A critical review of water, energy, fertiliser and product recovery from municipal wastewater treatment plants – market potentials, technologies and bottlenecks

Abstract

In recent decades, academia has elaborated a wide range of technological solutions to recover water, energy, fertiliser and other products from centralised municipal wastewater treatment plants. Drivers for this work range from low resource recovery potential and cost-effectiveness to a high demand for energy and the large environmental footprint of current treatment-plant designs. This critical review aims to inform innovators and decision-makers in water management utilities about the vast technical possibilities and market supply potentials related to designing (or redesigning) a municipal wastewater treatment process from a resource recovery perspective, but also the bottlenecks. Information and data have been extracted from literature to provide a holistic overview of this growing research field. First, reviewed data is used to calculate the potential of 12 resources recoverable from municipal wastewater treatment plants to supply national resource consumption. Second, resource recovery technologies which have been investigated in academia are reviewed comprehensively and critically. The third section of the paper reviews nine non-technical bottlenecks mentioned in literature, which have to be overcome to successfully implement these technologies into wastewater treatment process designs.

Keywords: wastewater resource recovery technology, wastewater reclamation, sustainable energy, circular economy implementation, critical review

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Content

List of Abbreviations

AC	Activated carbon
AGS	Aerobic granular sludge
AnMBR	Anaerobic membrane bioreactor
AOP	Advanced oxidation process
BES	Bioelectrochemical systems
BOD	Biological oxygen demand
CAS	Conventional activated sludge process
CHP	Combined heat and power
COD	Chemical oxygen demand
CPR	Chemical phosphorous removal
EBPR	Enhanced biological phosphorous removal
EPS	Extracellular polymeric substances
GAC	Granular activated carbon
GHG	Greenhouse gas
H_2O_2	Hydrogen peroxide
HL-MBR	High-loaded membrane bioreactor
MBR	Membrane bioreactor
MEC	Microbial electrolysis cell
MF	Microfiltration
MFC	Microbial fuel cell
MW	Megawatt
N	Nitrogen
P	Phosphorous
PAC	Powder activated carbon
PCP	Polychlorinated biphenyl
PHA	Polyhydroxyalkanoates
PJ	Petajoule
RO	Reverse osmosis
RRR	Resource recovery route
SCP	Single-cell protein
SRT	Sludge retention time
STOWA	Dutch Foundation for Applied Water Research
TJ	Terajoule
TOC	Total organic carbon
TSS	Total suspended solids
UF	Ultrafiltration
VFA	Volatile fatty acids
WMU	Water management utility
WRRF	Water resource recovery facility
WWTP	Wastewater treatment plant

Introduction

Wastewater resource recovery technologies have been extensively elaborated by the scientific community in recent decades, but their large-scale implementation at municipal wastewater treatment plants (WWTPs) is still poor. This can be explained by various technical, but primarily non-technical, reasons. Wastewater management plays a significant role in sustainable urban development (UNEP 2010). Traditionally, the goal of wastewater treatment was to protect downstream users from health risks. In more recent decades, protecting nature by preventing nutrient pollution in surface waters has become an extra goal. Consequently, nitrogen (N) and phosphorous (P) removal technologies have been implemented into WWTPs (Verstraete et al. 2009). The most widely used wastewater treatment technology is the conventional activated sludge process (CAS), in which aerobic micro-organisms metabolise the organic fraction present in the wastewater under constant oxygen supply (Oh et al. 2010). Although the CAS process succeeds in reaching legal effluent quality standards, it is considered unsustainable due to its low resource recovery potential and cost-effectiveness on the one hand, and its high energy demand and large environmental footprint on the other (Verstraete and Vlaeminck 2011).

The urge for more sustainable development, including more circular use of resources, and the resource inefficiency of current wastewater treatment practices have driven a paradigm shift within the scientific community with regard to wastewater solutions. It now proposes a transition from pollutant removal towards resource recovery, with wastewater recognised as a resource rather than a waste stream (Guest et al. 2009; Ma et al. 2013; van Loosdrecht and Brdjanovic 2014). By establishing more circular resource flows, the water sector can contribute to national and European sustainable development goals. As large-scale centralised WWTPs also represent centralised collection points for a variety of resources – namely water, energy, nutrients and products – their redesign from treatment facilities into resource factories provides possibilities for the circular economy. Within academia, it seems clear that current wastewater treatment practices are based on outdated concepts established in the early 20th century. Evolving new practices seems inevitable if we are to cope with population growth and improving standards of living, which are pushing our use of natural resources towards limits beyond sustainability (Daigger 2009).

Although the rationale and necessity to perceive wastewater as a resource has been emphasised, most water management utilities (WMUs) in Europe still focus on wastewater collection and treatment rather than resource recovery. Despite frequent scientific output over a long period on technological solutions to establish a more circular economy-based water sector, the implementation of full-scale resource recovery technologies in the wastewater segment is still very limited (Stanchev et al. 2017). The implementation of resource-oriented processes can be difficult because changing the current wastewater handling system incurs costs, creates operational distractions and itself consumes resources (Daigger 2009). Due to increasing numbers of available resource recovery technologies, WWTP process design has moved from being a simple technical problem into a complex issue that requires an integrated approach in order to make effective decisions (Bozkurt et al. 2017). The question of which of the growing range of available technical options we should focus on remains open. Uncertainty about which techniques are most useful and how to combine them is standing in the way of creating so-called ‘wastewater-resource factories’ (Li et al. 2015).

In addition to technical uncertainties valid for many emerging resource recovery technologies, various non-technological bottlenecks could hinder the successful implementation of such technologies into wastewater treatment processes. In particular, the market potential of and competition against recovered resources introduce uncertainties (van der Hoek et al. 2016). The water sector has hitherto been poorly equipped to address factors outside its traditional engineering-centred scope. Institutional compartmentalisation within the sector impedes integrated water-resource management and so must be remedied in order to make progress in developing resource-oriented wastewater management strategies (Guest et al. 2009). Consequently, there is a need for WMUs to strategically plan the transition from wastewater treatment towards resource recovery. The transfer of scientific insights to decision-makers in WMUs is an important requirement for this planning process. Resource recovery technologies can only be implemented and potentials can only be exploited if decision-makers at WMUs have a clear understanding of available and emerging technologies.

Previous reviews looking at wastewater resource recovery provide very valuable insights into particular branches of this broad and complex research field. Outstanding examples include the reviews on biological recovery routes (Puyol et al. 2016), energy and product recovery from sewage sludge (Tyagi and Lo 2016), phosphorous recovery from domestic wastewater (Le Corre et al. 2009; Rittmann et al. 2011; Egle et al. 2016), platforms for energy and nutrient recovery from domestic wastewater (Batstone et al. 2015), bioelectrochemical recovery systems (Wang and Ren 2013; Kelly and He 2014) and nutrient recovery with microalgae-based treatment systems (Cai et al. 2013). Despite these valuable contributions, though, as yet there is no review available that provides a holistic overview of the field.

This paper seeks to fill that gap, providing a holistic overview of resource recovery from municipal WWTPs. First, literature has been reviewed for data to calculate the potential of 12 resources recoverable at municipal WWTPs to supply markets in the Netherlands and Flanders (Belgium). Next, resource recovery technologies that have been investigated in academia are reviewed comprehensively and critically. Finally, the third section of the paper identifies, categorises and analyses bottlenecks from reviewed literature, which have to be overcome to successfully implement these technologies into WWTPs. Covering the market supply potential, the vast technical possibilities and the bottlenecks, this paper informs innovators and decision-makers at WMUs holistically about wastewater resource recovery. Although the effective treatment of wastewater for safe and environmentally-friendly discharge will remain the primary objective in WWTP design in the future, it is time to improve these plants' sustainability performance by integrating innovative resource recovery technologies into treatment-process designs (Bdour et al. 2009).

Market supply potentials of recovered resources

The potential of resources recoverable from municipal wastewater to satisfy societal demand for them is shown in Table 1. It reveals what role municipal WWTPs could potentially play in a circular economy if resource recovery routes were to be implemented nationwide. The supply potential for each resource is calculated on the one hand from the quantities which could be recovered from municipal wastewater under ideal circumstances and using the right technologies, and on other from the demand for those resources in the country. The calculations are based on the situation in the Netherlands. Data to calculate the supply potential has been collected from scientific articles and from official institutional reports. For the calculation of the nutrient supply potential, data collected in Flanders (Belgium) was used. The reason for choosing this source (Coppens et al. 2016) is that provides a very thorough, complete and up-to-date quantitative analysis of N and P flows within Flanders. No comparable analysis for the Netherlands is yet available. We assume, however, that N and P flows in Flanders are comparable with those in the Netherlands and so the calculated supply potentials for Flanders are also applicable there.

Resource demand		**Potential resource recovery**		**Supply potential (%)**
Water demand	***Netherlands***	**Water recovery**	***Netherlands***	**Water**
Water abstraction[a]	9482 mil. m3/a	Effluents[a1]	1909 mil m^3/a	20
		Treated by MF-UF[a2]	1622 mil. m^3/a	17
		Treated by MF-UF/RO[a3]	1217 mil. m^3/a	13
Energy demand	***Netherlands***	**Energy recovery**	***Netherlands***	**Energy**
Natural gas[b]	1227 PJ/a	CH_4 from COD (anaerobic digestion)[b1]	9 PJ/a	1
Electricity[c]	379 PJ/a	Electricity CH_4 (CHP)[c1]	4 PJ/a	1
		Electricity sludge co-combustion[c2]	0.5 PJ/a	0.1
Derived heat[d]	88 PJ/a	Heat CH_4 (CHP)[d1]	4 PJ/a	4
		Heat (effluent)[d2]	40 PJ/a	46
N demand	***Flanders***	**N recovery**	***Flanders***	**N**
N applied to crops[e]	169 kt N/a	Influent N[e1]	24 kt N/a	14
		N in activated sludge[e2]	5 kt N/a	2.9
		Sludge N recoverable (biodrying concept)[e3]	3 kt N/a	2

Industrial N fixation[f]	574 kt N/a	Influent-N[f1]	24 kt N/a	4
		N in activated sludge[f2]	5 kt N/a	0.8
		Sludge N recoverable (biodrying concept)[f3]	3 kt N/a	1
P demand	***Flanders***	**P recovery**	***Flanders***	**P**
P applied to crops[g]	24 kt P/a	Influent P[g1]	3.3 kt P/a	14
		P recovery as struvite [g2]	1.2 kt P/a	5
		P in activated sludge[g3]	3.0 kt P/a	13
		Sludge P recoverable (wet chemical technology)[g4]	2.7 kt P/a	11
Imports of mined P[h]	44 kt P/a	Influent-P[h1]	3.3 kt P/a	8
		P recovery as struvite [h2]	1.2 kt P/a	3
		P in activated sludge[h3]	3.0 kt P/a	7
		Sludge P recoverable (wet chemical technology)[h4]	2.7 kt P/a	6
Cellulose demand	***Netherlands***	**Cellulose recovery**	***Netherlands***	**Cellulose**
Paper (production)[i]	2671 kt/a	Cellulose in influent[i1]	180 kt/a	7
Energy demand (see above)	***Netherlands***	**Cellulose to energy**	***Netherlands***	
		CH_4 from cellulose (anaerobic digestion)[j1]	1.9 PJ/a	0.2
		Electricity CH4 (CHP)[k1]	0.7 PJ/a	0.2
		Electricity (cellulose pellets combustion)[k2]	0.7 PJ/a	0.2
		Heat CH_4 (CHP)[l1]	88 PJ/a	1
		Heat (cellulose pellets combustion)[l2]	1.2 PJ/a	1
VFA demand	***Global***	**VFA recovery**	***Netherlands***	**VFA**
Acetate market size[m]	16000 kt/a	Acetate recovery[m1]	142 kt/a	1
Propionate market size[m]	380 kt/a	Propionate recovery[m2]	64 kt/a	17
Butyrate market size[m]	500 kt/a	Butyrate recovery[m3]	29 kt/a	6
PHA demand	***Europe***	**PHA recovery**	***Netherlands***	**PHA**
Bio-PHA production[n]	147 kt/a	PHA recovery[n1]	103 kt/a	70
Alginate demand	***Global***	**EPS recovery**	***Netherlands***	**EPS**
Alginate production[o]	30 kt/a	Potential EPS production[o1]	76 kt/a	252
Fodder demand	***Flanders***	**SCP recovery**	***Flanders***	**SCP**
Fodder N consumption[p]	149 kt/a	Influent-N[p1]	24 kt/a	16
		SCP from anaerobic sludge digestate[p2]	4.8 kt/a	3
CO_2 demand	***Netherlands***	**CO_2 recovery**	***Netherlands***	**CO_2**
Industrial CO_2 consumption[q]	1239 kt/a	CO_2 emissions from biogas at WWTPs [q1]	53 kt/a	4

Notes: [a]Water removed from any freshwater source in 2014, either permanently or temporarily; mine water and drainage water as well as water abstractions from precipitation are included (Eurostat 2018a). [a1]Influent into Dutch WWTPs per year = 1928 million m^3 (Roest et al. 2010); water content in wastewater = 99% (WWAP 2017). [a2]Water recovery efficiency: microfiltration-ultrafiltration unit = 85% (Verstraete and Vlaeminck 2011). [a3]Water recovery efficiency: microfiltration-ultrafiltration unit = 85%, reverse osmosis unit = 75% (Verstraete and Vlaeminck 2011).

[b]Natural gas gross consumption 2017 (Eurostat, 2018b). [b1]CH_4 recoverable from wastewater per year in the Netherlands by anaerobic COD digestion under ideal conditions: all COD enters anaerobic digester and is recovered at a rate of 80% (Frijns et al. 2013).

[c]Supply, transformation and consumption of electricity available for final consumption in 2016 (Eurostat 2018b). [c1]CHP electricity conversion efficiency = 38% (Verstraete and Vlaeminck 2011). [c2]Theoretical energy in sludge organic matter in NL = 4100 TJ/a; energy required to evaporate the water content of the sludge = 2900 TJ/a; actual potential energy of sludge incineration NL = 1200 TJ/a (Frijns et al. 2013); electrical efficiency of coal-fired power plant = 40% (Faaij 2006).

[d]Supply, transformation and consumption of heat energy available for final consumption and derived from gas, coal or biomass combustion in 2016 (Eurostat 2018c). [d1]CHP heat conversion efficiency = 40% (Verstraete and Vlaeminck 2011). [d2]Total recoverable heat energy from effluent by heat pumps in the Netherlands, assuming $\Delta T = 5^{o}$ C and operation time = 100% (Roest et al. 2010).

