effluent) flow-rate ratio of 500, when the temperature and pH values were kept at 50 °C and 10, respectively. The recovery of ammonia increased to approximately 90%, when the air to liquid (effluent) was increased to 2000 maintaining other parameters same. The same study noted that pH levels have the most significant effect on ammonia removal. At a lower pH value of 8.5, only ~27% ammonia removal was observed while ~93% ammonia was removed, when pH was increased to 10.5. Using the optimised process, up to ~98% ammonia removal was achieved using ammonia stripping technology [87].

Steam stripping of ammonia is mainly dependent on the steam rate used for the process. Wickramanayake employed a pilot plant in the investigation of ammonia steam stripping from wastewater [89]. In that study, ammonia removal was increased from 93% to 99.9% when the steam to wastewater flow rates was raised from 155.77 to 455.34 g/l. It was also found that ammonia removal was 3 times higher when the pH of wastewater (containing 5000 mg/L $SO_4^=$ ions) was adjusted using caustic soda rather than lime. Tettenborn et al. focused on ammonia removal from stored urine using a steam stripping pilot plant designed to handle 800 people's urine [90]. Their ammonia removal was ranged from 91% to 100%. The ammonia removal rate in that study was impacted by several factors including pH (studied between 8.5 and 11, adjusted by caustic soda or caustic potash), urine flow rate (studied between 70 and 100 L/h) and steam flow rate (studied between 15 and 35 kg/h).

Both air stripping and stream stripping of ammonia have challenges as well as advantages. The challenges of ammonia air stripping system include: (i) relatively high operational costs due to the requirements of air and buffer solution, (ii) pre-treatment of feed requiring additional process unit (i.e., solid–liquid separation) leading to introduce additional cost, and (iii) production of spent waste stream requiring further treatment [306]. In the case of ammonia steam stripping, it requires considering the complexity of steam production [91]. The advantage of ammonia air stripping is that it is usually easy to use and maintain. Ammonia stream stripping's major advantage is that the off-gas vapour does not require any post-treatment and the vapour phase can be condensed into a liquid stream.

4.2. Nitrification/distillation

The nitrification/distillation technology is mainly suitable for nutrient recovery from urine at an estate or suburb levels. A pilot scale nitrification/distillation plant was tested in Switzerland to concentrate the nutrients from source separated urine as a part of VUNA project, funded by Bill and Melinda Gates Foundations [92]. The pilot scale nitrification reactor comprised of two columns of 120 L volume with a diameter of 32 cm. This technology recovers nutrients of urine and concentrates the nutrients to a very low volume (distillation). Nitrification has been integrated with other processes such as denitrification and anammox processes for nutrient recovery from wastewater. For instance, simultaneous partial nitrification, anammox and denitrification (SNAD) process with a photobioreactor (PBR) have been investigated for nutrient removal [93]. The SNAD-PBR process exhibited 90% and 100% removal efficiency of total nitrogen and phosphorus, respectively from synthetic wastewater. Salehi et al. also investigated simultaneous nitrification, denitrification phosphorus recovery (SNDPr) process for synthetic municipal wastewater treatment and reported about >70 mg/L phosphorus recovery in the phosphorus rich liquor [94]. Windey et al. reported about 84% nitrogen removal in laboratory-scale treatment of high-salinity wastewater employing oxygen-limited autotrophic nitrification–denitrification process [95]. Apart from these, denitrification-partial nitrification-anammox process has been tested at pilot-scale for landfill leachate treatment over a period of 166 days [96].

Nitrification comprises two bacterial groups including nitrite-oxidising and ammonia oxidising bacteria and therefore, it is considered as an aerobic biological process. Both bacterial groups' activities need to be well-balanced, since unbalancing may lead to nitrite accumulation and inhibit the nitrite oxidising bacteria. Ammonia in urine is oxidised to nitrate during the process of nitrification at pH less than 6. No significant ammonia loss occurs under such a low pH condition [97] during nitrification, though a significant amount of nitrogen is lost as NH_3 gas due to urine breakdown prior to nitrification process. The biological treatment of urine may involve heterotroph and nitrifying bacteria as well as tuning the dosing of alkalinity. This treatment degrades up to 90% organic substances [97]. In this process, the suspended biomass may be sedimented in a settler, and then the solution is to be transferred to a distillation reactor. It is noted that preliminary stabilisation via acid dosage or nitrification is useful to prevent high-ammonia loss from urine during the distillation process [98].

This technology recovers nutrients as ammonium nitrate (NH_4NO_3). Because of the hazardous explosive nature of this chemical, it is not attractive as a fertiliser. However, there are several potential applications of this chemical. The process is energy-intensive; however, the amount of energy invested to evaporate water during the distillation process can be recovered by many techniques, including heat exchanger and vapour compression. Nitrification/distillation technology requires more energy for the process, when treated urine with low ammonium concentration compared to highly ammonium concentrated urine [92]. Hence, at high ammonium concentration, this technology is economically favourable and therefore, this technology is mainly suitable for estate or suburb levels.

4.3. Precipitation

Precipitation of struvite ($MgNH_4PO_4·6H_2O$) to recover nutrients, particularly phosphorus from nutrient-rich waste streams, has received significant interest in the past decade [99,100]. This technology can be applied at a source and centralised level for phosphorus recovery from urine and wastewater, respectively. Precipitation of struvite at pilot-scale has been installed and tested in many countries around the world. GMB BioEnergie developed Europe's first urine processing facility (named SaNiPhos) of capacity 5000 m^3 urine per year in the Netherlands for struvite precipitation and nitrogen recovery [101]. Later in 2010, GMB BioEnergie installed Europe's first full-scale struvite precipitation and nitrogen recovery plant. Other pilot-scale struvite precipitation plant of capacity 158 m^3 of urine was developed in Tiel, the Netherlands [81] and in Leeuwarden, the Netherlands [102]. Additionally, pilot-scale facilities for struvite precipitation from source
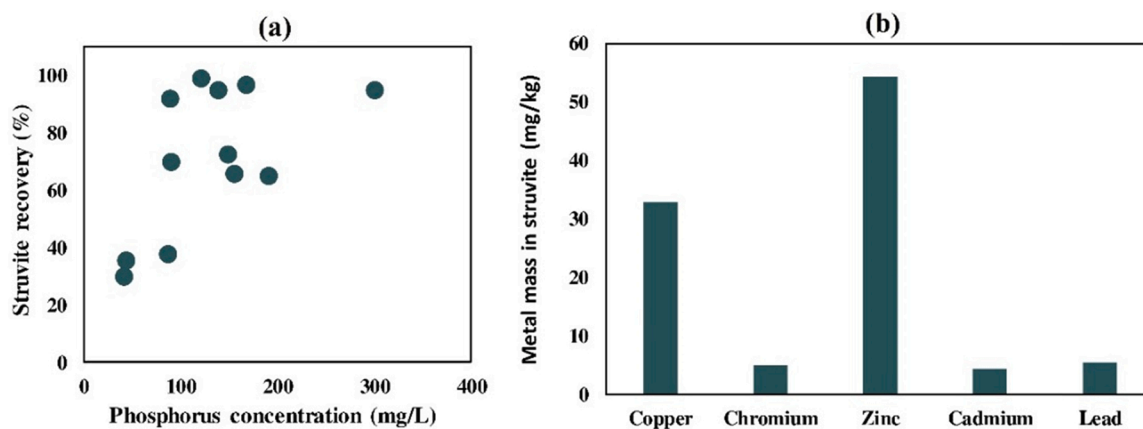
Fig. 2. Challenges in phosphorus recovery via struvite precipitation system: (a) Effects of phosphorus concentration on struvite precipitation efficiency (average data) and (b) presence of interference components in struvite precipitates (average data).
(a) (Prepared from Refs. [108,109,114,116,122,123,131–135]) and (b) (Prepared from Refs. [132,136–139]).

separated urine are also in operation in Nepal and Vietnam [80,103]. In this process, supersaturated conditions for precipitation are created in the waste streams either by adding a basic solution or aeration stripping of CO_2 and the supply of magnesium salts for magnesium-limited waste streams. Crystallisation of struvite also offers the removal of nitrogen simultaneously with phosphorus. Previous studies have demonstrated nutrients recovery via precipitation of struvite from several nutrient-rich streams including wastewater [104–106], anaerobically digested sludge [107–109], urine [110–112], swine manure [113–116], dairy manure [117,118], swine wastewater [119,120] and wastewater from fertiliser industry [121].

The phosphorus concentration limits the performance of struvite precipitation for nutrient recovery (Fig. 2(a)). The reaction kinetics for precipitation of struvite are substantially inclined by the phosphorus concentration in the feed stream. Studies have shown that effective precipitation of struvite could only be achieved, when phosphorus concentration in the feed stream is more than 100 mg/L [108,110,122, 123]. Low phosphorus concentration in the feed stream reduces struvite recovery performance and requires a longer induction time, leading to greatly reducing the economic feasibility of precipitation process for struvite recovery. The supply of high phosphorus concentration is challenging for wastewater since the typical phosphorus concentrations for influent wastewater is 6–25 mg/L [73,74,124]. When orthophosphate concentrations in the feed stream are low (<55 mg/L), the addition of supplementary phosphorus salts (e.g., H_3PO_4 and KH_2PO_4) are required for precipitation of struvite [125,126]. Among all the potential waste sources, manure and urine combinations might be an interesting choice as phosphate contents are significant in these sources, and a potentially effective phosphorus recovery could be possible via precipitation of struvite [110,127]. Human urine contains 350–2500 mg/L phosphorus [55,128,129], while poultry manure contains 370–600 mg/L phosphorus [130]. The high-level phosphate concentrations in these streams offer significant opportunities for their recovery processes to be integrated into existing wastewater treatment facilities.

Temperature, pH and mixing speeds of the solution influence parameters for the crystallisation process during struvite recovery scheme. It was reported that struvite solubility decreases from ~3000 mg/L to less than 100 mg/L with an increase of pH from 5 to 7.5 [140]. Bouropoulos and Koutsoukos found the maximum phosphorus removal efficiency at a pH range of 8.5–9.5 [141]. The pH range at which struvite can precipitate was identified to be approximately 7–11, with increasing driving force of precipitation as pH value increases [136,142]. Compared with pH profile, the temperature has less influence on the precipitation of struvite. The precipitation of struvite decreases at higher temperatures and a temperature range of 25–35 °C is usually chosen to

study struvite precipitation [143]. A different study suggested that temperature could also impact on struvite solubility and crystal morphology [144]. Turbulence during mixing the solution can also affect the precipitation of struvite. With an increase in turbulence, more CO_2 releases, leading to increased pH which favours struvite formation; however, high mixing speeds can enhance crystal breakage resulting in smaller crystals [145].