[e]Represents the total anthropogenic N fertiliser input in Flanders (organic waste, manure, processed manure, synthetic fertiliser) and excludes atmospheric N fixation from legumes (Coppens et al. 2016).

[f]N produced with Haber-Bosch process (Coppens et al. 2016). [e1, f1]Calculated based on Coppens et al. 2016, N fluxes into WWTPs assuming that influent N could be fully recovered. [e2, f2]Assumed fraction of influent N ending up in sludge = 20% (Siegrist et al. 2008; Matassa et al. 2015). [e3, f3]N removal efficiency from sludge applying the biodrying concept = 70% (Winkler et al. 2013).

[g]Represents the total anthropogenic P fertiliser input in Flanders (organic waste, manure, processed manure, synthetic fertiliser) (Coppens et al. 2016). [h](Coppens et al. 2016). [g1, h1]Calculated based on Coppens et al. 2016, P fluxes into WWTPs assuming that influent P could be fully recovered. [g2, h2]Influent P recovery rate as struvite = 35% (Cornel and Schaum 2009). [g3, h3]Influent P ending up in activated sludge = 90% (Cornel and Schaum 2009). [g4, h4]Influent P ending up in activated sludge = 90%; P recoverable from sludge with wet chemical technologies = 90% (Cornel and Schaum 2009).

[i]Comprises the sum of graphic papers, sanitary and household papers, packaging materials and other paper and paperboard; excludes manufactured paper products such as boxes, cartons, books and magazines (Eurostat 2018d). [i]Mussatto and van Loosdrecht (2016); assuming full influent-cellulose fraction is sieved out (Ruiken 2010).

[j1]Total COD into Dutch WWTPs per year = 946,000 t (Frijns et al. 2013); cellulose fraction in influent COD = 31% (Visser et al. 2016); biodegradability of cellulose in separated anaerobic digester = 100% (Ruiken et al. 2013); share of COD load anaerobically converted into biogas = 80% (McCarty et al. 2011); CH_4 content of biogas = 65% (Frijns et al. 2013).

[k1]CHP electricity conversion efficiency = 38% (Verstraete and Vlaeminck 2011). [k2]Total cellulose entering Dutch WWTPs per year = 180,000 t (Mussatto and van Loosdrecht 2016); heating value of pellets = 13.8 MJ/kg; combustion energy conversion efficiency to electricity = 29% (Visser et al. 2016).

[l1]CHP heat conversion efficiency = 40% (Verstraete and Vlaeminck 2011). [l2]Total cellulose entering Dutch WWTPs per year = 180,000 t (Mussatto and van Loosdrecht 2016); heating value of pellets = 13.8 MJ/kg; combustion energy conversion efficiency to heat = 50% (Visser et al. 2016).

[m]Global VFA market sizes (Baumann and Westermann 2016). [m1-m3]Total COD in Dutch influent = 946,000 t (Frijns et al. 2013); influent COD up-concentrated = 75% (bioflocculation HL-MBR); VFA yield per COD in optimised alkaline fermentation = 33%; acetate fraction in VFA fermentation broth = 60.5%; propionate fraction in VFA fermentation broth = 27.5%; butyrate fraction in VFA fermentation broth = 12.5% (Khiewwijit et al. 2015).

[n]European PHA market size (de Jong et al. 2012). [n1]For VFA fermentation parameters, see [m1-m3]; PHA yield per VFA-COD = 44% (Fernández-Dacosta et al. 2015); assumed PHA downstream process yield = 100%.

[o]Global conventional alginate production (Pawar and Edgar 2012). [o1]EPS recovery: total COD into Dutch WWTPs per year = 946,000 t (Frijns et al. 2013); sludge yield per COD = 40% (Wan et al. 2016); EPS content in granular sludge = 17.5% (van der Roest et al. 2015); assumed EPS downstream process yield = 100%.

[p] Total N in fodder consumed in Flanders (Coppens et al. 2016). [p1]Calculated based on Coppens et al. 2016, P fluxes into WWTPs assuming that influent N could be fully recovered. [p2]Assumed fraction of influent N ending up in sludge (sludge N) = 20% (Siegrist et al. 2008; Matassa et al. 2015); assumed fraction of sludge N that is solubilised in the liquor after anaerobic sludge digestion = 100%; assumed N conversion efficiency into protein = 100% (Matassa et al. 2015).

[q, q1]Hogendoorn et al. (2014).

Table 1. Calculated market supply potentials of water, energy, nutrients and products recoverable from municipal WWTPs in the Netherlands or Flanders.

Water supply potential

Water reuse from municipal WWTPs can significantly reduce a city's freshwater demand (Verstraete et al., 2009). A well-studied success story for water reclamation and reuse is the city of Windhoek (Namibia), where

25% of the city's potable water supply stems from wastewater (Verstraete and Vlaeminck 2011). Other urban examples include the city of Chennai (India), where the reuse of 40% of the generated wastewater satisfies 15% of the city's water demand (IWA 2018). At Xi'an University in China, a decentralised treatment system produces water for various non-potable uses, such as toilet flushing, gardening and waterfront landscaping, and has cut freshwater consumption on the campus by 50% (Wang et al. 2015b). In the water-scarce city of Monterey (California, USA), a large agricultural area is supplied with almost 80,000 m^3/day of nutrient-rich reclaimed municipal wastewater to irrigate and fertilise crops (McCarty et al. 2011). At the state level, Israel and Singapore are two examples of countries with nationwide wastewater reuse schemes. In Israel, almost a quarter of the country's water demand is met by reclaimed wastewater (Wang et al. 2015b), while Singapore achieves 40% with its NEWater reclamation plant (PUB 2016).

However, wastewater entering a municipal WWTP contains only water used domestically, fractions of industrial water and storm water. Water used in the agricultural sector, which is the second largest consumer of water in Western countries, after industry (Ranade and Bhandari 2014), does not reach these plants. Even if a large fraction of WWTP influent is reclaimed, then, it can only partly satisfy total regional demand for fresh water. As shown in the examples in Table 3, the effluents discharged by Dutch WWTPs equate to 20% of the total volume of fresh water abstracted in the Netherlands. Although the application of filtration technologies to these effluents implies water losses, advanced treatments could produce different water qualities suitable for various reuse purposes, depending on the process applied. Microfiltration and ultrafiltration could reduce Dutch freshwater abstraction by 17%, while reverse osmosis might decrease this number to 13%. Only the latter technology could reclaim water of high enough quality to enter the potable supply, so the others would only be useful if the reclaimed water was intended to be used in a non-potable context.

Energy supply potential

A municipal WWTP can be responsible for a significant share of the total energy consumption by its operating local authority (Schopf et al. 2018). On the other hand, the potential chemical energy held in typical municipal wastewater has been measured as being five times higher than that needed for CAS process operations (Wan et al. 2016). As shown in Table 3, 94 petajoules (PJ) per year is the theoretical maximum energy that could be recovered from Dutch WWTPs as CH_4, assuming that all the chemical oxygen demand (COD) in the influent were to enter an anaerobic digester to be converted into biogas at 80% efficiency. Currently, only about 25% of this maximum potential is exploited (Frijns et al. 2013).

Even under ideal conditions, however, CH_4 recovered from wastewater would substitute less than 1% of Dutch annual natural gas consumption. If the recovered CH_4 is converted into electricity and heat in a combined heat and power (CHP) unit of typical efficiency (c. 40%), less than 1% of the Dutch electricity consumption and only 4% of the derived heat currently used in the Netherlands could be supplied. Assuming that all excess sludge is dewatered and then co-combusted in coal-fired power plants, the amount of electricity obtained is only a negligible 0.1% of overall consumption. The main reason for the low energy-recovery potential of sludge incineration is that considerable energy is required to evaporate its water content, as it still often contains some 80% of that even after mechanical dewatering (Frijns et al. 2013).

The total thermal energy contained in WWTP effluent by far exceeds the on-site demand for heat, indicating that these plants have huge potential to feed district heating networks or provide heat for industrial purposes (Kretschmer et al. 2016). With a view to process optimisation, using this heat for sludge drying is also a promising possibility. The yearly average effluent temperature in Dutch WWTPs is 15 ºC. Assuming that a heat-exchange or heat-pump system is installed to recover heat energy of 5 ºC and that this operates 24 hours a day, 365 days a year, the total recoverable heat from municipal WWTP effluents in the Netherlands would be about 40 PJ (Roest et al. 2010). This equates with more 40% of the total heat energy derived from gas, coal or biomass combustion processes. Moreover, heat recovered from Dutch WWTP effluents has an energy recovery potential approximately ten times higher than that of heat derived from recovered CH_4 combustion in a CHP unit (see Table 3).

Nutrient supply potential

Close to 100% of the phosphorous (P) eaten in food is excreted by the human body. On a global scale, about 17% of all mined mineral P ends up in human excreta. Cities are P 'hotspots' and urine is the largest single source of the P emerging from them (Cordell et al. 2009). Table 3 shows that, in the Flanders region (Belgium), for example, the total P entering WWTPs is equal to 8% of Flemish industrial P ore imports and 14% of the total fertiliser orthophosphate P used in the region. Since P could be recovered from sludge incineration ash with efficiencies of about 90% (Cornel and Schaum 2009), this recovery pathway would lead to a realistic supply potential of 11% of Flemish fertiliser demand or 6% of Flemish industrial P ore imports. By contrast, if soluble P is recovered as struvite, the influent P recovery percentage lies between 10 and 50% depending on the treatment process applied (Cornel and Schaum 2009; Wilfert et al. 2015). The supply potential of the struvite recovery route is thus significantly lower (3%) than that of the sludge recovery route.

Thirty per cent of global N fertiliser demand could be met through wastewater N recovery practices. But in countries with intensive agriculture systems, like the Netherlands, this figure shrinks to just 18%, representing the fraction of fertiliser N that enters WWTPs (Mulder 2003). As shown in Table 3, much the same applies in Flanders, where 14% of total N fertiliser demand or 4% of that for industrially fixed N could theoretically be met from wastewater N recovery practices (assuming a 100% recovery rate of influent-N concentrations). But since only 20% of influent N is retained in the sludge after the CAS process, recovery rates using the technologies currently available are significantly lower (Siegrist et al. 2008; Matassa et al. 2015). The biodrying concept, for example, which converts sludge into an energetically favourable state and simultaneously recovers ammonium sulphate (Winkler et al. 2013), could satisfy only 2% of total Flemish demand for N fertiliser or less than 1% of that for industrially fixed N.

Product supply potential

As exemplified for the Dutch case, multiple products – amongst them cellulose, volatile fatty acids (VFAs), polyhydroxyalkanoates (PHAs), extracellular polymeric substances (EPS), single-cell protein (SCP) and CO_2 – can be recovered from wastewater. In principle, so too can other innovative products. Data on such routes is still limited, however, which raises uncertainties. The Dutch Foundation for Applied Water Research (STOWA), the joint scientific centre of the Dutch water boards, is currently developing wastewater resource recovery strategies focusing on five of the products named: cellulose, EPS, VFA, PHA and CO_2 (Efgf.nl 2019).

Cellulose fibres may represent 50% of the total suspended solids and a significant fraction of the inert solid fraction in municipal WWTP influents. In the Netherlands, more than 80% of consumed toilet paper ends up in WWTPs and could be recovered by taking a real cradle-to-cradle approach – although it does remain questionable whether customers would accept recycled toilet paper (Ruiken et al. 2013). As shown in Table 3, if the cellulose fibres are used as raw material for the Dutch paper and paper board industry, they have the potential to satisfy 7% of demand from this sector. In all, 180,000 t of toilet paper are flushed down Dutch toilets every year. As this represents approximately 180,000 trees (Mussatto and van Loosdrecht 2016), annual deforestation of 45 ha could be avoided by recycling toilet paper, assuming that the normal density of Dutch forests is 4000 trees/ha (Schelhaas 2008). Using sieved cellulose as feedstock for a separated anaerobic digestion unit, as tested by Ruiken et al. (2013), would only produce quantities of CH_4, electricity and heat equivalent to less than 1% of total societal demand. Not surprisingly, a similarly low energy-supply potential is expectable if the fibres are dried, pressed into energy pellets and combusted for electricity and heat generation as investigated by Visser et al. (2016).

VFAs produced in the Netherlands from up-concentrated COD combined with long sludge retention times could, depending on the VFA type, meet 1-17% of global market demand. But published data on the global production volumes of the three main VFAs differs considerably (Zhang and Yang 2009; Zacharof and Lovitt 2013; Baumann and Westermann 2016; Bhatia and Yang 2017), which makes this estimate uncertain. Country-specific market data about VFAs is not readily available for academic use, the only source being commercial market analysts selling reports for several thousand euros each (Baumann and Westermann 2016). If COD-derived VFAs are converted into PHA, a significant 70% of European PHA production could be supplied by the combined Dutch WWTPs.

If Dutch influents were invariably treated using aerobic granular sludge processes, and also assuming that EPS can be substituted for alginate due to their similar material properties, the potential supply of EPS recovered from Dutch municipal WWTPs would exceed global alginate production by a factor of around 2.5. If such a scenario were realised, it would certainly have a severe impact on the global alginate market, including prices.

Intensive livestock production relies on protein-rich fodder. If all Flemish influent N could be converted into protein fed to animals, 16% of the consumption of conventional fodder N stemming from protein-rich plants like soya beans could be avoided. The production of single-cell protein from wastewater as proposed by Matassa et

al. (2015) could be environmentally much more efficient than generating conventional fodder. Its potential to satisfy Flemish demand for fodder, however, is rather limited; it could substitute only 3% of conventional fodder N due to the fact that only the sludge-N fraction is converted; most of the influent N remains in the water line as ammonium or is denitrified.
Upgrading recovered biogas by extracting a rather pure CO_2 stream could contribute significantly towards achieving the greenhouse-gas emission-reduction target of the Dutch water boards. Less significantly, it could satisfy some industrial CO_2 consumption needs (4%) – although this should still be considered an important potential contribution, because the energy demand of CO_2 from biogas is around 80% lower than that from conventional processes (Hogendoorn et al. 2014).

Resource recovery technologies
By reusing resources contained in municipal wastewater, we could tackle water scarcity problems, lower fossil energy consumption and address global nutrient needs. In addition to water, energy and nutrient recovery, it should not be forgotten that a variety of products can be recovered from wastewater (van Loosdrecht and Brdjanovic 2014). This section critically discusses resource recovery routes (RRRs) for water, energy, nutrients and products. We define an RRR as the route taken by a resource entering a WWTP, extracted from the flow and then refined before finally being used. While resource extraction happens on site at the WWTP, refining and usage can be undertaken elsewhere.

Water reclamation and reuse technologies
Around 99 wt% of the matter contained in wastewater is water (WWAP 2017), so reclaiming and reusing this could be a more sustainable option than, for example, desalination or long-distance fresh-water transfers (European Commission 2018). Furthermore, the main driver for the reclamation and reuse of domestic wastewater is water scarcity caused by generally uneven global fresh-water distribution and climate change-related water stress (Wang et al. 2015b). Secondary wastewater treatment processes do not fully remove biological oxygen demand (BOD) and only eliminate 95% of total suspended solids (TSS) from effluents, which also contain residual concentrations of organic micropollutants like pharmaceuticals, polychlorinated biphenyls (PCPs) and pesticides. To meet the strict legal standards for microbe and micropollutant concentrations in reclaimed water, the effluent from secondary wastewater treatment processes needs to be further processed on advanced treatment lines (Eslamian 2016). Advanced treatment technologies can be divided into filtration, disinfection and advanced oxidation processes (Figure 1).

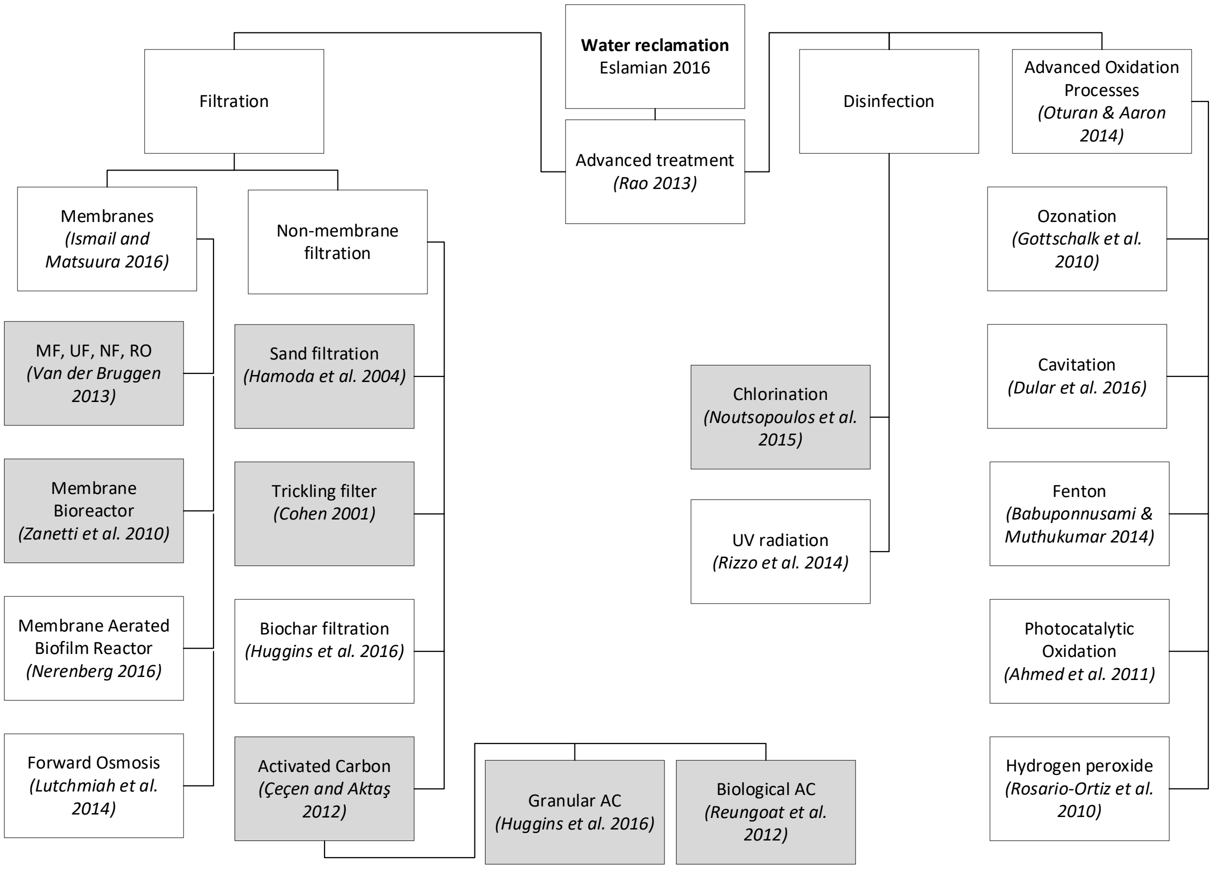

Figure 1. Examples of technologies to reclaim water from municipal WWTPs. Since a detailed presentation and discussion of each technology is beyond the scope of this paper, a scientific publication that explains or reviews it further is referenced. Grey shading indicates techniques that have been applied on a large scale at municipal WWTPs. Unshaded boxes show technologies that are not applied widely.

Membrane filtration

Membrane processes allow reliable advanced treatment and are considered a key technology for advanced wastewater reclamation and reuse strategies. Their advantages include the need for less space, action as a physical barrier against particle material and efficiency at retaining microorganisms without causing resistance or by-product formation. Membranes are included in several prominent large-scale advanced treatment designs used worldwide for artificial groundwater recharge, indirect potable reuse or industrial process-water production. Ultrafiltration membranes (UF) remove colloids, proteins, polysaccharides, most bacteria and even some viruses, and produce high-quality treated effluents (Rao 2013). Techniques using membranes with smaller pore sizes, namely nanofiltration (NF) and reverse osmosis (RO), are useful to separate ions and dissolved solids from water (Wintgens et al. 2005). A successful example of NF/RO membrane technology recovering water from wastewater for indirect potable reuse is found in Singapore, as part of the NEWater project. This consists of several treatment steps and generates significant amounts of reclaimed water to refill natural drinking-water reservoirs in the city state (Lee and Tan, 2016).

Membrane bioreactors (MBRs) might be especially useful for wastewater reuse applications because they include an initial membrane filtration step. A pilot application within the NEWater project, using MBR/RO/UV after primary sedimentation, successfully recovered water of potable quality (Lee and Tan, 2016). MBRs combine the activated sludge process with microporous membranes for solid-liquid separation and have been applied frequently, on a large scale, for municipal wastewater treatment (Zanetti et al. 2010). Possible advantageous features of MBRs are the separate control of sludge and hydraulic retention times and higher mixed liquor-suspended solid concentrations reducing the reactor size. On the other hand, MBRs can also have several disadvantages compared with the CAS process; for example, greater process complexity, less readily dewaterable sludge and greater sensitivity to shock loads. In addition, MBRs are associated with higher equipment and operational costs, due mainly to membrane cleaning and, at high loading rates, to higher aeration requirements (Judd et al. 2008).

Although membrane technologies can provide very high effluent qualities, useful for any type of water reuse, they are costly in operation. Membrane fouling in wastewater applications can be a significant problem, too, especially at high fluxes. Applying low fluxes reduces operational costs but raises capital costs, as more membrane units are necessary (Pearce 2008). To decrease potential fouling and clogging, effective operation requires extensive pretreatment of secondary effluents (Wintgens et al. 2005). Additional cost factors for efficient large-scale membrane-technology application for wastewater reuse are the complexity and disposal cost of the retentate (Banjoko and Sridhar 2016). Moreover, high pressure is generally needed for membrane filtration. The energy requirements for MF/RO systems are approximately 3 kWh per m^3 (Batstone et al. 2015) and may far exceed the recoverable chemical energy in the wastewater. Côté et al. (2005) calculated a total lifecycle cost of about US$0.3 per m^3 for water reclaimed by an UF/RO treatment. Verstraete et al. (2009) estimated an overall cost of approximately €0.8 per m^3 for the CAS process followed by UF/RO, including costs for retentate discharge and revenues from water valorisation. Reclaiming potable water for households and/or industries from wastewater was shown to be cost-ineffective for the Amsterdam region due to high process costs by comparison with conventional options (van der Hoek et al. 2016). Membrane-based filtration processes always require considerable electricity input (Batstone et al. 2015), although lower water viscosity in warm climates may decrease these energy requirements. In our resource-constrained world, however, increasing the consumption of one resource in order to make another available has to be considered very carefully (Daigger 2008).

Activated carbon filtration

Activated carbon (AC) filtration as advanced treatment process can produce higher effluent qualities, useful for water reuse. AC units can be made from various raw materials, including coal, peat, petroleum coke and nutshells. These carbonaceous substances are activated by physical and/or chemical agents under high temperatures, endowing them with effective filtering capacity for COD, total organic carbon (TOC), chlorine and a wide range of hydrophobic organic pollutants like pharmaceuticals (Stefanakis, 2016). Two major driving forces cause the adsorption of solubilised pollutants to the surface of AC filters: (i) the solubility of the dissolved pollutant; and (ii) the affinity of the contaminant for the adsorbent. AC is applied as a powder (PAC) with a grain diameter of less than 0.07mm or as granular activated carbon (GAC). PAC can be added directly to the activated sludge unit prior to advanced filtration steps, whereas GAC is used in a separate pressure or gravity-driven filtration unit. While PAC needs to be disposed of after use together with the sludge, GAC can be regenerated cost-effectively on site (Trussel 2012).

Various studies have shown the effectiveness of combining AC filtration with other advanced treatment steps for the removal of water pollutants. Ormad et al., (2008) showed that AC coupled with oxidation by ozone removes 90% of different types of pesticides during drinking-water production. AC in combination with ozonation improves the removal/degradation of various emerging pollutants, since AC can function as a catalyst in the ozonation reaction while ozone increases the pore size and active surface area of AC (Qu et al. 2007; Gerrity et al. 2011; Reungoat et al. 2012). Furthermore, if AC is applied upstream of membrane filtration units, the filtration performance of the membrane systems is significantly improved (Kim et al. 2007; Sagbo et al. 2008; Gai and Kim 2008). But, compared with other alternatives, the cost-effectiveness of AC as a membrane pretreatment step may be questionable. Possible shortcomings of AC filtration are that compounds of low molecular weight and high polarity, such as amines, nitrosamines, glycols and certain ethers, are not adsorbed (Çeçen 2012). In addition, contaminants are transported from the water to the filter but are not degraded, so subsequent filter disposal or cleaning has to be considered as an additional cost (Oller et al. 2011).

Advanced oxidation processes

The removal of emerging pollutants like pharmaceuticals is a growing concern in wastewater treatment (Ranade and Bhandari 2014) and certainly need to be considered in water-reclamation processes. Advanced oxidation processes (AOPs) form hydroxyl radicals (•OH) as highly reactive oxidant agents for the destruction of a wide range of non-biodegradable organic contaminants like pharmaceuticals, dyes or pesticides, as well as bacteria, protozoa and viruses. AOPs are often run by external energy sources such as electric power or light. They are usually applied as final polishing and disinfection step after biological treatment, but can also be used as a pretreatment step that breaks down organic contaminants to enhance subsequent biological treatment measures (Petrovic et al., 2011). AOP systems can be configured according to the contaminant composition, concentration and required effluent quality. Besides the sequential application of various AOPs to enhance selectivity of several classes of different pollutants, the combined application of single AOPs can significantly enhance the oxidation rate of organics (Comninellis et al. 2008). Various publications provide a thorough overview of the vast range of possible combinations of AOPs to treat recalcitrant pollutants in industrial or municipal wastewater (Oller et al., 2011; Oturan and Aaron, 2014; Petrovic et al., 2011; Wang and Xu, 2012). But the application of AOPs may also have shortcomings, like high costs for reagents such as ozone and hydrogen peroxide or for the required energy source, such as ultraviolet light (Agustina et al. 2005). The following paragraphs briefly describe ozone and ultraviolet irradiation, the most widely used AOP techniques. Unless membrane treatment in the form of RO is already applied, an additional disinfection unit may be needed for safe wastewater reuse.

Ozone (O_3) is a commonly used oxidising agent, often produced on site from dry air or pure oxygen. It is useful for the elimination of bacteria, viruses and protozoa and therefore a suitable process for water reuse. While higher pressure, pH value and contact time enhance pollutant degradation efficiency, a higher temperature would limit it. The main disadvantages of ozonation are its high energy demand and the short stability of ozone itself, which can make the process costly. For water containing certain levels of bromide, there is a potential risk of its conversion to bromate during ozonation, which can lead to the formation of carcinogenic bromated organic compounds. This is especially relevant in seawater desalination and drinking-water treatment, and to a lesser extent in wastewater effluent polishing. After ozonation, activated carbon filtration is often applied to reduce the content of biodegradable compounds in the flow (Stefanakis, 2016).

Ultraviolet (UV) irradiation is considered a fast, efficient, safe and cost-effective process, and is thus one of the most prominent alternatives to chemical disinfection methods (Brahmi et al., 2010). UV light wavelengths hold enough energy to let pollutant molecules release electrons and therefore become unstable. In addition to this direct photolytic action on compounds dissolved in the water, UV technology may degrade other contaminants through the photochemically-assisted production of oxidants like hydroxyl radicals and through photochemically-assisted catalytic processes (Masschelein and Rice 2002). Micro-organisms have evolved mechanisms to repair their partially denatured DNA after UV light exposure, however, which can lead to DNA reactivation after the treatment. This potential risk is dependent on the UV dose applied, the stability of added disinfectants, contact time, pH, temperature and the number and type of micro-organisms present in the wastewater. Moreover, physiochemical parameters of the treated effluent like turbidity, hardness, suspended solids, iron, manganese and humic-acids content can be disruptive factors preventing UV light waves from reaching all micro-organisms (Brahmi et al. 2010). After treating advanced municipal wastewater effluent with UV light, Guo et al. (2009) concluded that microbial communities change after the treatment in respect of the types of bacteria present but that the total amounts of bacteria in the water can increase to the same level as in non-disinfected effluent within only five days. UV irradiation therefore requires careful adjustment of the factors just described in order to ensure sufficient contaminant removal from wastewater (Guo et al., 2009).

To eliminate bacteria, viruses and protozoa for safe water reuse, chlorination is the most widely applied method. Chlorine is applied around the world for wastewater disinfection, as chlorine gas, hypochlorite solution or in solid form (Stefanakis 2016). Despite its effectiveness in destroying pathogens, chlorination does not come without potential risks. Harmless substances can react with the disinfectant and form harmful molecules, the so-called chlorination by-products (Jegatheesan et al. 2013). In addition, research has shown that some viruses and bacteria are resistant to chlorination. It is therefore advisable to combine this technique with additional and advanced treatment methods for safe water reclamation (Shareefdeen et al. 2016). Typical chlorine doses are 5-20 mg l^{-1} for a contact time of 30-60 min. If residual chlorine concentrations in the reclaimed water are too high for its intended reuse type, a dechlorination step is required. This can increase the cost of chlorination by about 20-30% (Lazarova et al. 1999).