The precipitation of struvite is challenged by the presence of high suspended solids (>1000 mg/L) level in the waste stream [146]. The co-precipitation of calcium, ferric, aluminium [122,147] and organic compounds [134,148,149] significantly reduces the recovery and purity of struvite and inhibits the increase of crystal size due to blocking of sites, where crystals could be formed [150,151]. The presence of toxic heavy metal ions (Fig. 2(b)) and emerging organic contaminants in wastewater also significantly reduces struvite purity and impacts on its agricultural application [152]. Studies have shown the presence of toxic heavy metals in struvite crystals with arsenic concentration up to 570 mg/kg [153–155]. The presence of such contaminants in struvite fertiliser is being banned from the agricultural application. Hence, because of the presence of unwanted species, liquid matrix needs to undergo pre-treatment such as acid leaching, chelating agent (e.g. ethylenediaminetetraacetic acid, oxalic acid) treatment as well as microwave treatment prior to phosphorus recovery for minimising the inhibitory effect of struvite precipitation [156,157].

In the precipitation process, calcium phosphate can also be produced along with struvite precipitation. The precipitation of struvite, calcium carbonate and calcium phosphate are competitive processes. Therefore, nitrification prior to precipitation process removes ammonium and inorganic carbon, leading to the enhancement of calcium phosphate [158]. Monballiu et al. reported about 75–90% phosphate precipitation as calcium phosphate from the nitrified effluent of upstream anaerobic sludge blanket reactor [158–160]. Song et al. also observed the adverse effect of carbonate on calcium phosphate precipitation due to the formation of ion pairs between carbonate and calcium, leading to the reduction in the thermodynamic driving force for calcium phosphate precipitation. However, at pH values ≥ 9.00, no significant effect of carbonate was reported on calcium phosphate the precipitation [161]. Ferguson et al. investigated the effect of pH on the precipitation of calcium phosphate and found that the slightly alkaline condition enhanced the precipitation of calcium phosphate [162].

4.4. Electrochemical technologies

Electrochemical technologies such as electrochemical precipitation and microbial fuel cell have received significant interest for nutrient recovery from both source-separated systems and centralised WWTPs.

Tarpeh et al. demonstrated that electrochemical technology could remove 93% of nitrogen from source-separated systems [163]. Electrochemical struvite precipitation does not require the addition of magnesium salts, as magnesium metal serves as the anode in the system, participates in the reaction by providing electrons and forming soluble magnesium ions. Ben Moussa et al. confirmed the feasibility of struvite precipitation by electrolysis and found that struvite precipitation can occur in neutral environments [164]. This was likely due to a high local pH at the anode allowing for struvite crystal growth.

Microbial fuel cell (MFC) is one of the attractive electrochemical methods for nutrient recovery. In this process, organic matter is converted to electricity, which is generated by bacterial catalysis. MFC consists of a single or double compartment, where a cation exchange membrane separates anode and cathode. The anode compartment is filled with the organic feed, whereas the cathode compartment is filled with the catholyte. On the degradation of organics by the microbes, it produces electrons, hydrogen ions and carbon dioxide. Electrons and hydrogen ions are transferred to the cathode compartment by an external circuit and through the membrane, respectively. At the cathode compartment, oxygen serves as a terminal electron acceptor and when it combines with proton, it forms a water molecule [165]. The ammonium ions, generated from the hydrolysis of urea of urine, can be transferred to the cathode compartment via the cation exchange membrane. It converts to volatile ammonia due to an increased pH at the cathode compartment [166]. Previous studies noted that MFC could be a sustainable technology for the recovery of nitrogen and energy from urine or wastewater [167–169]. Although MFC showed a prospective application for nitrogen recovery, research on the recovery of nutrients in this technology is very limited. The pilot-scale MFC plants so far built have primarily focused on the treatment different wastewater including brewery wastewater and municipal wastewater for electricity generation [170–173]. The world's first MFC pilot-scale plant was established at Carlton United Breweries plant in Yatala, Australia in 2007 [174]. Therefore, further study needs to be done aiming to achieve complete recovery of nutrients with minimum energy and chemical requirements. The electrolysis process has the potential to recover nutrients due to its simplicity and high degradation rate per surface area and can be directly mounted into the toilet. The shortcomings of this technology include high-energy demand, expensive electrodes and the production of chlorinated by-products [175]. Although nitrogen removal rates may be high in electrolysis processes, the actual ammonia recovery is usually low compared to other technologies due to side reactions. Therefore, this technology may be suitable to apply in such a location, where the preventing environmental issues and hygiene are of the highest priority compared to nutrient recovery.

4.5. Ion exchange

A wide range of ion exchange materials has been studied extensively for the recovery of nutrients from different sources of the liquid stream. This process can be applied at both source and centralised levels. The principle of the ion-exchange process is the sorption of ions with a high internal surface area, and synthetically produced ion-exchange resins are usually functionalised copolymers of styrene and divinylbenzene. The ion-exchange mechanism stands on a simple electrostatic interaction. The affinity of the exchanged ions depends majorly on the size, charge and hydration degree of the exchanged ions. These factors significantly impact on the thermodynamics and kinetics of exchange reactions [176].

Studies found that clinoptilolite is a very effective ion exchanger for removing ammonium ions from wastewater [177] as clinoptilolite is a cation exchanger and ammonium is a cation. Beler-Baykal et al. [178] used clinoptilolite in an up-flow packed bed reactor for the removal of ammonium and potassium from urine. They found that clinoptilolite could remove 94% ammonium and 99% potassium. Jorgensen and Weatherley [179] noted that the performance loss of the clinoptilolite

was observed only 10% even after 10 cycles of regular operation and regeneration. They also found that this medium was successful for nitrogen removal at low nitrogen loading rates. Sendrowski and Boyer [180] used hybrid anion exchange (HAIX) resin for the recovery of phosphate. They found that the resin could separate phosphate from the urine up to 97% within less than 5 min only. Martin et al. showed that the HAIX resin was able to remove a significant proportion of the phosphate and eventually reached capacity after treating 1000 bed volumes [181]. There was a gradual decline in removal rates as the resin exchange sites become associated with phosphate. They also noted that there is no evidence to suggest that it exchanges other ions in preference to phosphate [181]. Tarpeh et al. recovered ammonium nitrogen from human urine employing the ion-exchange method [182]. In addition, these investigators suggested that the cost of urine treatment using the ion exchange method was 40% cheaper compared with disposing of human urine without treatment [163]. Liberti et al. proposed a combined anion and cation exchange scheme where both ammonium and phosphate ions were concentrated for recovering magnesium ammonium phosphate [183]. Lind et al. [184] used clinoptilolite, wollastonite and a natural zeolite to recover nitrogen and obtained 65–80% nitrogen recovery.

A limited number of synthetically produced ion-exchange resins, so far, has been studied. Further studies are required to enhance knowledge in this area. The primary operating cost is the use of acid for ammonia recovery and resin regeneration. When considering capital cost, a large-scale unit is economically beneficial and therefore, this technology is not expected to be suitable for households. However, detailed techno-commercial studies would help in comparing the data.

4.6. Adsorption

Adsorption is an effective technique that can be used at the source and centralised levels for nutrient recovery. The performance of adsorption technology depends on the selection of an appropriate adsorbent. The common criterion for selecting adsorbents are adsorption capacity, reuse, local availability, compatibility, kinetics, and cost. Common adsorbents for nutrient recovery are hydrotalcite, biochar, activated carbon, zeolite, resin, blast furnace slag, aluminium oxide, iron oxide, zirconium oxide, activated red mud and activated alumina. For instance, Terry used hydrotalcite for the effective removal of phosphorus as well as nitrogen [185]. However, the pH requirement is in the strongly acidic range (2.0–2.1), limiting the applicability of the process. Simha et al. studied the feasibility of microwave activated coconut shells for urea biosorption from human urine in a fixed-bed column [186]. They found that approximately 80% urea recovery can be achieved at optimal operating conditions. Previously, the feasibility of using activated carbon produced from coconut shell for removing urea from human urine was studied [187]. Kini and Hari used ground granulated blast furnace slag (GGBS) as an adsorbent for the adsorption of phosphate from human urine, and noted that more than 90% adsorption of phosphate can be achieved from 50% diluted human urine using 700 g/L of GGBS [188]. Li et al. employed MgO impregnated sugarcane crop harvest residue biochar for biosorption of phosphate and ammonium from real swine wastewater of a pig farm [189]. The study showed that one gram of MgO impregnated biochar captured about 398 mg phosphate and 22 mg ammonium from swine wastewater. Huggins et al. compared the phosphate and ammonium adsorption performance of granular activated carbon and lodgepole pine wood granular biochar from brewery wastewater [190]. Batch adsorption studies showed more than two times higher phosphate (1 mg/g) and ammonium (3.6 mg/g) adsorption by granular biochar compared to those by granular activated carbon, due to highly macroporous structure of biochar. Sooksawat et al. reported about 83.8% phosphate adsorption from swine wastewater using chemically modified rice husk biochar [191].

There are limited studies on nutrients adsorption from real urine and wastewater. The fate of adsorbed nutrients is yet to be established. The

Table 3
Summary of nutrient enrichment by emerging membrane processes.

Feed matrix	Technologies	Technologies condition	Reconcentrate times	Recovered nutrient	Water recovery	References
Digested sludge	FO	Crossflow FO cellulose triacetate (CTA) membrane, Seawater (draw solutions)	3-fold pre-concentration of digested sludge	Phosphate	80%	[134]
Municipal wastewater	FO	Crossflow FO CTA membrane, ten different organic and inorganic (draw solutions)	10-fold pre-concentration of wastewater	NA	90%	[203]
Municipal wastewater	FO	Flat sheet (TFC, CTA), NaCl draw solution	10-fold nutrient concentrated	Nitrogen, phosphorus	NA	[196]
Urine	FO	Flat sheet FO CTA membrane, NaCl Draw	NA	Nutrients (N, K, P)	NA	[195]
Urine	ED	Five cell pairs and a total voltage of 9 V, recirculation flow rate 65 L/h.	3.2-fold nutrient concentrated	NA	NA	[152]
Synthetic wastewater	OMBR	TFC flat sheet submerged FO membrane, NaCl	6-fold nutrient enrichment	Phosphorus	NA	[204]
Municipal wastewater	OMBR	CTA flat sheet submerged FO membrane with draw solution of $MgCl_2$ and NaCl.	NA	Phosphorus	NA	[205]
Digested sludge	FO+MD	Flat sheet (TFC, CTA), $MgCl_2$ draw solution PTFE direct contact membrane with 0.03 μm	NA	Phosphorus	NA	[206]
Raw sewage	MF-OMBR+RO	CTA flat sheet submerged FO membrane with NaCl draw solution, flat sheet polysulfone RO membrane, PVDF MF	NA	Phosphorus	NA	[207]
Municipal wastewater	MF-OMBR	CTA flat sheet submerged FO membrane with NaCl draw solution.	7-fold phosphate concentrated	Phosphorus	NA	[149]

FO: Forward osmosis, ED: Electrodialysis, OMBR: Osmotic membrane bioreactor, MD: Membrane distillation, RO: Reverse osmosis, MF: Microfiltration, TFC: Thin-film composite, PVDF: Polyvinylidene fluoride.

commercial viability of these processes will depend on the cost of the adsorbent as well as the operating cost. Additionally, adsorption technology for nutrient recovery is associated with the challenges of effective desorption of nutrient, which depends on the adsorbent properties as well as reusability of the adsorbent. Further studies will require to understand the techno-commercial perspective of these technologies, desorption kinetics and effectiveness as well as reusability of the adsorbent.

4.7. Membrane processes for the recovery of nutrients

Membrane processes can be applied at both source and centralised levels for efficient nutrient recovery. These processes are of diverse nature, and therefore, this section is divided into two subsections to provide a broad picture of these processes.

4.7.1. Emerging membrane technologies

Some examples of emerging membrane technologies include forward osmosis (FO), electrodialysis (ED), and membrane distillation (MD) that could be favourable in mining valuable nutrients from urine. FO works on the principle of chemical potential gradient. Due to the difference in the chemical potential of two different solutions, water transfers from high water chemical potential to low water chemical potential through the FO membrane. ED applies the influence of an electric potential for the transportation of ions through a semi-permeable membrane. MD utilises a hydrophobic microporous membrane for separating vapour phase from the feed stream. In this process, the feed solution should have a high temperature than that of the permeate to create a vapour pressure gradient across the membrane. Compared to conventional pressure-driven membrane processes, these membrane processes (FO, ED and MD) are expected to have lower energy consumption and lower fouling propensity due to low applied hydraulic pressure [192–194].

Recently, FO, ED and MD have been demonstrated with their unique mass transfer properties for facilitating valuable element recovery (Table 3). The application of the FO process for resource recovery has been shown with achieving high enrichment factors for different liquid streams. For example, Zhang et al. found that FO could concentre 3–4 times of the nutrients from urban source-separated urine [195]. Xue et al. [196] reported that there was a ten-fold concentration of ammonium and phosphate in the secondary treated municipal wastewater with water recovery of 93% in an FO process using seawater draw

solution. Ansari et al. [134] reported 92% phosphate recovery from digested sludge centrate by the FO process. Studies have been reported that the ED process selectively separated phosphate from wastewater effluent containing various ions and achieved a concentration factor of up to seven [197,198]. The ED process has noticed similar selective phosphate enrichment in urine nutrient recovery, resulting in a purified phosphate crystal [152,199]. Derese et al. employed MD and recovered up to 95% ammonia from hydrolised urea by increasing pH to 10.5 or higher [200]. An integrated treatment process, including ED with subsequent ozonation was investigated at pilot-scale in Switzerland [201]. The test of the plant was carried out at a flow rate of 40 L/d hydrolysed urine and produced fertiliser called "Urevit" (similar to ammonium nitrate fertiliser). Another pilot-scale ED plant integrated with precipitation system was installed at the Luggage Point WWTP in Brisbane, Australia [202].

Coupling FO with MD could re-concentrate diluted draw solution and supplement nutrient recovery with clean water production (Fig. 3). In an FO-MD hybrid system, FO focuses on concentrating orthophosphate and ammonium content for the recovery of phosphorus in the form of struvite ($MgNH_4PO_4 \cdot 6H_2O$), while MD focuses on the recovery of the draw solution and production of clean water from the digested sludge centrate [206,208]. The investigators noted that FO achieved a concentration factor of five by $MgCl_2$ draw solution, subsequently obtained a high strength waste stream comprising ammonium (1210 mg/L) and phosphate (615 mg/L). This $MgCl_2$ draw solution provides the driving force for nutrient enrichment and incorporates nutrient precipitation. Volpin et al. utilised FO and MD together to extract concentrated fertiliser as well as distilled water from human urine [209]. The recycled water from this hybrid method can be utilised on-site for irrigation or flushing toilets.

Integrating FO with a membrane bioreactor (MBR) is also becoming an attractive option for wastewater treatment and nutrient recovery, known as the osmotic membrane bioreactor (OMBR) [149,204,205, 210]. MBR processes, used in wastewater treatment plants, integrate a membrane filtration unit with a biological process such as a suspended growth bioreactor. Microfiltration (MF) and ultrafiltration (UF), commonly used membrane filtrations in MBR processes, have separation ranges 100–1000 nm and 5–100 nm, respectively. The OMBR concept provides several benefits including high nutrient rejection and supply of magnesium for facilitating nutrient precipitation by reverse draw solution diffusion. Qiu and Ting noted that OMBR could enrich the
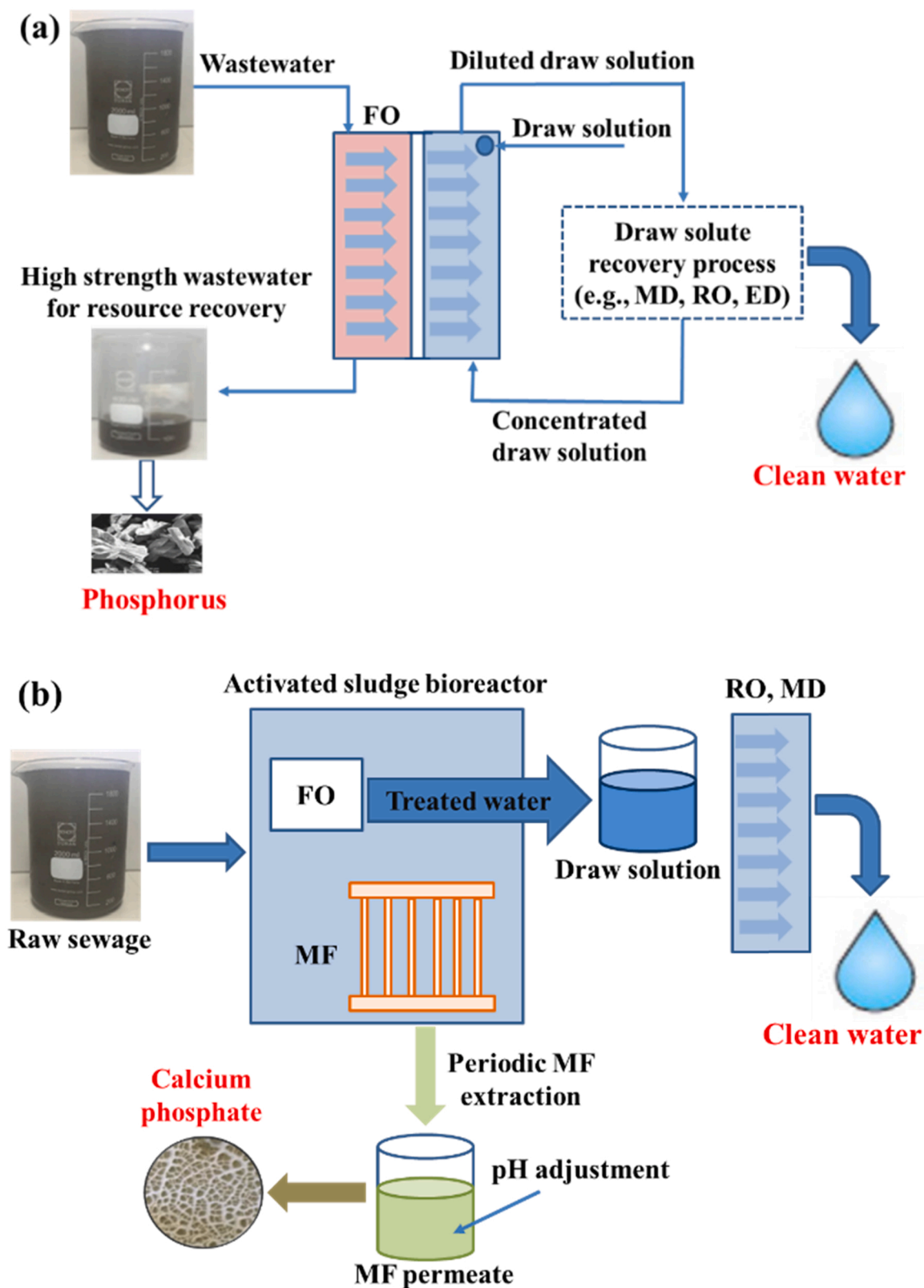
Fig. 3. Phosphorus and clean water recovery using (a) integrated FO with MD, RO or ED and (b) hybrid MF-OMBR with MD or RO. (a) (Modified from Xie et al. [206]).

concentrations of salinity followed by precipitation of salts in amorphous calcium phosphate form by adjusting pH between 8 and 9.5 [205]. Qiu et al. additionally studied an OMBR using seawater brine draw and achieved 90% phosphorus recovery in the form of amorphous

calcium phosphate [149]. Huang et al. found that the OMBR could enrich phosphate ions by six times (phosphorus concentration in feed solution and concentrated was 156 mg/L and 890–990 mg/L, respectively) [204]. However, salinity build-up in the bioreactor is a key

challenge for applying the OMBR system due to high salt rejection by the FO membrane and transportation of reverse salt from the draw solution. The elevated salinity in bioreactor could reduce water flux due to reduce osmotic driving force and reduce pollutant removal efficiency due to changes in microbial community in the system [207,211,212], thus reducing biological performance and aggravating membrane fouling [210].