Summary: water reclamation and reuse

In addition to specifically technology-related bottlenecks obstructing successful wastewater reclamation and reuse, there are also more general ones. Taken together, these indicate that such reuse might only be a valid option in water-constrained regions like Singapore or in delta zones where salt water is abundant but fresh water is not. One of the general bottlenecks is that potential users might be scattered across the city, requiring a dedicated distribution network. Since water reuse is rather a new concept in urban planning, current infrastructure seldom takes the distribution of reclaimed water into account. Consequently, there is little room to install a new separate pipeline network, whilst retrofitting is costly, impractical and inconvenient (Yi et al. 2011).

Beyond that, water reuse including a new distribution network may have a higher lifecycle impact than surface-water treatment and distribution via the conventional pipeline system. But if non-potable water qualities are produced, new distribution lines – and hence increased costs – are inevitable (Garcia and Pargament 2015). In Tokyo's Shinjuku district, a second pipeline system has been successfully installed to flush toilets with reclaimed wastewater. Due to the high density of high-rise buildings in this area, the pipes are mostly above ground in the buildings themselves. Compared with an underground network, this has kept costs relatively low (Lazarova et al. 2013). In cities that withdraw their water from aquifers or natural bodies of water, the recharge of those sources with reclaimed water (indirect reuse) might be the preferred option due to its much easier practicalities and lower costs, compared with building new distribution systems to reach end users. The Catalan Water Agency, for example, promotes aquifer recharge to prevent water scarcity during periods of drought but also to refill the aquifer as a hydraulic barrier against saltwater intrusion. A similar approach is implemented at the Torreele facility in Belgium (Van Houtte and Verbauwhede 2013). Ideally, potential large-scale water users like industries or farms should be located close to the WWTP so that they can be supplied through a single pipeline in order to keep distribution costs low (Wang et al. 2015b). In practice, however, the topographical location of WWTPs is usually down-gradient so as to make use of gravity for wastewater flow. This can make the distribution of reclaimed water costlier, because it needs to be pumped uphill back to the city or other areas of usage (McCarty et al. 2011). In addition, the temporal variability in demand for and the supply of reused wastewater is an important issue to consider in distribution planning (Garcia and Pargament 2015).

Another reported bottleneck in wastewater reclamation is health concerns, especially if the water produced is destined for direct or indirect potable reuse. When the water board in Amsterdam, the Netherlands, analysed and assessed potential alternative fresh-water sources, potable water reuse was evaluated as too risky. Since enough fresh water is already available in Amsterdam, anyway, other alternatives were chosen (Rook et al. 2013). However, the importance of social acceptance is illustrated by a case from San Diego, California, where 90% of the local water supply stems from sources several hundred kilometres away. A wastewater reclamation technology implemented here eventually had to be scrapped due to public safety concerns. Similar cases are reported form Toowoomba, Australia, and the Californian cities of San Ramon-Dublin and Los Angeles (Guest et al. 2009). When it comes to wastewater reclamation and reuse, it is widely agreed that, without public acceptance, it is difficult for any water management utility (WMU) to finance, construct and operate adequate processes to prevent future supply shortages during periods of drought. Social acceptance therefore needs to be perceived as a potential problem at an early stage in water reuse project planning. Public participation is essential to meet people's needs, to collect local knowledge so as to help improve the design of the project and to build vital institutional trust (Garcia and Pargament 2015). On the other hand, if citizens have experience of immediate and severe water shortages, their acceptance of such schemes rises even when these involve direct potable reuse. This has been the case, for example, with the system in place for almost 40 years now in Windhoek, Namibia (WWAP 2017). If shortages are not perceived as a threat, the willingness to pay for water services is low and that makes it difficult to implement reuse schemes which are cost effective (Bdour et al. 2009).

The use of reclaimed water for the irrigation of crops also entails potential risks, including the uptake by plants of sodium and other ions that can lead to yield losses, alter soil structures, change water infiltration rates and contaminate soils (Pedrero et al. 2010). Various cases have shown the significant contribution that reclaimed water can make to more sustainable agricultural production. Lazarova et al. (2013) describe a variety of successful reuse projects undertaken in co-operation with the agricultural sector. However, a lack of common legal standards and policies is a serious bottleneck obstructing the wider implementation of water reuse projects in Europe, because this increases planning and investment uncertainties (Fawell et al. 2016). Government policies to make water reuse an attractive business venture for financial service providers and investors are also needed in other parts of the world, such as China (Yi et al. 2011). In this context, it is welcome that the European Commission has proposed a European Innovation Partnership (EIP) for Water and identified wastewater reclamation and reuse as one of its top five priorities. In 2018 the Commission published an initial proposal for a regulation on minimum requirements for water reuse. Its general objective is to increase the uptake of this solution for agricultural irrigation wherever it is relevant and cost-effective (European Commission 2018).

Energy recovery technologies

Global energy demand is expected to grow by approximately 50% between 2010 and 2040, and fossil fuels will likely satisfy almost 80% of this. Consequently, fossil-related emissions are projected to increase in a similar range (EIA 2013). These projections drive the need to substantially decrease the energy intensity of WWTPs by designing treatment processes with a focus on energy efficiency and recovery. Currently, the treatment of municipal wastewater accounts for about 4% of the national electricity consumption in the United States (Wang et al. 2015a) as well as in the United Kingdom (Oh et al. 2010). As shown in Figure 2, the recovery of fuels from wastewater is achievable through the application of different technologies. The chemical energy in typical municipal wastewater has been measured to be 17.8 kJ/g COD (Heidrich et al. 2011). This is about five times the electrical energy needed to operate the conventional activated sludge (CAS) process (Wan et al. 2016). But in that a significant fraction of the energy stored in the COD is lost as heat during microbial metabolism (Frijns et al. 2013). Its current configuration hardly achieves energy self-sufficiency, which usually ranges between 30 and 50% (Wan et al. 2016), depending on the country concerned.

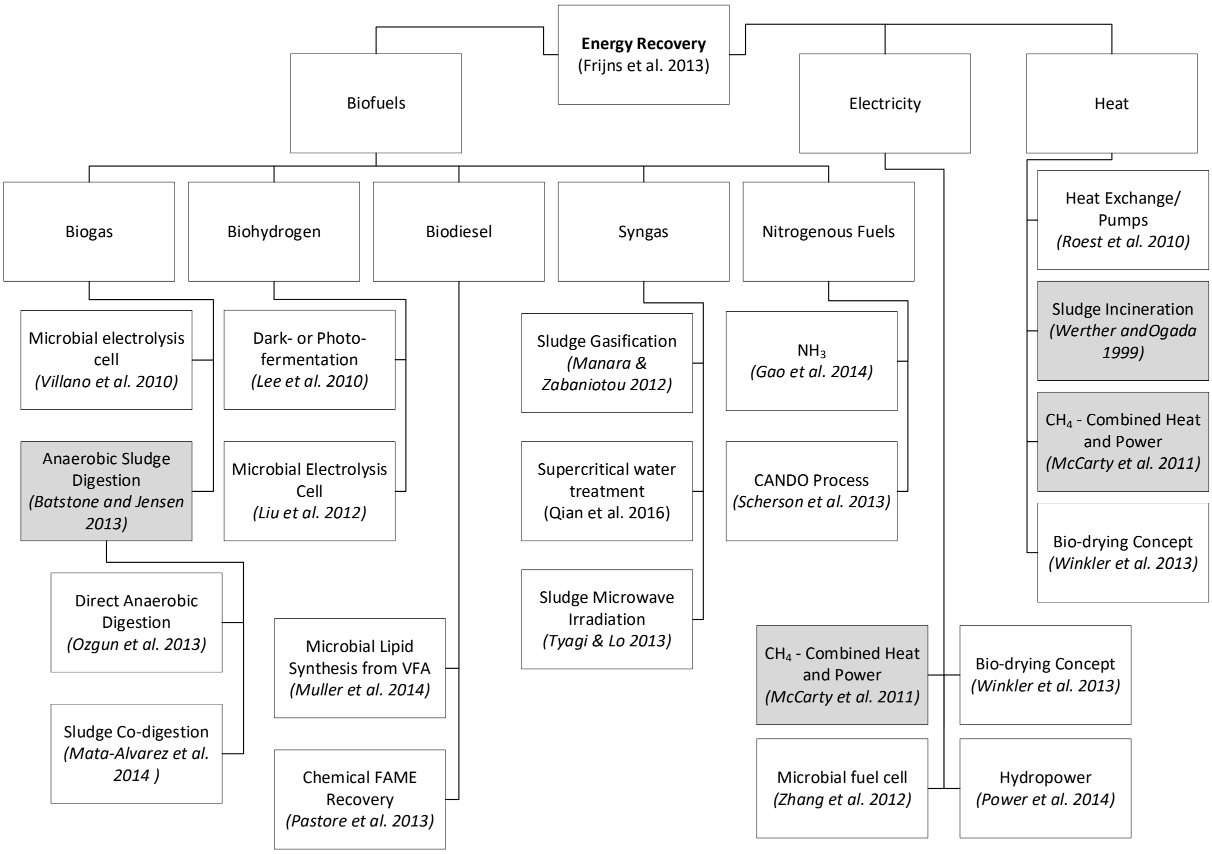

Figure 2. Examples of technologies to recover energy from municipal WWTPs. Since a detailed presentation and discussion of each technology is beyond the scope of this paper, a scientific publication that explains or reviews it further is referenced. Grey shading indicates techniques that have been applied on a large scale at municipal WWTPs. Unshaded boxes show technologies that are not applied widely.

Methane

The production of biogas by anaerobic sludge digestion is currently the most widely used energy recovery method, applied worldwide on different scales (Rulkens, 2008). About 80% of the biodegradable COD fraction in the sludge can be converted into harvestable biogas in completely mixed reactors (McCarty et al. 2011). In advanced reactor configurations, biodegradation efficiency and the recovery of dissolved methane from the broth may be improved (Ma et al. 2015). If the recovered methane is not used on site, it needs to be pressurised and transported to potential customers. This can be too expensive in countries where CH_4 is cheaply available and distributed using a comprehensive pipeline grid (Rabaey and Rozendal, 2010). One important cost factor with digesters is heating, since at moderate temperatures up to 40% of the produced methane is dissolved in the broth.

This dissolved methane might ultimately contribute to climate change. Anaerobic wastewater treatment and sludge digestion therefore need to be properly controlled in order to minimise the risk of methane leakage (Frijns et al. 2013).

One promising concept to maximise the recovery of biogas is maximum COD capture at the entrance of the plant, followed by digestion of the primary sludge (Frijns et al. 2013). Up-concentration of COD can be achieved by applying either chemically-enhanced primary treatment or high-rate activated sludge as an A stage in a WWTP (Wan et al. 2016). On average, plants applying this energy-recovery route consume 40% less net energy (Frijns et al. 2013). But using the generated biogas for combined heat and power recovery implies high energy conversion losses of about 60%. Converting 60% of influent COD with anaerobic digestion and CHP generates only approximately half of the energy required for total COD removal as part of a CAS process (Wan et al. 2016).

It is also possible to treat wastewater directly, anaerobically. For example, in anaerobic membrane bioreactors (AnMBRs) or up-flow anaerobic sludge blanket reactors (UASBs). These processes may provide low-energy carbon removal, but they also require additional post-treatment steps due to insufficient pathogen removal (Batstone et al. 2015). The organic carbon concentrations in municipal wastewater, however, are too low for direct anaerobic treatment. Consequently, anaerobic digesters are only ever used in large conventional plants for treatment of the sludge line, not the water line (Logan and Rabaey, 2012).

Other biofuels

As well as methane, other fuels can also be recovered from municipal wastewater streams. In conventional biofuel production using sugar, 40-80% of the overall production costs are related to the feedstock alone. Converting wastewater COD into biofuels may therefore offer significant economic potential (Chang et al. 2010), although downstream processing and the high dilution of recoverable matter remain major challenges (Puyol et al. 2017). However, syngas can be produced by fast gasification of wet sewage sludge (Manara and Zabaniotou 2012) – a thermal conversion process which converts any carbonaceous material into, for the most part, carbon monoxide and hydrogen in a controlled oxygen environment, sometimes at high pressures of 15-150 bar (Sohi et al. 2009). If sewage sludge-derived syngas is used as a fuel, it needs to be cleaned as it contains undesirable impurities that may damage fuel cells, engines or turbines (Manara and Zabaniotou 2012).

Syngas can also be obtained from municipal sewage sludge using supercritical water treatment processes. During supercritical water gasification or partial oxidation processes, the temperature and pressure are raised above the critical point of water (374 °C, 221 bar). In these conditions, biomass is converted into syngas at high rates and energetic efficiencies. In addition to syngas, a disposable clean-water stream and solids (metal oxides, salts) leave the process (Goto et al. 1999). The advantage over other sludge-handling technologies is that the sludge is converted into an energy carrier in much shorter residence times of only a few minutes. Moreover, excess sludge from WWTPs does not need to be dewatered before being fed to supercritical water reactors (Yakaboylu et al. 2015). Although existing thermodynamic equilibrium models can predict the major product compounds formed in reactors, not all parameters determining the final gas composition are yet clear. One operational challenge is corrosion of the reactors due to harsh operating conditions. Another is salt precipitation and clogging due to the rapid decrease in the solubility of salts in supercritical water conditions (Yakaboylu et al. 2015). Several commercial applications have partially demonstrated the economic feasibility of the process (Qian et al. 2016). Possible success and failure factors, COD destruction efficiencies and research needs in respect of commercial processes have already been reported and reviewed elsewhere (Qian et al. 2016).

Hydrogen can also be recovered from wastewater by biological means, in a two-step anaerobic sludge treatment process limited to hydrolysis and acidogenic fermentation by phototropic and/or lithotrophic micro-organisms. Photofermentation is frequently employed together with dark fermentation because the latter converts only about one third of the COD into hydrogen and the rest into VFA, which can subsequently be used in photofermentation to enhance overall hydrogen production (Lee et al., 2014). However, the major bottleneck in fermentative H_2 production is quite low yields (Lee et al. 2010).

Another fuel that can be derived from sludge is biodiesel. Lipids can represent a significant proportion of the organic fraction in municipal wastewater and specialised micro-organisms can assimilate and accumulate these anaerobically. Harvesting this lipid-rich biomass by simply skimming the surface of wastewater treatment reactors could provide feedstock for high-yield biodiesel production (Muller et al. 2014). The use of phototrophic microalgae that treat the wastewater in high-rate ponds is a well-studied production route for biodiesel (Puyol et al. 2016). One major bottleneck, however, is that the performance of phototrophic organisms depends on climatic conditions that are not available all year round in countries with a winter season (Khiewwijit et al., 2016). In addition, land use for this type of biodiesel production is high (Park et al. 2011), as are the costs of photo-bioreactors and algae harvesting (Gao et al. 2014).