Several studies reported to use microfiltration (MF) or ultrafiltration (UF) based OMBR process to alleviate the salt accumulation in the bioreactor and facilitate recovery of phosphorus from enriched nutrient streams [149,207,210,212]. Wang et al. found a stable operation of MF-OMBR in terms of water flux and biological performance [210,213]. Qiu et al. noted that hybrid MF-OMBR systems could recover phosphorus in the form of calcium phosphate, where the FO and MF membranes were operated in parallel [149]. Studies also noted that the use of ionic organic draw solutes could control salinity build-up in OMBR because organic salts that diffuse into the bioreactor can be biodegraded by activated sludge [203,214]. Luo et al. suggested that organic salts presented considerably low leakage, which rendered their suitability for OMBR applications [207,212,215]. However, very limited studies have been evaluated ionic organic solutes in the OMBR applications.

The OMBR system can be used as a stand-alone process or integrated with other processes to produce clean water and recover the draw solute. For example, in a hybrid OMBR-RO system, inorganic salts were retained by FO membrane and phosphate in the bioreactor, while the MF membrane periodically bled them out for phosphorus recovery by pH adjustment, and RO could produce fresh water [207,212]. Lu and He [216] investigated a hybrid OMBR-ED system to alleviate the salinity build-up and noted that ED recovered salts were reused as a draw solution in the OMBR system.

Gas permeable membrane has been used to treat different wastewater streams and showed approximately 80–100% ammonia recovery [217–222]. Garcia-Gonzalez and Vanotti studied the effect of pH on the recovery of ammonium using gas permeable hydrophobic membrane (GPHM) from swine manure. They observed around 88–94% ammonium removal by adjusting the pH of manure to 9 [223]. GPHM has also been used for simultaneous recovery of both nitrogen and phosphorus in the same process as termed as "N-P harvest" technique. In this process, ammonia is extracted in ammonia separation tank using a submerged GPHM and then the effluent wastewater is further treated in phosphorus recovery tank to precipitate phosphorus [224]. Pradhan et al. reported more than 98% nitrogen and phosphorous recovery from human urine using "N-P harvest" technique [225]. Simultaneous recovery of nitrogen and phosphorous from 1 m^3 urine via this approach can make a profit of 1.5 euros. A pilot-scale continuous trial of the "N-P harvest" process (100 L/h) has also been performed in Finland as a part of the large project "A breakthrough in the circular economy and clean solutions" funded by the Finnish government [226]. The business model and market study of the "N-P harvest" technique as well as the end-product market study suggested a promising future of this technology in Finland [226–228].

4.7.2. Anaerobic membrane bioreactor (AnMBR) with adsorber bed

Synthetic anaerobic membrane bioreactor (AnMBR) integrated with adsorber bed has been demonstrated for recovering NH$_4$-N from centralised WWTPs [229,230]. The anaerobic bioreactor is equipped with a membrane module. The influent wastewater is treated in an anaerobic bioreactor biologically yielding methane-rich biogas [231]. Its partially treated wastewater is passed through a membrane module ensuring that ammonium free effluent/permeate is produced. After the biogas release, the wastewater stream is passed through a dialyser. The dialyser unit has a hollow fibre membrane constructed from cellulase with a surface area in the domain of 1 m^2. With the outer body, it looks like a shell-tube exchanger. The wastewater from the anaerobic bioreactor is fed into the shell side of the dialyser, while ammonium rich stripping water/-dialysate is passed through the cellulose membrane from the shell side to

the tube side. The dialysate is then fed to the zeolite bed column, where ammonium is adsorbed. Once the bed is saturated, it is regenerated by a sulphuric acid (H$_2$SO$_4$) solution. This regeneration process produces ammonium sulphate ((NH$_4$)$_2$SO$_4$), which is then separated from H$_2$SO$_4$ by cooling. The zeolite in the adsorber bed can be replaced by ion exchange resins or biochar [187,232].

Instead of transforming NH$_4$-N to innocuous nitrogen by nitrification and denitrification processes, the AnMBR-adsorber bed process recovers nitrogen while producing biogas containing primarily methane. Therefore, this process is capable to recover both nutrients and energy. In the conventional wastewater treatment facilities, nitrogen is not recovered, rather converted to nitrite followed by nitrate, where a series of biological processes are utilised. AnMBR-adsorber bed process is simple, when compared to conventional wastewater treatment facilities. In addition, this process recovers nutrients and offers circular water solutions. Needless to say, this process is focused for centralised facilities. For a smaller site such as a household or estate, this process may not be cost-effective.

It is our opinion that biochar can also be used as an adsorbent in the adsorbent bed. The advantages of using biochar include: (i) both nitrogen and phosphorus nutrients could be adsorbed on biochar bed (potassium may also be possible to recover; however, it requires further study to confirm [187]), (ii) the nutrients loaded biochar can be used as a fertiliser and therefore, no acid dosing is needed [233], (iii) the overall operation cost could be much lower if biosolids' biochar is used [187], and (iv) there is a possibility that the technology can be applied cost-effectively in a wide range of scale (household, estate and centralised systems; requires detailed investigation [233]).

Previously, biomass-based biochar was used for recovering nutrients from cow-urine [233,234]. Cow urine is very similar to human urine (detailed later in case studies). The nutrients loaded biochar showed nearly double production of pumpkin when compared to biochar only pumpkin production. Biosolids is an in-house product of wastewater treatment facilities. Moreover, many wastewater industries are still in search for the sustainable disposal/reuse of this material. The slow pyrolysis of biosolids can produce highly porous biochar [235]. This highly porous biochar is expected to adsorb nutrients efficiently from centralised systems. As wastewater industries produce biosolids on a continuous basis, there is no need to consider regenerating biosolids' biochar. The nutrients loaded biosolids' biochar can be a high-value fertiliser and utilised for growing crops.

There are some challenges on considering biochar for nutrients recovery. The main challenge is the presence of pharmaceutical ingredients in urine/wastewater stream. The selectivity of nutrients over pharmaceutical ingredients adsorption needs to be established for the AnMBR-biochar process. A high-selectivity of nutrients over pharmaceutical ingredients is possible as biosolids' biochar can be functionalised purposefully [236–239].

All the membrane-based processes have several supreme features for nutrient recovery from wastewater and can effectively retain nutrients. In addition to nutrient recovery, integrated membrane processes may also be used to recover clean water. In principle, membrane-based processes have the promising potential of mining nutrients along with water purification both in centralised and decentralised settings. Although most of these processes are investigated at lab-scale [92,126,127,132,134], a few have been advanced to pilot scale stage [156–160] for wastewater reclamation. The major limitation of the membrane-based process is the fouling of the membrane, which can reduce the flux and increases the overall operational cost. In addition, there is a clear need to evaluate not only the technical and economic feasibility of these recovering nutrients but also product purity, process efficiency and life cycle costs. It is thus critical to integrate fundamental science with engineering research to effectively recover nutrients before their large-scale implementation.

4.8. Thermal processes

Thermal treatment processes such as evaporation and freeze crystallisation of urine reduce the volume and concentrate the nutrient. However, urine needs to be stabilised prior to the thermal treatment, which can be done via acidification or alkalinisation processes as discussed in Section 3. The evaporation of stabilised urine recovers nutrient as concentrated liquid; however, further evaporation until dehydration produces a solid product. The concentration of nitrogen in the recovered product usually varies between 5% and 13% [240]. Simha et al. investigated nitrogen recovery from fresh human urine via alkaline dehydration process and reported more than 90% nitrogen recovery in the solid product at 60 °C and pH> 12 [241]. In another study, Simha et al. observed a lower (70%) nitrogen recovery from ion-exchanged urine at a dehydration temperature of 50 °C and pH> 10 [242]. However, the dehydration temperature for the maximum recovery of nitrogen (80%) from urine, alkalised by MgO, was recommended as <40 °C [243]. Vasiljev et al. demonstrated up to 99% recovery of nitrogen via dehydration of Mg-doped alkaline urine at 38 °C using an incubator [66]. Senecal and Vinnerås dehydrated urine in the presence of wood ash at 35 and 65 °C and observed that around 90% and 64% of nitrogen, respectively retained as urea in the ash [244]. Dutta and Vinnerås studied the effect of dehydration temperature (20, 35, 60 °C) on the retention of nitrogen as urea in the drying agents (i.e., ash and lime) at a pH> 10 [245]. The highest nitrogen retention (74%) was observed at 35 °C, whereas evaporation rate at 20 °C was very low. Apart from the alkaline urine, nitrogen retention from acidified urine has also been investigated via dehydration [246]. Antonini et al. investigated solar evaporation of urine to recover nitrogen and phosphorous and obtained about 0.36 kg of solid fertiliser from 50 L of undiluted urine after 26 days of sun exposure [63]. Therefore, it can be concluded from the above discussion that the nitrogen retention is dependent on dehydration temperature, pH and drying agent. Approximately 1–36% of nitrogen was lost during the dehydration process for a dehydration temperature up to 65 °C. Pilot-scale alkaline urine dehydration plant are being installed and field tested in Sweden, France and Finland [247,248].

During freeze crystallisation process, clean water from the aqueous urine solution forms ice crystal, leaving the remaining concentrated nutrient solution. Randall and Nathoo performed a thermodynamic comparison of freeze crystallisation with reverse osmosis for nutrient recovery from stored urine and developed a protocol for estimating economic benefits of freeze crystallisation process [249]. Gulyas et al. studied the crystallisation process between −6 and −16 °C for the concentration nutrients in urine, collected from no-mix toilets [250]. Lind et al. reduced urine volume by 75% and concentrated about 80% of the original nutrients in urine via freezing process at −14 °C [251]. Ganrot et al. studied the freezing-thawing approach for nutrient recovery from human urine and reduced the urine volume to 40% and concentrated 60% of the nutrients [128].

In the case of wastewater, evaporation and freeze crystallisation processes may not be feasible as they need to handle an extensively large volume of water and thereby require significantly high energy for water evaporation and crystallisation compared to that of urine. Freeze crystallisation is energy efficient compared to evaporation due to the higher latent heat of evaporation than the heat of crystallisation. The evaporation process could be attractive with natural solar evaporation and suitable for the nearby farming areas, where the produced solid fertiliser can be directly used for land application. However, these aspects require further comprehensive investigations.

5. Community acceptance

Reshaping human behaviour for a source-separated toilet and applying human excreta in agricultural purpose are sensitive matters. It raises many questions including: (i) how users would respond to source-separated/no-mix toilets, (ii) how farmers feel in using recovered urine

Table 4
Examples of human food crops, fodder and energy crops [256–258].

	Definition	Examples
Human food crops	World's major human food supplier crops	Wheat, rice, vegetables, fruits, nuts, lentils etc.
Fodder crops	Crops primarily cultivated for animal feed	Grass such as bermuda grass, brome, false-oat grass, fescue; Clover such as alsike clover, red clover, subterranean clover, Brassica such as rapeseed, rutabaga, turnip etc.
Energy crops	Low-cost and low-maintenance crops primarily grown for energy production	Solid biomass such as willow and poplar; Gas biomass such as millet, white sweet clover; Liquid biomass such as copra, castor, groundnut, jatropha etc.

for fertiliser purpose, (iii) what concerns farmers would have on using urine as a fertiliser, and (iv) what type of crops farmers would consider to grow using urine as a fertiliser. There have been social studies on this topic conducted by various researchers and organisations. Lienert and Larsen reported data from 38 pilot projects of 7 European countries with 2700 respondents [252]. The idea of source-separated toilets was, in general, well accepted by the respondents. The acceptance range was between 75% and 95% and included data from three countries – Germany, Switzerland and Sweden. The number of respondents who were not interested to replace the existing toilet was relatively less – approximately 20% in Sweden and 50% in Austria. Approximately 38% of respondents in Sweden and 58% of respondents in Switzerland were even keen to pay more to replace the conventional toilets. Lienert and Larsen found that the primary motivation for the users for considering source-separated toilet was an ecological concern [252]. This has been a great finding as it highlights that communities realise the importance of preserving the environment and how source-separated toilets can help there. The case study on the acceptance of urine diverting toilets in the Netherlands suggested that around 64% of 338 respondents were willing to adapt this new sanitation if they were the owner due to the environmental benefits [253]. More than 60% of total respondents (132) were willing to cost extra for the urine diverting toilets in Hawaii, indicating a very encouraging result [254]. The perception survey conducted in on-campus residents at the University of Florida, the USA also showed very promising results. More than 80% of 8800 respondents voted for urine diverting toilets and would like to pay extra for the new sanitation approach, considering the water conservation as the main benefit. However, the respondents had a concern about the use of urine-based fertiliser [255].

The reaction of farmers in accepting urine as a fertiliser was much lower than the acceptance of source-separated toilets by the general public, as evidenced in the study conducted by Lienert and Larsen [252]. The majority Swiss farmers considered it to be a good idea in using urine-based fertiliser to farming lands. In contrast, most farmers from Berlin, Germany, were unsure about applying urine in their farming lands. Swiss farmers were okay to pay for urine-based fertilisers while the majority of farmers from Berlin (Germany), would consider using urine as a fertiliser if it would be free. In this study, the authors have identified a diverse range of reasons for farmers to accept urine as a fertiliser. They were concerned about the quality of fertiliser, public acceptance of crops being produced by urine and rapidly changing government regulations. Along with ecological and hygiene concerns, any concerns were mentioned. While considering human and animal food crops, farmers preferred to produce animal food crops or fodders. The production of energy crops was of the least worry for the farmers. A few examples of human food crops, fodder and energy crops are given in Table 4.

Simha et al. performed a comprehensive on behaviour change dynamics and social acceptance of new sanitation and human urine-
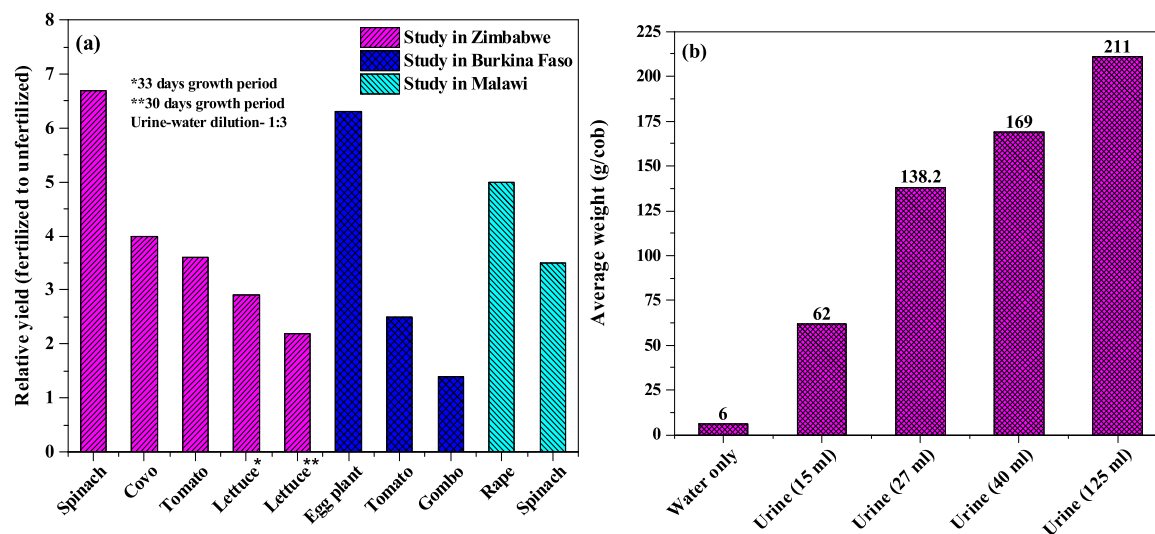
Fig. 4. (a) Effect of urine as a fertiliser on the yield of different vegetables and (b) effect of urine volume (per plant per week) on the production of maize cob. (a) (Prepared from Refs. [268,270]) and (b) (Prepared from Ref. [270]).

fertilised food [259]. They surveyed a total of 3763 respondents from 20 universities of 16 countries and identified about 68% of the respondents accepted human urine as fertiliser, 59% of the respondents were willing to eat urine-fertilised food, and 63% of the respondents were willing to pay the same amount as they usually pay for food. The assessment, conducted in South Africa, suggested that more than 80% of 225 respondents were not willing to eat human urine fertilised spinach and maize due to the believe that human urine can pose health risks; however, ~38% agreed to eat vegetables, fertilised by animal urine [260]. The attitudes and perceptions study on using human excreta fertiliser in Ghana also recommended similar findings that the majority of the respondents (~64%) were not willing to use sanitised human urine for cultivating their own crops or consume human excreta fertilised crops [261]. However, most of the respondents (~94%) would like to use animal excreta in their land as fertiliser. Out of 444 participants from Egypt, Iraq and Turkey, about 62% accepted diverting toilets, while 56% would like to use urine-based fertiliser [262]. Th USA consumers accepted human urine-based fertiliser the most for nonedible plants, whereas the least for crops, which can be used for human consumption [263]. The acceptance perception of human urine recycling as fertiliser was studied from both farmers and consumers viewpoint in India [264, 265]. About 59% of farmers (120) were positive to use urine-based fertiliser due to improved soil quality and potential cost savings, while 55% of consumers (1252) from university community considered human urine as fertiliser.

Therefore, the acceptance of urine-based fertiliser varied geographically and was dependent on a number of other factors. For instance, health risks are considered as one of the main factors for farmers not to use urine-based fertiliser in their land. Additionally, socio-cultural preferences and religious taboos related to urine recycling can also influence the certain community not to use urine-diverting toilets and consume urine fertilised crops [266,267].

6. Case studies on urine and wastewater utilisation

6.1. Case study 1: urine diverting toilets

A urine-diverting toilet is a specially designed toilet, which discharges urine and faces separately. Urine is often considered storing at 20 °C for 6 months to remove pathogens and recommended to use for almost all types of crops [268]. In Australia, Commonwealth Scientific and Industrial Research Organisation (CSIRO) and RMIT University with the financial assistance of Yarra Valley Water (a retail water utility in

Melbourne) carried out a study on the urine-diverting toilet [269]. It is known as the Kinglake West Sewerage Project. Urine collected from the household was applied to Turf Grass harvested at Green Acres farm. Urine was benchmarked with commercially available fertiliser, Seasol. The study suggests 4000 L of yellow water (urine) is needed to replace 5 L of Seasol. This might be because that urine was converted to ammonia and a significant portion was released to atmosphere. No data is available in the public domain on the nutrient concentration at the time of urine application to farming land.

In Burkina Faso (West Africa), the land application of urine on the yield of tomato, eggplant and gombo was studied for three years [268]. The production of urine applied tomato, eggplant and gombo were almost the same with mineral fertiliser applied vegetables; however, significantly increased from 1.9 to 4.7 tonne/hectare, 2.5–16.1 tonne/hectare and 1.5–2.2 tonne/hectare, respectively, when compared with crops produced with no fertiliser application.

The suitability of diluted urine as fertiliser was also investigated in Zimbabwe for spinach and maize growth, which were planted in containers [270]. Both spinach and maize were found to grow significantly in the presence of diluted urine when compared to no fertiliser application. The best dilution ratio of urine-water was reported between 1:3 and 1:5. Another study conducted in Malawi, showed 5 times higher production of rapeseed and 3.5 times higher production of spinach in containers with 3:1 urine/water as fertiliser, when compared with water only production as shown in Fig. 4(a) [270]. The agronomic trial in the same study exhibited that the average weight of urine fertilised maize cob was approximately 35 times more than that of water only case as illustrated in Fig. 4(b) and significantly increased with the increase in the quantity of urine dose. Under the "Chisungu Primary School Water and Sanitation Project" in Chisungu Primary School, Epworth, Zimbabwe, diluted urine (1:1 with water) was used in the maize field near the school [271]. The field study found approximately 30 times higher maize cobs production with diluted urine when compared to that of no fertiliser case. However, urine used in this trial was collected from urinals and stored in a tank before use. In Sweden, the large-scale application of urine as fertiliser was investigated in approximately 2 ha of barley field [272]. The urine was collected 2–3 times each year and stored in 3 tanks of 150 m^3 for 6 months prior to use in the field.