Nitrogenous fuels can also be recovered from wastewater. One route for this is the so-called CANDO process, which involves three steps: (i) nitritation of NH_4^+ to NO_2^-; (ii) partial anoxic reduction of NO_2^- to N_2O; and (iii) chemical N_2O conversion to N_2 with energy recovery. Another route recovers NH_3 directly from concentrated side streams – by stripping, for example. NH_3 can be burned to generate power or used as a transport fuel. It can even be converted, by nitritation and further abiotic or biological reduction, into N_2O for co-combustion with methane recovered by sludge digestion. Generally, however, processes recovering ammonia for fuel consume more energy than they recover. For this reason, recovering ammonia for fertiliser would seem preferable. The major problem with these routes is the low N concentrations in municipal wastewater, particularly in combination with high process costs (Gao et al. 2014).

Sludge incineration

When sewage sludge is incinerated, complete oxidation of its organic content is achieved, thus forming CO_2, water and inert material (ash) that have to be disposed of. The latter can, for instance, be used as aggregate for building materials (Tyagi and Lo 2016). The combustion heat can be recovered as electricity. Raw sewage sludge has a 30-40% higher heating value than digested sludge, which makes it theoretically attractive as a combustion fuel to produce electricity. Whether sludge digestion or incineration is the energetically favourable route, however, depends on specific and local conditions like the treatment system, the methods used for sludge drying and the type of incineration (Frijns et al. 2013). Various plant configurations for the large-scale combustion of biomass, including dried sewage sludge, are applied worldwide and recover energy from the organic matter. Typical electrical efficiencies of stand-alone biomass combustion plants range between 25 and 30%. To be economically viable, such plants rely on low cost fuels, carbon taxes or fixed tariffs for the electricity they generate. Fluidised bed technology in combustion plants can improve electrical efficiencies to 40%, at lower cost and with higher fuel flexibility. Co-combustion of sludge in coal-fired power plants is another method widely applied in the EU, and it achieves similar efficiencies (Faaij 2006).

The major drawback of sludge incineration is the typically high water content of waste sludge. To reach a positive energy balance from combustion, that needs to be reduced below 30% – which usually requires energy and therefore creates costs (McCarty et al. 2011). The actual potential of sludge incineration is much lower than the energy content of the organic matter in the sludge because a lot of energy is required to evaporate its water content (Frijns et al. 2013). As a solution to this problem, significant heat energy can be recovered from WWTP effluent by heat-exchanger and heat-pump systems (Tassou 1988). To improve the heating value of waste sludge, this low-cost heat can be supplied to dewatering and drying systems at the plant.

Bioelectrochemical systems

In bioelectrochemical systems (BES), COD is oxidised by micro-organisms and the electrons generated during this process are then used to produce energy or other valuable compounds (Wang and Ren, 2013). Within these systems, microbial electrosynthetic processes can take place in which the electricity-driven reduction of CO_2 and the reduction or oxidation of other organic feedstocks like wastewater occur. A BES consists of an anode compartment, a cathode compartment and a membrane separating the two. An oxidation process (e.g. wastewater or acetate oxidation) occurs on the anode side, and reductive reactions (e.g. O_2 reduction or H_2 evolution) on the cathode side (Rabaey and Rozendal, 2010). Since electrons are donated to or received from electrodes, redox balances can be achieved by micro-organisms without the oxidation of substrates or the production of reduced by-products (Puyol et al. 2017). Electrons can be transferred either directly between the cell and the electrode or via soluble molecules able to become reduced and oxidised and to receive electrons from cells to transport them to the electrode, and vice versa. The efficiency of a scaled-up BES depends strongly on those electron transfer rates, which current research efforts are seeking to maximise (Logan and Rabaey, 2012).

A BES can be operated in three modes.

- As a microbial fuel cell (MFC) to deliver electricity directly.
- As a microbial electrolysis cell (MEC) in which the anode and the cathode are connected without a resistor.

- As an MEC into which power is invested to increase the reaction rate and/or to enable thermodynamically unfavourable reactions (Rabaey and Rozendal 2010).

In addition to electricity generation, in theory three product groups are particularly suited to wastewater resource recovery by means of a BES, in that this offers real advantages over conventional production techniques.

- Bulk chemicals like biofuels, platform chemicals and plastics.
- High-value chemicals like pharmaceutical precursors, antibiotics and pesticides.
- Inorganics like nutrients, which can serve as fertilisers and so on (Puyol et al. 2017).

Despite remarkable research progress, major bottlenecks hindering large-scale BES-based wastewater resource recovery are high overall costs (especially for expensive metal catalysts and membranes) and the fact that most research is limited to lab-scale applications. Outside the laboratory, the performance of pilot plants remains unstable due to water leakage, low power output, influent fluctuations and unfavourable product formations. To become a viable alternative to conventional wastewater treatment, BES need to be scaled up to at least cubic-metre proportions, with reactor configurations that allow easy integration into current plant designs and infrastructures (Wang and Ren, 2013). Due to these technical bottlenecks and the low value of electricity, energy recovery by BES is considered likely to remain, at best, a niche application in wastewater treatment (Kelly and He, 2014). As for BES-based H_2 production, limited rates of microbial metabolism and rather restricted physical and chemical operational conditions are severe limitations (Schröder 2008). Moreover, MECs cannot compete with methane production in conventional anaerobic digesters, even at moderate temperatures (Clauwaert and Verstraete 2009). Consequently, methane production via electromethanogenesis is most unlikely to replace anaerobic digestion for methane recovery from high-strength wastewaters (Cheng et al. 2009; Villano et al. 2013). To sum up, bioelectrochemical routes are still far from being an practical solution for resource recovery at WWTPs.

Thermal energy

Municipal wastewater contains 2.5 times more thermal energy than the theoretical maximum chemical energy stored in the COD (assuming a 6 °C effluent temperature change) (Ma et al. 2013). Thermal energy in WWTP effluent stems from household and industrial water heating and, marginally, from microbial reaction heat released during the treatment process (Hartley 2013). Since the temperature of the effluent shows relatively small seasonal variations by comparison with atmospheric temperatures, it can serve as a stable source of heat recoverable using heat pumps. It is recommended that the effluent be used as an intake source for heat pumps because the influent still contains many contaminants that can cause fouling problems in the equipment. In addition, the decrease of the influent temperature by heat exchangers may adversely affect biological reactions during treatment (Chae and Kang 2013). Heat pumps use electricity to extract low-temperature thermal energy from the wastewater and usually provide 3 or 4 units of heat energy per unit of electrical energy consumed (Mo and Zhang 2013). In addition to heating or cooling buildings, one potentially interesting on-site use of recovered thermal energy is sludge drying.

As with water reuse, however, the potential mismatch between supply and demand in terms of time and location represents a potential bottleneck hindering thermal energy recovery. One possible solution to this problem is the use of thermal energy storage facilities like aquifers (van der Hoek et al. 2016). Selling surplus heat to nearby consumers is recommendable, but especially in the spring and autumn demand may be insufficient due to a reduced need for district heating or cooling (Chae and Kang 2013). In 2008 it was reported that more than 500 heat pumps for wastewater were already operational, with capacities of 10-20 MW (Schmid 2008). Large-scale district-heating systems using thermal energy derived from wastewater have been established in many parts of the world (Mo and Zhang 2013). Especially in Japan, it has been shown that heating and cooling systems using wastewater can reduce energy consumption substantially. In Osaka, for example, energy savings by the city government reached 20-30% as a result of introducing thermal energy recovery from effluents. In the city of Sapporo, effluents are used directly to melt large quantities of snow every winter (Shareefdeen et al. 2016).

Hydropower

Applying hydropower technologies to effluents is a well-known means of recovering electricity by taking advantage of constant discharge from WWTPs and, depending on the location, a certain hydraulic head. Useful technologies range from the Archimedes screw to water wheels and turbines, all of which display reliable performance when applied to an effluent flow. If such technologies are applied to untreated wastewater, materials like stainless steel are required to prevent corrosion (Berger et al. 2013). The power output of a hydropower technology depends on the rate of flow and the hydraulic head. As with any other energy-recovery route, its economic viability is also influenced by non-technical factors such as electricity prices, taxes, financial incentives and the cost of connection to the power grid. If the recovered electricity is used on site, the system becomes economically more attractive when energy prices rose. Economic viability is always site-specific, therefore, and depends on physical circumstances like the technology selected, not to mention market conditions – future as well as present (Power et al. 2014). Although individual large-scale applications in Australia, the UK and Ireland have proven the economic viability of hydropower technologies at WWTPs, a detailed analysis of this factor is lacking in most of the scientific case studies. The most important parameter for the hydropower potential of a WWTP effluent stream is the rate of flow, which is subject to seasonal, economic, infrastructural and demographic variations. Installations are usually designed for a defined flow and pressure, and so these parameters should be kept as constant as possible in order achieve consistent performance (McNabola et al. 2014).

Summary: energy recovery

Although complete recovery of all the energy contained in wastewater may be unrealistic due to conversion losses, energy-neutral or even energy-positive WWTPs are increasingly becoming practicable (Gao et al. 2014). At least 12 plants in Europe and the USA have been reported as reaching more than 90% energy self-sufficiency (Gu et al. 2017). The European research project Powerstep is currently elaborating designs for energy-neutral and energy-positive WWTPs through six different case studies (Ganora et al. 2019). The recovery of methane to generate electricity can usually offset 25-50% of a WWTP's energy needs, assuming that conventional treatment technology is used (McCarty et al. 2011). If thermal energy recovery from effluent is applied along with chemical energy recovery, carbon neutrality or better can be achieved (Hao et al. 2015). However, the heavy focus by the water industry on energy sustainability has also been criticised as misleading because, it is argued, wastewater treatment should prioritise the optimisation of the hydrological cycle over energy and climate concerns (Guest et al. 2009). Moreover, materials rather than energy can also be recovered from COD. This aspect is gaining increasing attention, as discussed below.

Fertiliser recovery technologies

WWTPs are linked to global nutrient cycles because a fraction of the N and P applied as fertiliser in agriculture ends up in the wastewater stream (Daigger 2009). One global estimate suggests that fertiliser production accounts for more than 1% of the world's demand for energy and emissions of anthropogenic greenhouse gas (GHG). Over 90% of these emissions are related to ammonium fertiliser production (Sheik et al., 2014). From a resource-efficiency perspective, it is a paradox to produce ammonia fertiliser by the Haber-Bosch process, with its high energy consumption, and then to destroy it again after use at WWTPs by biological nitrification and denitrification, which also consume large amounts of energy. Ammonia recovery therefore offers potential energy savings, as long as it can be achieved with lower energy consumption than industrial production (Daigger 2009).

Compared with N, the recovery of P is much more urgent because it is a finite resource with projected scarcity (Khiewwijit et al., 2016). P mining from rocks has a huge environmental impact due to its generation of by-products like gypsum, which are often contaminated with radioactive elements and heavy metals and are not disposed of in an environmental friendly manner (Verstraete et al. 2009). P enters the wastewater stream in faecal matter, household detergents and industrial effluents (Sedlak 1991), at a typical concentration of about 6 mg P l^{-1} (Xie et al. 2016). If influent P is not removed during the treatment process, it can reach bodies of surface water and cause their ecological destruction (Cordell et al. 2009).

Nutrient-recovery technologies have been studied widely, resulting in a variety of solutions (Figure 3). Since the efficiency of nutrient recovery typically decreases with lower concentrations in the wastewater stream, a sequential three-step framework has been recommended (Mehta et al. 2015):

1. Nutrient accumulation by either biological, chemical or physical methods.
2. Release of nutrients by either biological, chemical or thermal methods.
3. Nutrient extraction and recovery in the form of concentrated products, by chemical or physical methods.

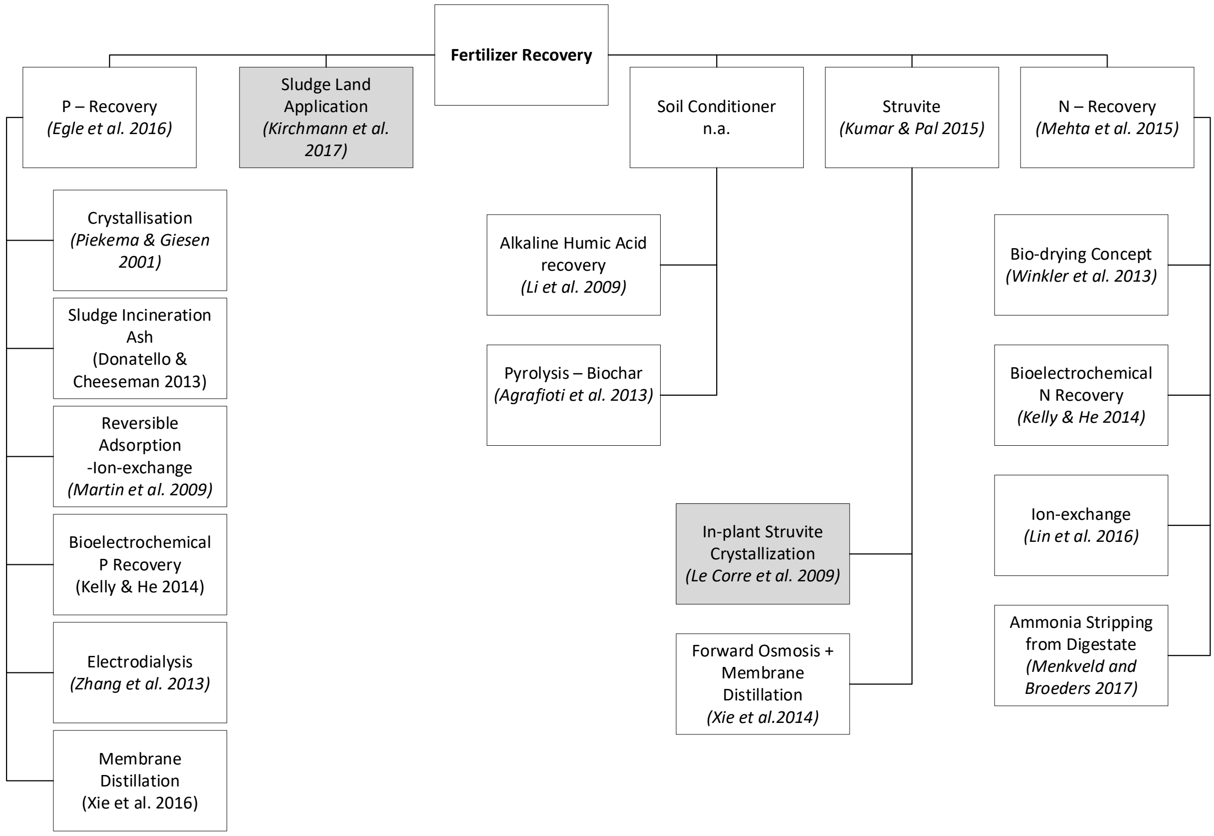

Figure 3. Examples of technologies to recover fertiliser from municipal WWTPs. Since a detailed presentation and discussion of each technology is beyond the scope of this paper, a scientific publication that explains or reviews it further is referenced. Grey shading indicates techniques that have been applied on a large scale at municipal WWTPs. Unshaded boxes show technologies that are not applied widely.