Both diluted and concentrated urine can be used as fertilisers; however, the dilution level depends on the corresponding nitrogen requirement. FO usually collects urine in concentrated form. So far, no study was found that used FO produced urine to the production of crops. However, studies conducted for urine-diverting toilets may provide
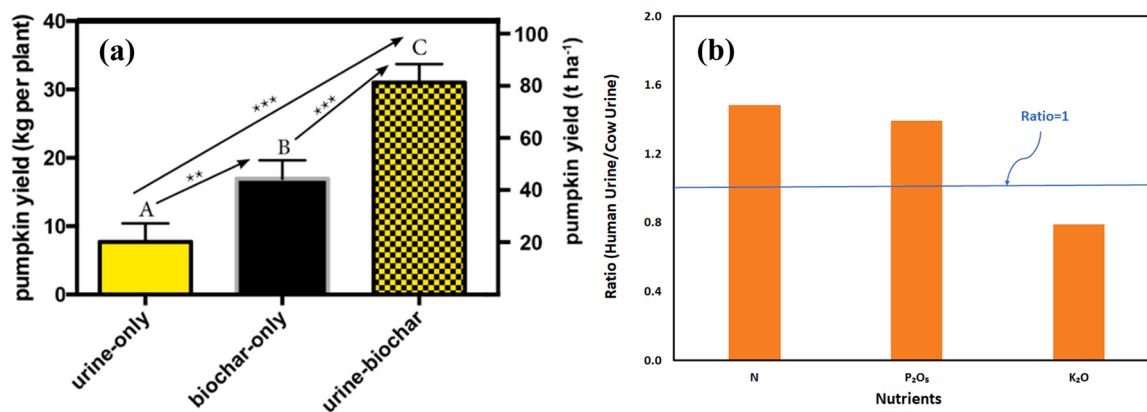
Fig. 5. (a) Effect of urine loading in biochar for pumpkin yield [233] and (b) comparison of nutrients between human urine and cow urine. (b) (Prepared from Ref. [277])

some insight on the application of FO produced concentrated urine.

6.2. Case study 2: nano-membrane toilet

Cranfield University developed the nano-membrane self-sustained technology, UK, under Bill & Melinda Gates Foundation [273]. The main focus of this technology was to produce energy and reusable water rather than fertiliser. The toilets are waterless and water from human excreta is evaporated in the process. The vapour is passed through membranes and condenses on nano-coated hydrophilic beads at the back of the toilet. The solids are handled by two approaches such as (i) collecting the dewatered sludge and processing in a centralised facility, and (ii) in situ combustions of dewatered sludge and produce energy to power the toilet. The capital cost of the nano-membrane toilet is expected to be 750 US$ with an operating cost of 0.05 US$/user/day [274]. The capacity of the toilet is about 10 persons with a lifetime of 7 years. The technology would be able to recover approximately 1.5 L water/user/day with an electricity generation of 90 Wh/user/day, indicating the revenue potential of 0.03 US$/user/day. The first case study of this toilet was carried out in Kumasi of Ghana. The toilet was designed to handle 20 L of waste mixture per day. The recovered sludge was used to produce electricity via gasification process.

6.3. Case study 3: struvite precipitation

The first large-scale struvite production facility was developed by Stuttgart University team in the "Abwasserzweckverband Raum Offenburg" municipal WWTP site (Germany) and the plant started struvite production at the end of the year 2011 [275]. The capacity of the plant was 12 m^3 digested sludge per charge with 90% recovery of phosphorus as struvite in aqueous solution. Prior to the large-scale production, the team developed "Stuttgart process" in the year 2003–2004 for the recovery of phosphorus as struvite from digested sludge. Semi-pilot scale (1 m^3 reactor) operations were carried out for the "Stuttgart process". The phosphorus from digested sludge was dissolved in acidic solution and then precipitated dissolved phosphorus struvite by adjusting pH over 7. The precipitated struvite was used as fertiliser in Georg-August University for canola uptake study. The struvite demonstrated better plant uptake when compared with that of commercial fertilisers such as triple super phosphate due to the lower solubility of struvite leading to long-term phosphorus availability. Another research group also investigated the quality of struvite as a fertiliser from Sweden [276]. The study found that the growth of wheat with struvite as a fertiliser was 1.5–2 times to that of optimum fertiliser.

6.4. Case study 4: urine loaded biochar

Biochar enriched with cow urine was directly applied to land as a fertiliser and the yield of pumpkin was compared with that of urine only and biochar only [233]. The biochar was produced at a low cost from *Eupatorium adenophorum* (sticky snakeroot or crofton weed or Mexican devil) biomass. The field study was carried out in a village, named Nalang in Dhading district of Nepal. The urine loaded biochar retained high level of nutrients leading to increase pumpkin production. Urine loaded biochar showed the best performance: the production of pumpkin with this biochar was 4 times of urine only and twice of biochar only applications as shown in Fig. 5(a). Human urine contains more nitrogen and phosphorus but slightly less potassium compared to cow urine (Fig. 5(b)) [277]. The land application of human urine and cow urine as fertiliser showed that human urine resulted in a higher yield of French bean and maize grain than cow urine [277]. Therefore, biochar loaded with human urine is expected to perform better as a fertiliser than that of biochar loaded with cow urine.

6.5. Case study 5: wastewater application in soil

The treated and untreated wastewater have been applied for agriculture in many countries, including Spain, Ghana, Mexico, Germany, India, Pakistan and Italy. In Pakistan, nearly 26% of vegetable is cultivated using wastewater [278]. In Ghana, about 11,500 ha of agricultural land is irrigated with diluted wastewater [279]. India uses approximately 2600 Mm^3 wastewater for irrigation of crops [280]. Kalavrouziotis et al. used treated municipal wastewater for cultivation of broccoli and brussels sprouts in Agrinio, Greece [281]. The use of treated wastewater increased the concentration of some heavy metals such as Ni (3.91–4.15 µg/g) and Pb (9.82–10.40 µg/g) in edible parts of broccoli and Cd (0.8–1.17 µg/g), Co (2.35–2.70 µg/g) and Ni 5.70–6.17 µg/g in edible parts of brussels sprouts. This high concentration of heavy metals in edible parts of plant can pose potential health risks. In another case study, Kalavrouziotis et al. also used treated municipal wastewater to investigate the heavy metal uptake by onion and lettuce and to compare that for fresh irrigation water [281]. The study reported no statistically significant difference in heavy metal concentration in plants, cultivated by treated wastewater and fresh water. Segura et al. observed the positive effect of wastewater utilisation for melon cultivation, as the use of wastewater reduced the N and K fertiliser consumption by 40.8% and 17.8%, respectively [282]. The heavy metal concentration in leaf was within the phytotoxicity levels. Rattan et al. conducted a case study on the utilisation of sewage effluents for irrigating agricultural lands under Keshopur Effluent Irrigation Scheme (KEIS) of Delhi, India [280]. They noticed 38–79% increase in organic carbon content in the case of sewage-irrigated soils compared to that of tubewell water-irrigated soils.

Only the concentration of Zn was higher than the phytotoxicity level.

7. Techno-economic analysis

The techno-economic analysis is considered to be one of the important tools for assessing the commercial-scale feasibility of technologies for nutrient recovery for WWTP. The following discussion focuses on the findings of the techno-economic analysis of a few nutrient recovery technologies at pilot-scale investigation.

Lin et al. [283] developed a comprehensive economic and environmental model for assessing the feasibility of ion exchange technologies (Section 4.5) to recover nitrogen in a WWTP site with a capacity of 1000 m^3/day and compared with the nitrification/denitrification route. The life span of the technologies was considered 20 years, and the plant was assumed to be run 360 days per year. The economic analysis found a lower nitrogen recovery cost (14.6 US\$/kg) for ion exchange compared to nitrification/denitrification (16.6 US\$/kg). The study also suggested integration of anaerobic digestion with ion exchange as the most economically compatible route than the nitrification/denitrification due to the higher synergies and lower energy as well as operating cost of ion exchange process. The techno-economic analysis of the ED process (Section 4.7.1) conducted at WWTP in Madison, Wisconsin, USA, with a capacity of 189270 m^3/day suggested that nitrogen could be recovered at the cost of 0.392 US\$/kg [284].

Sagberg and Berg [285] performed a cost optimisation study for an ammonia stripping (Section 4.1) system of 274 tonnes per year nitrogen production capacity in a WWTP site located in Oslo, Norway. The analysis suggested a nitrogen recovery cost of 0.78 €/kg, considering 70% nitrogen recovery and capital cost by 10% of the investment cost. The recovery of nitrogen using air stripping depends on the pH value. Several researchers claimed that the pH value between 11 and 12 is economically optimum for the air stripping process [85,86]. The percentage of nitrogen recovery increased from 27% to 92% when the value of pH increased from 8.5 to 11. Morales et al. [129] investigated a pilot-scale nitrogen and phosphorus recovery plant with a capacity of 600 L where struvite precipitation and ammonia stripping process was integrated with CO_2 stripper. In that integrated process, struvite was precipitated by dosing magnesium in urine and then the treated urine was mixed with anaerobic digester liquid from WWTP for further treatment in CO_2 stripper followed by ammonia stripping. The economic analysis of this integrated process concluded that the extra cost associated with the process was compensated by revenue from the increased nitrogen recovery and the estimated nitrogen recovery cost was approximately 3–4.5 €/kg.

Garcia-Belinchón et al. [286] conducted an in-depth economic analysis of struvite precipitation (Section 4.3) technology for phosphorus recovery integrated with a full-scale anaerobic digester in a Spanish WWTP site. They demonstrated that the higher value of phosphorus in feed concentrates and removal of P-biological instead of P-physicochemical can bring more economic benefits. For a case of 750 m^3/day concentrate consisting of 150 mg/L phosphorus, the analysis showed that the operating cost could be recovered completely by selling the recovered struvite, and the capital cost can be recovered in 7 years with a savings of 279 k€/year. In contrast, the economic analysis of Oxley Creek WWTP with a capacity of 55,000 m^3/day located in Brisbane, Australia estimated that the production of struvite from a 27,000 L reactor could generate a profit ranging from 13,000 and 149,000 AU\$ per year [136].

It can be concluded from the above discussion that the techno-economic feasibility of the nutrient recovery technologies is highly influenced by the nutrient recovery performance, purity of the fertiliser as well as synergies of the process. Therefore, the techno-economic assessment is still associated with uncertainty due to the difficulty in properly estimating revenues from the nutrient, such as the actual market values of nutrient as fertiliser. Therefore, the techno-economic assessment needs to be revisited for any technology when matured

Table 5
Challenges and opportunities of centralised and source-separated technologies.