Sludge land application

Currently, wastewater fertiliser recovery takes place either indirectly through struvite precipitation or directly by spreading sewage sludge onto agricultural land (Van Leeuwen et al. 2016). About 40% of all sludge generated in the EU is recycled using the latter method (Wilfert et al., 2015). However, contamination can be a problem when sludge is applied to arable land. High contaminant loads have been found in bacterial biomass leaving WWTPs as secondary sludge (Sheik et al., 2014). Unfortunately, moreover, sludge has a low nutrient content and is therefore a low-quality fertiliser compared with conventional fertiliser products. Nevertheless, it can still contribute towards the stabilisation of soil's organic carbon content. The transportation of dewatered sludge to the field can also be a bottleneck, since it is expensive due to the product's high water content, 70-90% (Kirchmann et al. 2017).

Struvite

Struvite precipitation as a recovery route for ammonia and phosphate has gained a lot of interest in research in recent decades, and is applicable on large scale (Le Corre et al., 2009). Struvite is magnesium ammonium phosphate ($MgNH_4PO_4 \cdot 6H_2O$), a mineral commonly formed at WWTPs through spontaneous precipitation if Mg concentrations are high enough – although this is often not the case. The formation and growth of struvite crystals at WWTPs is affected by various parameters, such as pH, temperature, mixing energy and turbulences and the presence of other ions like calcium or carbonates (Jaffer et al., 2002). Struvite precipitation is usually introduced to solve operational problems, in particular the clogging of equipment (Zhang et al. 2013). The N and P fractions in struvite are slowly soluble, which makes it usable as a slow-release commercial fertiliser suitable for soils with low pH value (Sheik et al., 2014; Xie et al., 2016).

It has been shown that effective struvite precipitation can only be achieved if P concentrations are above $100mg/l^{-1}$, and also depends on ammonium concentration and pH value. Lower P concentrations lead to significantly lower recovery rates and longer precipitation reaction times, and require higher pH values. Consequently, struvite precipitation is probably not feasible for wastewater with low phosphate-P concentrations (Xie et al., 2016; Zhang et al., 2013). Usually, nutrient enrichment is required prior to struvite precipitation and recovery from side streams at WWTPs. Using an enhanced biological phosphorous removal (EBPR) process like supernatant from anaerobic sludge digestion or sludge dewatering processes is most feasible. In most cases, Mg salt has to be added to fully remove soluble P as struvite from these streams (Münch and Barr 2001). The majority of WWTPs, however, have chemical P-removal systems which preclude struvite formation (Wilfert et al., 2015). Due to those wastewater P fractions which are fixed in biomass or bound to metals like Fe and consequently unavailable for struvite formation, the efficiency of the recovery of influent P as struvite is usually only 10-40% (Cornel and Schaum 2009). Even if favourable conditions for struvite precipitation, such as low total suspended solids (TSS) and high solubilised NH_4^+ and PO_4^{3-} concentrations, are established intentionally by continuously removing biomass (Sheik et al., 2014), the recoverable amounts are rather low and unlikely to exceed 1kg of struvite per $100m^3$ of wastewater (Shu et al. 2006).

Le Corre et al. (2009) reveal that the cost of recovering struvite after sludge digestion with the aid of chemical additives (e.g. magnesium salt), including manpower and maintenance, could reach €2 per $m^{-3\ of}$ raw wastewater. This is economically unviable. The cost-effectiveness of struvite recovery from the water line without prior P concentration by EBPR or chemical P removal (CPR) has not been calculated (Khiewwijit et al., 2016). However, since struvite recovery can significantly reduce volumes of sludge due to its subsequent enhanced dewaterability, this technique may decrease sludge handling and disposal costs (Le Corre et al., 2009). In addition, it prevents the clogging of pipes (Zhang et al., 2013). These operational cost benefits should also be included when the cost-effectiveness of struvite recovery is assessed. The market value of struvite, as a relatively new fertiliser, is uncertain and may be influenced by rates of production and regional demand (Le Corre et al., 2009). In addition, fractions of heavy metals and organic contaminants present in wastewater could end up in the product and so limit its safe agricultural application (Xie et al., 2016). Lin et al. (2013a), for example, have revealed that recovered struvite crystals can contain arsenic concentrations of up to 570 mg/kg^{-1}. Successful struvite recovery can also be hindered by a lack of legal regulation. After it was first successfully recovered in the Netherlands in 2006, it took about ten years before the legal framework was finally adjusted to allow the application of struvite in agriculture (van der Hoek et al. 2016). Even since the law was changed in its favour, however, no breakthrough in the implementation of struvite recovery yet seems to have occurred. It must therefore be questionable how severely that legislative bottleneck actually impacted use of the technique.

Sludge incineration ash

Technologies that recover P from sludge incineration ash are currently in focus because they promise high influent-P recovery rates. They do require special incinerators in order to obtain high recovery efficiencies, though, and these can be very costly (Wilfert et al. 2018). Moreover, this technique is still under development and not all its pros and cons are yet known. But one clear advantage over other P recovery routes is that it occurs at the very end of the process and so does not conflict with other measures taken at the WWTP (van der Hoek et al. 2016). Like the use of sewage sludge in the environment, however, ash is associated with heavy metal contamination. Whilst chemical extraction can be used to obtain pure phosphates from it, post-treatment of the treated ash – at greater cost – may then be required for heavy metal removal. Alternatively, ashes can be used in the construction industry without any pretreatment. But this does not involve P recovery (Mehta et al. 2015).

Soil conditioner

Used alongside mechanical and thermal methods, alkaline treatment is a simple and highly efficient chemical means of disintegrating sludge. Apart from reducing the volume of the sludge even further after conventional dewatering processes have been applied, it also responds to the fact that the released water contains large amounts of dissolved organics like proteins, humic acids, lipids and polysaccharides, plus residual NaOH. Most of these can be degraded further by subsequent treatment processes, but in the case of humic acids that is more difficult due to their high recalcitrance to microbial degradation. Applied as a soil conditioner, humic acids contribute to the slow release of nutrients and high cation-exchange and pH-buffer capacity, as well as the retention of heavy metals and xenobiotics in soils (Réveillé et al. 2003). The extraction of humic acids from alkaline sludge treatment supernatant can be achieved by membrane filtration with a 45µm mesh (Li et al., 2009), but the cost-effectiveness and detailed impact of humic-acid recovery remain to be analysed.

Another soil conditioner recoverable from sewage sludge is biochar, which could alternatively also be used as a coal-like fuel. The production of biochar and its storage in soils is often suggested as a potential means to sequester atmospheric carbon (Woolf et al. 2010). Biochar is obtained from sludge pyrolysis, which is the process of thermally cracking organic matter via an external heat source and without the supply of air (Chun et al. 2013). As well as carbon sequestration, biochar's potential addition to soils is associated with a wide range of other possible secondary benefits like the liming of acidic soils, reducing plant aluminium availability, increasing cation-exchange capacities, reducing nutrient leaching, remediating sites contaminated by heavy metals and chemicals, increasing agrochemical sorption and reducing net GHG emissions from soil (Spokas 2013). In general, though, our understanding of the impact of biochar on single or combined soil attributes remains poor. Because of this, the consequences of its application for crop yields and its related potential impact on global warming are hard to predict and very site-specific (Jeffery et al. 2011).

Membrane-based nutrient recovery

Electrodialysis, membrane distillation and forward osmosis are emerging nutrient-recovery technologies, reviewed extensively by Xie et al. (2016). The attractiveness of membrane-based technologies for wastewater nutrient recovery lies in the separated streams of concentrated nutrient ions and the abatement of chemical products for ion precipitation (Korzenowski et al. 2014). But no detailed techno-economic analyses revealing demand for energy, CO_2 footprint, system robustness, operating costs, product quality and market demands are yet available. These technologies therefore remain a fairly theoretical option, still a long way from practical application at large-scale wastewater treatment facilities (Xie et al., 2016).

Summary: fertiliser recovery

One general bottleneck hindering energy and cost-effective nutrient recovery from wastewater is the rather low quantities obtainable, certainly by comparison with industrial fertiliser production systems, giving this route a competitive disadvantage (Khiewwijit et al., 2016). Numerous new P-recovery technologies have been developed for various access points at WWTPs, and in some cases actually implemented at full scale in recent years. A thorough assessment of these emerging routes is provided by Egle et al. (2016). Since global demand for fertiliser is expected to increase by 4% a year due to population growth (Elser and Bennett 2011), it can be expected that P fertiliser recovery from wastewater will gain further importance in the future. Its cost, however, is likely to exceed that of P ore-derived fertiliser products several times over, as shown by Cornel and Schaum (2009) for German market conditions. As well as conventional fertiliser products, manure from livestock production also competes with nutrients recovered from wastewater. Coppens et al. (2016) show that, in Flanders, P entering WWTPs could fulfil 14% of total local fertiliser P demand while P contained in manure could easily satisfy this demand alone (Coppens et al. 2016). It is therefore likely that wastewater-derived P fertiliser is redundant in livestock-intensive regions, as shown in Figure 4.

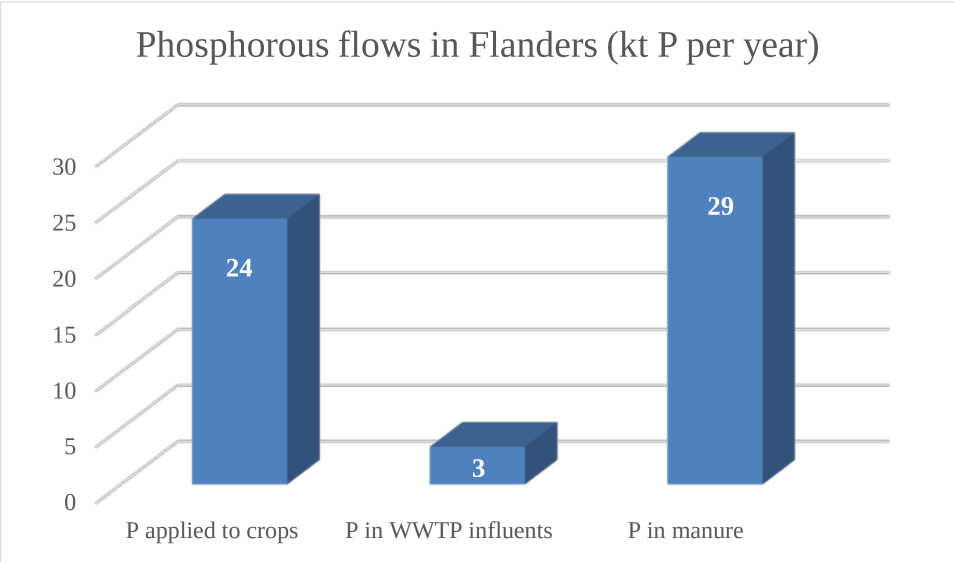

Figure 4. Phosphorous flows (kt/yr^{-1}) in the livestock-intensive region of Flanders (Belgium), based on Coppens et al. (2016).

However, P can be recovered at a WWTP at different stages in the process. Although 30% of influent P is not solubilised as phosphate (PO_4^{3-}) but bound to organics, much of the remainder will likely solubilise by hydrolysis in the primary clarifier at the start of the process (Henze and Comeau 2008). After primary treatment, therefore, P is predominantly present in the liquid phase. Following secondary treatment with either EBPR or/and CPR, 90% of the influent P is contained in the sludge as either metal phosphates or polyphosphate in biomass. It might therefore be most efficient to apply a recovery step after the biological treatment process – for example, recovery from sludge incineration ash. This can achieve a recovery rate of up to 90% (Cornel and Schaum 2009).

The recovery of N from municipal wastewater could save fossil energy used to produce N fertilisers by the highly energy-intensive Haber-Bosch process (Khiewwijit et al., 2016). Usually, at least 75% of WWTP influent N is solubilised ammonium (NH_4^+) (Henze and Comeau 2008). This fraction is highly diluted, which makes ammonium recovery an energy-intensive process and thus too costly (Kuntke et al. 2012). At typical municipal wastewater concentrations of 20-70 mg/N l^{-1}, physical-chemical ammonia recovery technologies (e.g. stripping and thermal evaporation) would not be economical. During the CAS process, ammonia is converted biologically into nitrogen gas released into the atmosphere. The 25% organic influent N consists partly of urea and hydrolysed proteins, both also present in a solubilised form. Consequently, the reported values of influent-N fractions that end up as organic N in the sludge during the CAS treatment are only about 20% (Siegrist et al. 2008; Matassa et al. 2015). Current N-recovery technologies are usually limited to this minor N fraction. Because of this, in recent years greater attention has been paid to more energy and carbon-efficient biological N removal technologies, such as the combined nitritation-anammox processes, rather than N-recovery practices (Khiewwijit et al., 2016). However, an extensive overview of economic N-recovery constraints has been produced and still appears to be valid (Wilsenach et al. 2003).

Product recovery technologies

Besides nutrients, various other products can be recovered from wastewater, as shown in Figure 5. A number of publications point out the potential contribution towards sustainable development achievable by applying product recovery technologies at WWTPs (van Loosdrecht and Brdjanovic 2014; Van der Hoek et al. 2015; Puyol et al. 2017). Although some of these routes are attracting increased interest in terms of upscaling their applications, none is yet reported as being widely used.