		Centralised	Source-separated
Challenges	Concentration of nutrients	Low	High
	Focus	Treatment	Nutrient recovery
	Infrastructure modification	Difficult and expensive	Relatively simple and cost effective
	Odour	Minimal	Can be significant
	Pharmaceutically active ingredients	Treated in the process	May present in the end product
	Pathogen	Treated in the process	Requires storing and stabilisation
	Salt deposition	Managed in the process	Requires management particularly for waterless urinals
	Eutrophication	May result from any disturbance in treatment facility	May result from an odd event such as raining
Opportunities	Scopes for technology development	A shift from treatment to nutrient recovery	Development of affordable, cost-effective and hygienically safe technology
	Serving growing population	Possible to some extent with existing infrastructure upon nutrient recovery	Offers sustainable solution
	Circular economy	Partially possible upon nutrient recovery	Promising
	Fertiliser	Can make some contribution if nutrients are recovered	Can make significant contribution on world's fertiliser demand

information is available.

8. Benchmarking and qualitative comparison among the reviewed technologies

Centralised and source-separated technologies have been benchmarked for exploring their potential applications and performances. Current challenges and opportunities are highlighted in Table 5, while a detailed qualitative assessment is presented later in this section. Both centralised and source-separated systems have several challenges in terms of nutrients recovery, the cost for infrastructure modification and effective managements. The possible presence of pharmaceutically active ingredients in the end product [287–289], the possible presence of pathogen [290], salt deposition [291,292] and odour management [65,136,293] are primary challenges for source-separated systems. In contrast, low concentrations of nutrients is a major challenge for recovering nutrients from centralised systems. For the growing population, upgrading existing facilities can be a costly affair. This cost can be minimised if nutrients are somehow recovered in centralised facilities.

While there are several challenges, both centralised and source-separated systems offer several opportunities. Integrating a nutrient recovery technology with an existing centralised facility can help in partial recovery of nutrients leading to handling a larger volume of wastewater within the existing facility. The recovered nutrients may be utilised as a fertiliser and contribute partially to the circular economy. Source-separated technologies, in contrast, offer sustainable solutions for recovering nutrients from human urine and promise a circular economy in human waste space. Literature suggests that the human excreta can cover up to 70% of human fertiliser demand [294,295]. Source-separated technology can recover the majority of the nutrients and therefore fulfil a significant level of fertiliser demand. However, technologies are still in development in this space and affordable,

Table 6
Qualitative comparisons of the reviewed technologies [59,92,297,301–303].

		Ammonia Stripping	Nitrification/distillation	Struvite Precipitation	Ion Exchange	Zeolite integrated AnMBR	Biochar integrated AnMBR	Electrochemical	Forward Osmosis	Biosorption
Environmental	Greenhouse gas emission	CO_2, N_2O emission	CO_2, N_2O emission	CO_2 emission	CO_2 emission	CO_2 emission	CO_2 emission	CO_2 emission	Stabilised urine	Stabilised urine
Social	Public acceptance	Centralised, N fertiliser	Centralised, N fertiliser ($NaNO_3$)	Centralised, N, P fertiliser	Centralised, N fertiliser	Centralised, N fertiliser, natural resource (zeolite)	Centralised, N fertiliser, carbon sequestration	Decentralised, N fertiliser	Decentralised, N, P fertiliser	Decentralised, own use/ sellable fertiliser, carbon sequestration
Economical (AU$)	Capital costs	$60/Capita/y	> $60/Capita/y	$15/Capita/y	> $60/Capita/y	$6/Capita/y	$5/Capita/y	$12/Capita	> $60/Capita/y	$10/Capita/y
	Operation and maintenance costs	$7/Capita/y	~$7/Capita/y	$1.5/Capita/y	~$7/Capita/y	$0.8/Capita/y	$0.6/Capita/y	$30/Capita/y	~$7/Capita/y	$3/Capita/y
	Revenue generation	$57 K	~$9 K	$153 K	~$9 K	$9 K	> $9 K	$57 K	$6 K	> $57 K
Technological	Technology maturity	Full scale	Pilot plant	Full scale	Full scale	Pilot plant	Concept developed	Pilot plant	Pilot plant	Lab scale
	Nutrient removal Efficiency	N = 90%	N = 90%	N = 60%; P = 82%	N = 60%; P = 82%	N = 60%; P = 82%	N = 60%; P = 82%	N = 20%	N = 100%; P = 100%	N = 90%; P = 90%
	Energy consumption	>Moderate	High	>Moderate	Moderate	Moderate	Moderate	Electrical energy	Low	Low
	Resource depletion	Effluent treatment	Not applicable	Reactant	Resin use, effluent	Zeolite use, effluent	Waste recycle	Effluent treatment	Not applicable	Waste recycle

cost-effective and hygienically safe source-separated technologies are in demand.

The sustainability performance of wastewater treatment technologies is an imperative guide to the wastewater industry for taking a decision on appropriate technology for WWTP. A qualitative comparison on the sustainability of the reviewed wastewater treatment technologies is presented in Table 6, considering four sustainability indicators, such as environmental, technological, economic, and social aspects. In the comparative study, a number of sustainability dimensions were considered corresponding to each of the indicators, for instance, technological criteria had 4 dimensions, economical criteria had 3 dimensions and social and environmental criteria had one dimension each.

It can be seen from Table 6 that the forward osmosis (Section 4.7.1) and biosorption process (Section 4.7.2) produce stabilised urine. In contrast, the other technologies are associated with CO_2 emissions due to the hydrolysis of urine. Apart from this, the biosorption processes are considered environmentally sustainable due to environmentally friendly biosorbents instead of chemically synthesised compounds [296]. Therefore, the forward osmosis and biosorption could be more sustainable with respect to the environmental aspect compared to other technologies. The ammonia stripping (Section 4.1), struvite precipitation (Section 4.3), zeolite integrated AnMBR (Section 4.7.2) and biochar integrated AnMBR (Section 4.7.2) are mainly suited for centralised facility. On the contrary, electrochemical, forward osmosis, ion exchange, nitrification/distillation and biosorption may be suitable for decentralised application. Under economic sustainability, capital cost, operation and maintenance cost and revenue generation were considered as sustainability dimensions for the comparative analysis. Among the technologies reviewed in this study, ammonia stripping, forward osmosis, ion exchange and nitrification/distillation technologies are usually associated with higher capital investment costs [297]. The capital cost and operating cost of biochar integrated AnMBR are expected to be lower compared to the other technologies. The low operating cost of biochar integrated AnMBR would be due to the low-cost biomaterial (i. e., biochar from biomass or biosolids), compensation from biosolids management cost and no separation cost of ammonia, when biochar is used as a fertiliser. The capital and operating cost of zeolite integrated AnMBR is slightly higher than biochar integrated AnMBR process. The electrochemical process requires expensive electrodes and high energy, leading to the high operating cost of the technology [298]. However, struvite precipitation could produce significantly higher revenue, possibly due to the high market value of struvite fertiliser (330–550 AU $/tonne) than the nitrogen-containing product produced from the other technologies [136,297].

Nitrogen recovery in electrochemical technology (Section 4.4) is relatively low compared to other technologies due to side reactions [299]. Apart from electrochemical technology, the recoveries of nitrogen and phosphorus are nearly in the same range for all the reviewed technologies. The energy requirement of the technologies varies between low to medium, except nitrification/distillation technology, which requires high energy due to water evaporation [92,98]. So far, the majority of the technologies, except biochar integrated AnMBR and biosorption have been investigated at pilot-scale to full-scale plant [59, 92,300,301].

Briefly, the revenue generation from biochar integrated AnMBR is low but the process could be of renewed interest as centralised technology because of low capital and operating cost, high-quality fertiliser with macro and micronutrients and no waste (effluent) generation. In contrast, biosorption would be promising as a decentralised option in terms of environmental sustainability with moderate economic benefits.

9. Conclusions, current challenges, research gaps and future perspectives

Nutrients present in human urine, if recovered can make significant

contributions to continuous and sustainable supply of fertiliser, reduce our reliance on synthetic fertiliser and demonstrate a circular economy approach. This is highly important in the current scenario as food security in many countries is threatened by limited supply of cost-effective nutrients against the growing global population and increased food demand. Nutrients recovery from human urine cannot only reduce our reliance on synthetic fertiliser but significantly reduce the nutrient load to wastewater treatment plants, help reducing their operating cost and increase the life of their assets. There are several challenges in the pathway of recovering nutrients both at source and centralised levels. The critical research gaps and technological challenges are discussed below with relevant recommendations.

9.1. Nutrient recovery from the source separation of urine

9.1.1. Pathogen isolation and monitoring

Urine from a healthy human should not have any pathogen. However, in the urine separating toilets, pathogens may originate from faeces. The general approach is to store urine for a few weeks to a few months to ensure that the urine is pathogen-free. The downside of this approach is the degradation of urine to ammonia gas. There is a tremendous scope of research in this area where the aim could be to minimise/avoid urine storage by designing improved urinals and/or downstream processes while maintaining a high level of nutrients in the liquid [32,34]. For example, acetic acid addition to urine can keep pH very low, which helps kill pathogens and stabilise nitrogen in urine (i.e., avoiding its release as ammonia) [64]. Additionally, alkalinisation of urine using $Ca(OH)_2$, CaO or $Mg(OH)_2$ increases the pH of urine, leading to the prevention of enzymatic hydrolysis of urea [50,66].

9.1.2. Urine separation is a challenge for existing and new dwellings

Urine separation is a great challenge for existing dwellings. Traditionally, centralised wastewater treatment facilities collect as well as treat nutrients and wastewater together. Therefore, the separation of urine from existing dwellings can be more complex and add significant cost. There is a need to develop detailed assessments on the feasibility of urine separation from existing dwellings, how much these costs can be, who should pay for it, cost-benefit analyses, the length of transition as well as short and long-term goals for the transition of centralised treatment to urine separation and nutrient recovery at the source level. Such assessments would require multi-disciplinary research involving architects, scientists, engineers, government, regulatory bodies, water utilities and the community.

Waterless/minimum flush urinals are possible to be installed in new houses, apartments and commercial premises. However, the cost for this installation will be high compared to the existing centralised sewage management system. In addition, despite of significant water saving, the waterless urinals are associated with blockage issues in the piping system due to solids building up and scaling. However, nano-membrane toilet, developed by Parker [273] and novel fertiliser producing urinal, designed by Flanagan and Randall [44] can solve the issues of conventional waterless urinals. Recently, Gundlach et al. optimised the design of no-mix toilet integrating the *tea pot effect*, using computational fluid dynamics, which will efficiently guide the urine along a curved pipe network into the collector [304]. Assessments are necessary to explore the feasibility and cost-benefit analysis.