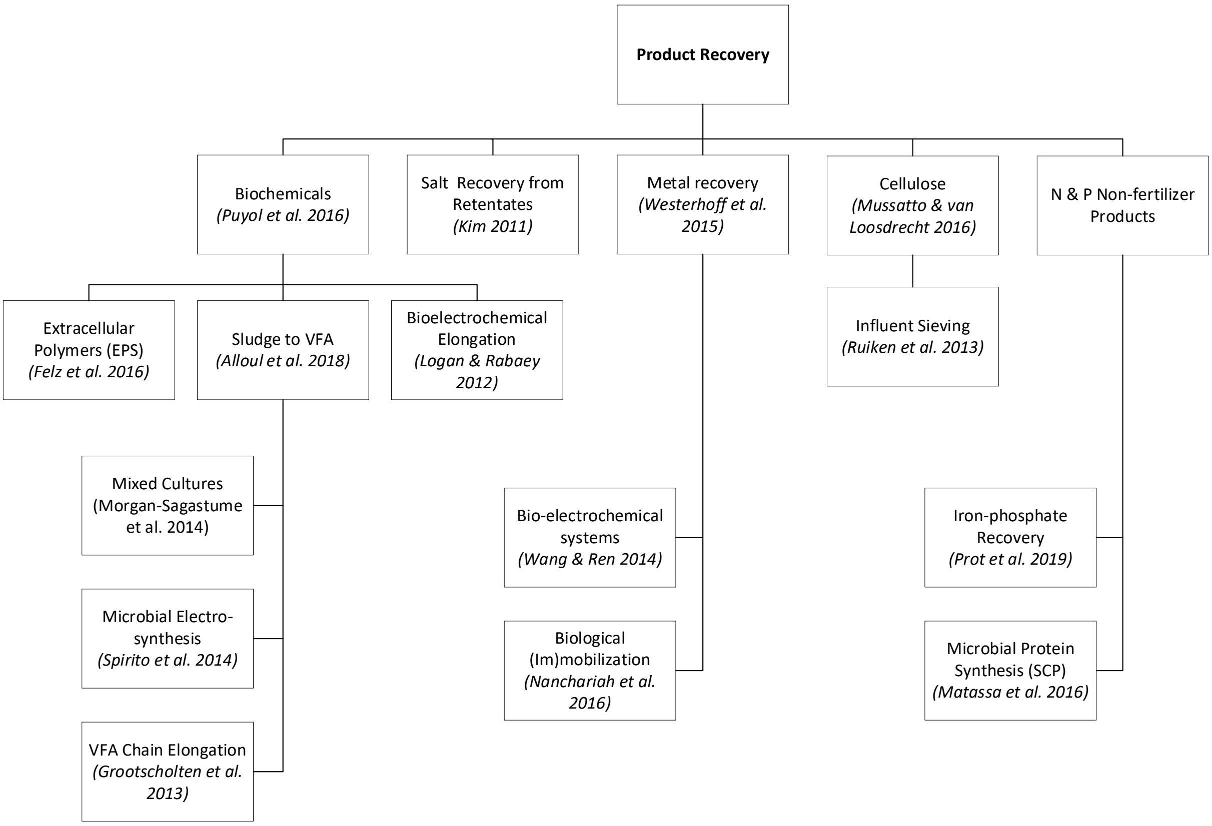

Figure 5. Examples of technologies to recover products from municipal WWTPs. Since a detailed presentation and discussion of each technology is beyond the scope of this paper, a scientific publication that explains or reviews it further is referenced.

Volatile fatty acids

One possible product recovery route is the integration of the carboxylate platform into wastewater treatment systems. Carboxylates are dissociated organic acids that can be produced by hydrolysing and fermenting primary sludge with undefined mixed microbial communities. To achieve that, it is necessary to inhibit methanogenic bacteria accumulation by applying a short sludge-retention time (SRT) to wash slow growing methanogens out of the reactor, and/or by establishing a very high pH value during fermentation (Chen et al. 2007). Important products of these procedures include VFAs, which consist primarily of the short-chain fatty acids acetate, propionate, lactate and n-butyrate. These are valuable products when separated from the fermentation broth because they act as substrates for secondary fermentation and electrochemical or thermochemical refinements to higher-value chemicals like fuels or bioplastics (Agler et al. 2011). VFA recovery from primary sludge can be improved either by adding activated sludge to the fermentation broth (Ji et al., 2010) or by use of a surfactant like sodium dodecylbenzene sulphonate and by maintaining a high pH value during fermentation (Khiewwijit et al., 2016). The fermentation liquids from a VFA fermenter can be used for treatment process optimisation, as they contain easily biodegradable carbon sources useful for biological nitrogen and phosphorous removal (Ji et al., 2010; Lee et al., 2014; Longo et al., 2015). Another advantage of VFA fermentation is the reduction of excess sludge quantities and of the associated disposal costs (Jie et al. 2014).

Kleerebezem et al. (2015) state that controlling the product spectrum in open-culture fermentation systems remains a major bottleneck in VFA recovery from waste streams, especially for products derived from carbohydrates. Another is the solubility of VFAs, because this leads to difficulties in efficient downstream processing (Grootscholten et al. 2013). VFAs can be distilled off the fermentation broth under atmospheric pressure, but that requires too high an input of energy to be economical (Chang et al. 2010). The same applies to the concentration of VFAs through nanofiltration or liquid-liquid extraction, whereas anion exchange might well be a more feasible downstream solution. Another possibility is to convert VFAs directly after fermentation, into an end product which is then separated from the liquid (Kleerebezem et al., 2015). However, studies examining all pertinent parameters of VFA production routes from waste streams remain very limited and most of variables have yet to be examined satisfactorily. Such uncertainties contribute to the fact that most waste-based VFA production concepts are still confined to laboratories (Lee et al. 2014).

Although it is evident that higher added-value products can be derived from VFAs, this does not imply that waste-based VFA production is economically preferable over methane generation. Only if calculations consider the costs of bioprocess operations and downstream processing, as well as potential subsidies for biogas production, can an economically substantiated decision be made (Kleerebezem et al., 2015). As an economically feasible recovery route with municipal wastewater, Khiewwijit et al. (2016) propose a COD up-concentration step with subsequent alkaline VFA fermentation. If COD is up-concentrated and fermented to VFAs, denitrification might underperform due to the lack of an easily degradable carbon source. Because of this, the development of N-removal processes that perform sufficiently at low COD concentrations is required (Alloul et al. 2018).

Polyhydroxyalkanoates (PHA)

One possibility for the refining of VFAs is to convert them into PHAs, which are fully biodegradable biopolyesters able to substitute fossil-fuel derived polymers. Due to their comparable properties and their potential for use as basic compound of polymeric compositions, PHAs are often referred to as bioplastics. PHAs act as carbon/energy storage polymers for more than 300 species of bacteria and archaea. These species can produce and store high concentrations of a PHA inside their cell (Laycock et al. 2013). Mixed-culture PHA production from wastewater and other organic waste streams is currently achieved using a three-step procedure.

- COD is fermented in an acidogenic reactor to produce VFAs.
- PHA-producing biomass is established and maintained in a separated reactor.
- Finally, the biomass is fed with the VFAs in a third reactor until the PHA content of the selected community is maximised (Moralejo-Gárate et al. 2014).

However, the PHA yield on the substrate and the efficiency of the downstream processing lead to costs 20-80% higher than those for petrochemical polymers of comparative quality (Fernández-Dacosta et al. 2015). Recovered bioplastics are not yet cost-competitive and therefore have limited market potential (van der Hoek et al. 2016). The development of new PHA utilisation routes and marketable applications remains a challenge for the future (Tamis and van Loosdrecht 2015).

Carbon-chain elongation

One rather innovative route for refining wastewater-derived VFAs in a way that overcomes their inefficient downstream processing is elongation of the carbon chains to form medium-chain fatty acids with higher monetary value (Leng et al. 2017). Such elongation can be achieved along different microbial pathways in anaerobic open-culture fermentation processes when reduced compounds are present (Spirito et al., 2014). The medium-chain fatty acids (MCFAs) thus obtained display much higher energy densities due to their lower oxygen-to-carbon ratio, and are therefore superior to VFAs as fuel-precursor chemicals (Steinbusch et al. 2011). Their increased hydrophobicity results in lower solubility, and thus in more energy and cost-efficient separation properties (Grootscholten et al. 2013). However, questions about how best to shape the microbiome and, if successful, how to construct a stable and resilient system suitable for industrial-scale application need further study. In addition, improved extraction technologies need to be developed, in particular to operate in line with the fermentation system (Spirito et al. 2014). Moreover, the metagenomics of impactful microbial cultures need to be analysed in order to further verify and define them (Leng et al. 2017).

Extracellular polymeric substances (EPS)

In recent years, the aerobic granular sludge (AGS) process – also known as the NEREDA process – has been applied successfully at several full-scale wastewater treatment plants around the world. AGS can be described as self-immobilised bacterial communities (Liu and Tay 2002). Its formation can be stimulated by discontinuous influent feeding (de Kreuk and van Loosdrecht 2004). EPS are responsible for the physical and chemical structure of the granules; they are bacteria-secreted sticky polymers consisting of proteins, polysaccharides, phospholipids, lipids and humic acids, which evoke cell adhesion and lead to the formation of aerobic granules. Extracting EPS from AGS is a potential future product recovery route that can yield a high-value product. In the

Netherlands, two full-scale demonstration systems for commercially viable and sustainable EPS recovery are currently planned (van der Roest et al. 2015).

A method using sodium carbonate (Na_2CO_3) and calcium ions (Ca^{2+}) extracts EPS from sludge in the form of stable ionic gel granules that, amongst other properties, behave in a similar way to alginate (Felz et al. 2016), even though they have a very different chemical composition. Recently, 'Kaumera' has been registered as a product name for EPS derived from AGS. However, alginate is conventionally produced from brown seaweed (Lee and Mooney 2012) and can form hydrogels that are biocompatible, non-toxic, non-immunogenic and biodegradable (Yang et al. 2011). Established alginate utilisations include pharmaceutical, food and technical applications, such as in printing paste for the textile industry (Draget 2009). It is likely that the alginate market is not the only potential niche for recovered EPS. Because their wide range of interesting material properties are still not fully understood, and also due to their novelty, it has yet to be demonstrated which conventionally produced niche polymers could be substituted by these materials and their composites. Tseggai (2016) indicates that the range of possible applications for EPS, both as a composite and as a raw material, is extensive. If alginate is to be substituted with wastewater derived EPS, however, that must be produced more cheaply than conventional alginate – not least because its current level of production, 30,000 thousand tonnes annually, is estimated to comprise only 10% of the alginate-like material potentially obtainable from wastewater. Which indicates a high unexploited potential for conventional production. This is especially valid if new chemical and biochemical techniques are developed to allow the creation of conventional but modified alginic-acid derivatives tailored for certain applications (Pawar and Edgar 2012).

Single-cell protein (SCP)

One well-documented product recovery technology is SCP synthesis. This process uses electrical energy from renewable energy surpluses to produce H_2 by electrolysis, to function as an electron donor for H_2 oxidising bacteria. In addition, ammonia stripped from sludge digestion liquids provides a third feedstock for the process. For the protein synthesis, minerals are also added to promote optimum growth of the biomass. As a result, ammonia-to-protein efficiencies of close to 100% can be achieved (Matassa et al. 2016). Used as feed for livestock, this protein could alleviate the pressure for land conversion since approximately 80% of agricultural land is used to grow fodder. If the protein obtained were used in food applications, though, consumer acceptance would be an issue (Matassa et al. 2015). Nevertheless, we believe that the inherent fear related to the use of products recovered from faecal matter could be overcome by education as well as the application of safe and effective technologies. Currently, the use of SCP produced from municipal wastewater is forbidden anyway by EU legislation (Alloul et al. 2018). If this technology were to be integrated into the CAS process, however, it would recover only the influent N ending up in the sludge (approximately 20% of the total) (Siegrist et al. 2008; Matassa et al. 2015). To harvest the solubilised ammonia in municipal wastewater as well, up-concentration techniques would have to be applied. Mehta et al. (2015) provide a detailed overview of emerging N-recovery technologies, which can be used for a more in-depth analysis of the topic.

Iron-phosphate

Significant iron (Fe) loads can enter a WWTP via Fe-rich industrial wastewater, groundwater infiltration and from Fe dosing of the sewerage system to prevent the emission of hydrogen sulphide (H_2S). Moreover, the addition of iron in the form of ferric (Fe^{III}) or ferrous (Fe^{II}) salts is the most common chemical P-removal (CPR) method used at WWTPs and can introduce significant iron-phosphate precipitates into their sludge lines. When CPR is applied, 40-50% of the total influent-P precipitate is in the form of vivianite ($Fe^{2+}Fe_2^{2+}(PO_4)_2 \cdot 8H_2O$) (Wilfert et al. 2016a). This is therefore likely to be the most abundant form of phosphate in digested sludge, and hence of particular interest when it comes to P recovery. However, the extraction of pure vivianite in crystal form still requires more knowledge about the factors determining its formation (Wilfert et al. 2018). Varying reaction conditions in different reactors (aerobic or anaerobic), amorphous and crystalline iron-phosphate molecule structures, the presence of humic substances and sulphates and varying oxidation-reduction potentials and pH values in different units at a plant make microbial and chemical-induced iron-phosphate reactions exceptionally diverse. In order to develop P-recovery pathways and possibly to control favoured iron-phosphate formations during the treatment process in the future, a better understanding of these mechanisms is needed (Wilfert et al., 2015). Nevertheless, an innovative pilot system using magnetic separation to recover vivianite from digested sewage sludge, called ViviMag, is currently under construction in the Netherlands (Wetsul.nl 2019; Prot et al. 2019).

Cellulose

Cellulose recovery from wastewater treatment processes has recently gained attention in scientific literature (Mussatto and van Loosdrecht 2016). Cellulose fibres in municipal wastewater originate mainly from toilet paper, which is a considerable fraction of the influent COD, and they are hardly degradable during aerobic treatment, especially under cold-weather conditions. And only 50% are anaerobically digested (Ruiken et al. 2013). Although cellulose recovery decreases biogas production by over 10%, cellulose extraction improves WWTP operations through lower aeration requirements and reduced excess sludge quantities, which may lead to an overall positive energy balance (van der Hoek et al. 2015). High recovery rates can be achieved by applying fine mesh sieves (<0.5mm) in the primary treatment line (Visser et al. 2016); these remove a significantly higher fraction of the cellulose fibres from the main line than do primary settling tanks (Ruiken 2010). Potential applications for recovered cellulose include soil conditioner, fuel for biomass combustion plants, feedstock for the fermentation industry (Ruiken et al. 2013), aggregate for construction materials such as asphalt and raw material for the paper pulp industry (Visser et al. 2016). Another interesting emerging application of cellulose is its refinement into nanocellulose, a nanocomposite with unique properties (Mussatto and van Loosdrecht 2016). The production of new toilet paper is also possible, but it is questionable whether consumers would accept this true cradle-to-cradle approach (Ruiken et al. 2013).