The scale-up decentralised urine treatment facility can be built in estate level. The source-separated urine (i.e., urine form no-mix/urine diverted toilets) from each household can be supplied to the facility for nutrient recovery. Although decentralised treatment of urine for nutrients recovery has a number of advantages, this approach poses barriers related to logistics of urine from household to the treatment location, economic viability and selection of appropriate technology. Therefore, comprehensive geographic information systems based geospatial analysis is required to understand the urine supply chain potential, human behaviour, features of infrastructure and optimum

location for urine treatment facility [305].

Without support from the government in subsidies and community acceptance, infrastructure for existing/new dwellings for source-level separation and nutrient recovery may not be possible. There is a need to develop a white paper in this area involving water utilities, government and community.

9.2. Nutrient recovery from the centralised facilities

Recovering nutrients from centralised facilities are challenging because the concentrations of nutrients significantly drop at centralised systems. An assessment is worth understanding if urine could be separately processed in the existing centralised systems. This may be done by implementing dual pipe arrangement; however, this will add significant costs to wastewater facilities. Therefore, a detailed cost-benefit analysis needs to be performed. Another approach is to explore nutrient recovery from low concentrations. Steps for this approach may include identifying possible materials to scavenge nutrients from wastewater, establishing adsorption coefficients, developing a process scheme and performing a detailed techno-commercial assessment.

9.3. Research gaps in technologies

9.3.1. Advancement in developed/developing technologies

Several technologies are in developing to an advanced stage in recovering urine-based nutrients. However, each of them has advantages as well as shortcomings. Therefore, advancements are required for each of the technologies. In the case of ammonium stripping technologies, steam stripping obviates off-gas treatment; however, steam stripping technology is complex and cost-intensive. The simpler design of steam strippers could be a possible advancement for this technology. Nitrification/distillation process is highly energy-intensive and the product, ammonium nitrate is explosive in nature. Energy optimisation as well as finding possible utilisation of this product would be a way forward for this technology.

Co-precipitation of calcium, ferric, aluminium, and organic compounds are barriers for the struvite production from human excreta. Selective precipitation of struvite needs to be studied to obtain high purity struvite. The high energy requirement is the primary bottleneck for electrochemical technologies. Integration of renewable energies could be a possibility to make this technology techno-commercially viable and this could be an area to explore. In the case of ion-exchange and adsorption technologies, more studies are required to expand the possible range of resins/adsorbents, understanding their mechanism and identify the best suit for nutrient recovery from wastewater. One of the major problems for membrane-based technologies is membrane fouling which adds significant costs. Therefore, research is required to optimise membrane fouling. In addition, techno-commercial assessment of all of these technologies would also be helpful considering scenario modelling.

9.3.2. Versatility of technologies

The approach to recover and utilise urine-based nutrients may vary for existing and new dwellings. It may be practical to perform this process in the household, estate or existing centralised system. The versatility of a particular technology to apply from small household to large centralised system may be important for it to be widely accepted. None of the developing or developed technology was proven for their wider applications. This is an important aspect and worth investigation.

9.4. Opportunities for developing of new processes/technologies

While several technologies are in advanced stages, the development of novel yet simple and cost-effective new processes/technologies are on demand for recovering nutrients from human excreta. From the qualitative analysis, AnMBR integrated with biochar or zeolite (for
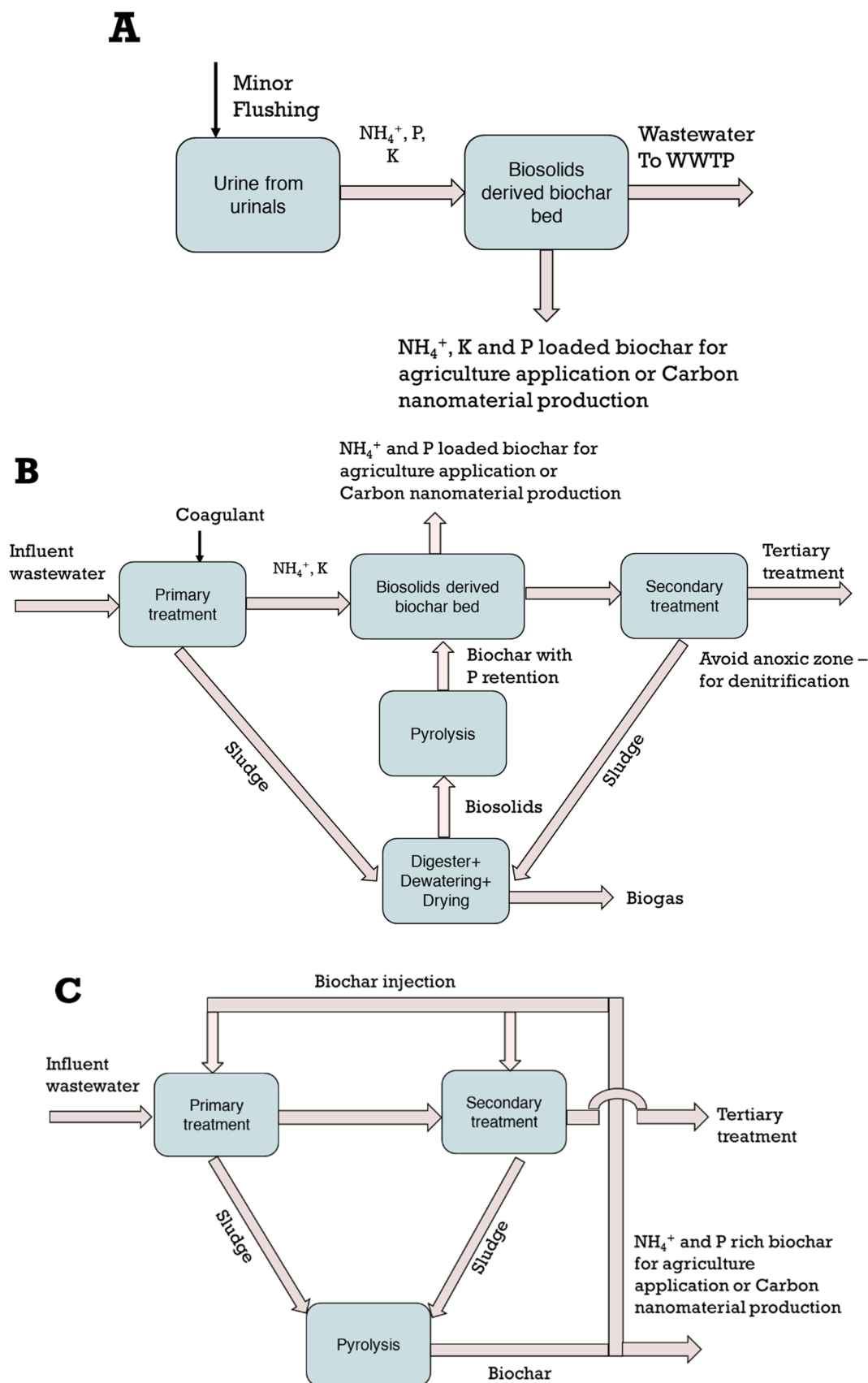
A

Minor Flushing → Urine from urinals → NH_4^+, P, K → Biosolids derived biochar bed → Wastewater To WWTP

↓ NH_4^+, K and P loaded biochar for agriculture application or Carbon nanomaterial production

B

NH_4^+ and P loaded biochar for agriculture application or Carbon nanomaterial production

Coagulant

Influent wastewater → Primary treatment → NH_4^+, K → Biosolids derived biochar bed → Secondary treatment → Tertiary treatment

Biochar with P retention

Pyrolysis

Biochar with P retention ← Pyrolysis ← Biosolids ← Digester+ Dewatering+ Drying → Biogas

Avoid anoxic zone – for denitrification

Sludge / Sludge

C

Biochar injection

Influent wastewater → Primary treatment → Secondary treatment → Tertiary treatment

Sludge / Sludge → Pyrolysis → Biochar

NH_4^+ and P rich biochar for agriculture application or Carbon nanomaterial production

Fig. 6. Application of biosolids derived biochar in the biosorption of nutrients: (Scheme A) Production of nutrient loaded biochar from source separated systems; (Scheme B) Production of nutrient loaded biochar from centralised systems; (Scheme C) Production of nutrient rich biochar from centralised systems.

centralised application) and biosorption (for centralised and source application) were found to be highly attractive. In particular, biosorption using biochar could be more attractive as it may not require regeneration of adsorbent bed (biochar). The nutrients loaded biochar could be used for agricultural production. Even further to this, the use of biosolids derived biochar in biosorption type processes can be more advantageous as they truly demonstrate a circular economy approach and allows water utilities to convert their biosolids into high value nutrient loaded biochar along with the reduction in the cost of nutrient removal as well as increase in the life of their assets.

Three biosorption schemes using biosolids derived biochar are suggested here (Fig. 6). At the source level, biosolids derived biochar can be used to trap nutrients of urine (Scheme A). The biosolids derived biochar bed may be used for a predetermined period of time. Once that period of time is over, the bed is expected to be saturated and can be taken out. The nutrient loaded biochar will then be ready for agricultural application. The biochar can be also used for chemical synthesis such as for the production of carbon nanomaterials. The biochar bed may be used at the centralised level after the primary treatment (Scheme B). As presented in the scheme, if a pyrolysis unit is installed in the wastewater facility, the treatment plant can produce biochar on its own and supply for capturing nutrients. Instead of using a bed, biosolids derived biochar may be injected in both primary and secondary treatment units as presented in Scheme C. In this case, the final solid product will be nutrient-rich biochar rather than nutrient-loaded biochar.

The adsorption kinetics are expected to be different for source and centralised systems due to the variations in concentration. Also, selectivity for nutrient adsorption over heavy metals and pharmaceuticals needs to be investigated in detail and if required then biochar should be functionalised to improve selectivity for nutrients over heavy metals and pharmaceuticals.

9.5. Social studies on the application of urine and urine-based fertilisers

Previous studies on community acceptance of urine separation suggest that farmers may not accept urine directly as a fertiliser for human food production. However, there is a wide acceptance in the community that urine separation and utilisation is senseful. A significant work is needed to engage and educate the farmers and wider community through social research demonstrating the benefits of nutrient recovery from urine at source or centralised levels.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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