Summary: product recovery

Initial findings concerning some of the product recovery routes reviewed above show promising results in terms of quantities and market prizes (van Loosdrecht and Brdjanovic 2014). Since most of these routes utilise the organic carbon in wastewater, methane recovery from COD by integrating anaerobic digestion into the CAS process has been criticised for its high energy losses, leading to an overall energy efficiency of only about 15% (Frijns et al. 2013; Khiewwijit et al. 2016). The recovery of COD as organic materials rather than energy is seen as a promising alternative due to the much higher monetary value of organic chemicals (Puyol et al. 2017). Since COD-derived product recovery routes may exclude each other or require trade-offs, the value of the recovered products can also be an important criterion when deciding in favour of one specific route over an alternative. This is the case, for example, with the recovery of EPS and PHA (van der Hoek et al. 2016). As mentioned above, however, the consumer perspective and their association of wastewater-derived products with faecal matter is a severe barrier to several innovative recovery routes. Developing value chains for these products therefore poses new challenges for water management utilities, as they are often in non-consumer niche markets (Stanchev et al. 2017). To ensure that they marketable, their technological development must involve input from regulators, managers of wastewater facilities, engineers, researchers and the public (Li et al. 2015). The financial and operational risk of upscaling innovative product recovery routes should be shared among these stakeholders to build confidence in pioneering applications (NSF et al. 2015).

Bottlenecks in wastewater resource recovery

As discussed above and presented in Table 2, a variety of issues that may hinder the successful implementation of resource recovery routes are mentioned in the scientific literature. These relate to nine different bottlenecks, which can be grouped into three categories (A, B, C).

Economics and value chain (A)

1. Process costs.
2. Resource quantity.
3. Resource quality.
4. Market value and competition.
5. Utilisation and application.
6. Distribution and transport.

Environment and health (B)

7. Emissions and health risks.

Society and policy (C)

8. Acceptance.
9. Policy.

Most of the bottlenecks are in the economics and value-chain development category. This reflects with the findings of van der Hoek et al. (2016), who state that market potential and competition, in particular, introduce uncertainties in respect of successful resource recovery from wastewater. However, some of the bottlenecks presented in Table 2 overlap into other categories and so should be perceived as interlinked rather than absolute. Moreover, bottlenecks should not be interpreted merely as barriers to the implementation of resource recovery routes, but more as starting points for WWTP process design and management strategies to overcome them. Their early consideration in the planning phase of resource-oriented wastewater treatment processes increases the chance of developing successful recovery routes.

Category A. Economics and value chain				
Bottleneck	**Description**	**Resource**	**Issue**	**Reference**
Process costs	A resource recovery process is not cost-effective due to excessive operational or investment costs.	Water	High energy demand of membrane technologies.	Verstraete et al. 2009; Batstone et al. 2015.
			Fouling as an additional cost factor for membrane technologies.	Yangali Quintanilla 2010.
			Disposal costs of membrane retentate.	Eslamian 2016.
			Advanced oxidation processes are energy-intensive and require expensive reagents.	Agustina et al. 2005.
		Energy	Microbial fuel cells: expensive equipment and operation	Oh et al. 2010; Zhou et al. 2013; Li et al. 2013.
			NH_3 recovery for fuel is not cost-effective.	Gao et al. 2014.
		Nutrients	P recovery costs exceed conventional P ore costs (under German market conditions).	Cornel and Schaum 2009.
			Struvite recovery processes may not be cost-effective.	Le Corre et al. 2009.
			No cost-effective processes for recovering P from Fe-P have yet been developed.	Wilfert et al. 2015.
			P recovery from sludge incineration ash requires specialised and expensive incinerators.	Wilfert et al. 2018.
		Products	PHA recovery processes can be more costly than conventional production routes.	Fernández-Dacosta et al. 2015.
			CO_2 recovery from biogas is economically feasible only if a biogas upgrading unit is already present.	Hogendoorn et al. 2014.
			Bioelectrochemical systems may require expensive electrodes.	Villano et al. 2010; Logan and Rabaey 2012.

			Microbial electrolysis cells using CO_2 for chemical production require extra energy input.	Rabaey and Rozendal 2010.
Resource quantity	Compared with conventional production systems, only small quantities of a resource can be recovered at a WWTP. This may be due to low process yields, low resource concentrations or low overall resource quantities in a wastewater stream.	Energy	Combined heat and power units for recovered CH_4 have high conversion losses	Wan et al. 2016.
			COD may be too diluted for effective direct anaerobic digestion of wastewater,	Logan and Rabaey 2012; Frijns et al. 2013.
			Dark fermentation of sludge shows very low H_2 yields,	Lee et al. 2010.
		Nutrients	Nutrient quantities recoverable from wastewater are low compared with industrial production rates.	Kleerebezem et al. 2015.
			Struvite: low P concentrations limit precipitation.	Zhang et al. 2013; Xie et al. 2016.
			Struvite: only soluble P fraction is recovered.	Wilfert et al. 2015.
			Low N concentrations may make NH_4 recovery uneconomic.	Kuntke et al. 2012; Khiewwijit et al. 2016.
		Products	VFA concentration in wastewater and fermenter effluent is too low for economic extraction.	Rabaey and Rozendal 2010.
			Optimisation by economies of scale is limited due to low resource quantities in wastewater.	Kleerebezem et al. 2015.
Resource quality	The quality of a recovered resource is not high enough to market easily. This may be due to contaminants or impurities in the resource.	Nutrients	Field application of sewage sludge: high water and low nutrient content.	Kirchmann et al. 2017.
			Possible contamination of struvite.	Lin et al. 2013b; Xie et al. 2016.
		Products	Recovered biochemicals often lack the purity demanded by chemical industries.	Puyol et al. 2017.
			Controlling the product spectrum in open-culture VFA fermentation is a challenge.	Kleerebezem et al. 2015.
Market value and competition	Conventional production systems potentially outcompete the resource recovery route. This may be due to various factors, including higher product quality and quantities or lower	Energy	CH_4 has a low market value	Rabaey and Rozendal 2010; Kleerebezem et al. 2015.
			Electricity has a low market value.	Puyol et al. 2017.
		Nutrients	Bulk nutrients from the fertiliser industry are available cheaply.	Khiewwijit et al. 2016; Puyol et al. 2017.

	production costs.		P-rich manure is often abundantly available as alternative fertiliser	Coppens et al. 2016.
			The market value of struvite is hard to estimate.	Le Corre et al. 2009.
		Products	Petrol-based plastics may outcompete bioplastics.	Tamis and van Loosdrecht 2015; van der Hoek et al. 2016.
			Finding real advantages of recovered biochemicals over fuel or sugar-based alternatives.	Puyol et al. 2017.
Utilisation and applications	The usefulness of recovered resources might be unknown. New market niches, applications and partners have to be found to make a resource recovery route successful.	Products	Identifying niche markets (local or otherwise) and applications to increase market potential.	Kleerebezem et al. 2015.
			Developing public-private partnerships to market products can be a challenge	Stanchev et al. 2017.
			New PHA product utilisation routes have to be found.	Tamis and van Loosdrecht 2015.
Logistics	If recovered resources are not used on site, distribution and transport have to be organised. This may be challenging due to geographical and temporal discrepancies between supply and demand, lack of infrastructure or cost.	Water	Temporal and geographical discrepancies between supply of and demand for water must be considered.	Garcia and Pargament 2015.
			Topographical location of WWTP might require uphill pumping of reclaimed water.	McCarty et al. 2011.
			Possible need for new pipeline infrastructure for reclaimed water.	Yi et al. 2011; Wang et al. 2015b.
		Energy	Temporal and geographical discrepancies between supply of and demand for thermal energy need to be balanced out	Chae and Kang 2013; van der Hoek et al. 2016.
			Costs of pressurising and transporting CH_4 if no connection to the natural-gas grid is present.	Rabaey and Rozendal 2010.
		Nutrients	In-field sludge application: transport between WWTP and arable land might be too costly.	Kirchmann et al. 2017.
Category B. Environment and health				
Bottleneck	**Description**	**Resources**	**Issue**	**Reference**
Emissions and health risks	The use of recovered resources or the recovery process may entail risks to human health due to contaminants, or may cause emissions and environmental problems. This may be	Water	Potable water reuse has been evaluated as too great a health risk (by Amsterdam water board).	Rook et al. 2013; van der Hoek et al. 2016.
			Incomplete removal of chemicals or pathogens during treatment may cause disease.	Grant et al. 2012.
			Chemical biocides used in tertiary treatment can generate harmful by-products.	Zanetti et al. 2010.

	due to insufficient process control.		Plant or soil contamination as consequence of wastewater reuse for irrigation	Pedrero et al. 2010.
		Energy	Unheated anaerobic digesters may promote emissions of solubilised CH_4.	Frijns et al. 2013.
		Nutrients	Struvite may be contaminated with emerging pollutants and heavy metals.	Lin et al. 2013b; Xie et al. 2016.
			PAO biomass may accumulate contaminants if sludge is applied to agricultural land.	Sheik et al. 2014.
Category C. Society and policy				
Bottleneck	**Description**	**Resources**	**Issue**	**Reference**
Acceptance	User acceptance of resources recovered from wastewater may be low due to fears or misconceptions about the risks they pose.	Water	Water reuse projects can rarely be implemented without social acceptance.	Bdour et al. 2009; Garcia and Pargament 2015.
			Direct potable water reuse raises psychological barriers.	Verstraete and Vlaeminck 2011.
		Products	Toilet-paper production from recovered cellulose may not be accepted by consumers	Ruiken et al. 2013.
			Single-cell protein: negative perception of faecal matter as source for feed/food production.	Matassa et al. 2016.
Policy	Resource recovery routes need adequate policy and legal frameworks to be successful. A lack of legislation, political will or economic incentives may hinder successful implementation.	Water	Government incentives are needed to make water reuse financially attractive (in China).	Yi et al. 2011.
			A lack of common regulations is a barrier to water reuse (in southern Europe).	Lavrnić et al. 2017.
			Lack of political will to put legislation and policies for water reuse into practice.	Guest et al. 2009.
		Energy	Anaerobic digestion needs to be subsidised to become competitive with natural gas.	Kleerebezem et al. 2015.
		Nutrients	Lack of legislation for in-field struvite application.	van der Hoek et al. 2016.
		Products	Legislation forbids the use of protein produced from faecal substrate (in Europe).	Alloul et al. 2018.

Table 2. Detailed overview of bottlenecks mentioned in scientific literature, which may hinder the successful implementation of resource recovery routes (RRRs) at municipal WWTPs.

The influence of water management utilities

Water management utilities (WMUs) could possibly influence or even overcome the listed bottlenecks to successful RRR implementation through pro-active planning of resource recovery routes. However, their power to tackle certain bottlenecks may be limited because these are at different influence levels (Figure 6). To reduce process costs, recover safe and environmentally benign products or to ensure that quality requirements for

recovered resources are met, the right decisions need to be made at the process-design level. Here, the WMU may have significant influence over the design of a process that meets all these requirements, because it traditionally possesses substantial expertise in process engineering and operations. To overcome bottlenecks related to the distribution and transport of recovered resources, as well as to find applications and utilisation possibilities, requires management actions beyond the scope of technical process design but still within the WMU's sphere of influence, if it makes the right management decisions. Similarly, the recovery of resources in competitive quantities can be managed actively. The volumes recovered might be limited by factors related to the technical process, such as process yields, or by the fact that the wastewater stream contains only small quantities of a resource, but once this is recognised it may still be possible, through management action, to increase output of a resource by integrating other waste streams into the recovery process (Lee et al. 2014). If, for example, VFAs are recovered from COD, the integration of solid organic waste to obtain higher product volumes may strengthen the WMU's market power as a VFA supplier. Joining forces with other WMUs to recover and market a resource collectively is another possible management-driven strategy to increase output.
However, the successful implementation of RRRs also depends on factors more difficult for a WMU to influence. These are related to the broader circumstances in which an RRR operates. Examples include relevant policy and legislative frameworks, market values and the competitive situation, as well as user acceptance of a particular recovered resource. Although it is more difficult to leverage positive change at this level, the WMU can still develop strategies to convince policymakers or users about the necessity or harmlessness of an RRR. In general, greater competitiveness can be achieved by finding niche markets or by forming strategic partnerships with stakeholders within the value chain to develop a common approach, making the most of synergies (Stanchev et al. 2017). In addition, co-operation between WMUs – for example, joining forces to apply a common recovery strategy across multiple WWTPs and so exploit economies of scale – could well enhance economic competitiveness.
WMUs may also need to find ways to gain support in scaling up innovative resource recovery technologies. The implementation of new practices requires access to reliable data in order to build confidence that the innovation is compatible with the current process. There is currently little benefit for a WMU in being a pioneer in resource recovery, so these utilities should seek therefore support from value-chain actors or political institutions to share the risks of innovation implementation (NSF et al. 2015).

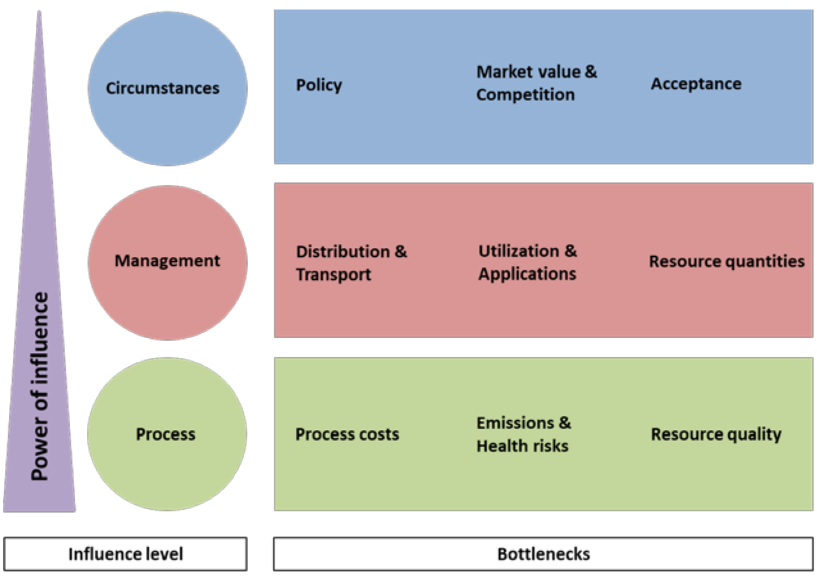

Figure 6: The power or influence of water management utilities (WMUs) on the identified bottlenecks.

Conclusion

Although domestic wastewater cannot fully satisfy the elemental or energy demands of industrialised societies, it does represent a substantial resource that should be fully utilised in the future. However, the data presented in Table 1 shows that not all RRRs can meet substantial shares of overall resource demands. The market potentials of recovered water, energy, nutrients and products depend on the volumes demanded, the quantities contained in wastewater and the recovery yields obtainable. Before future treatment processes are designed from a circular-economy perspective, it is useful to be aware of the likely ability of the proposed RRRs to satisfy overall demand for relevant resources and, on that basis, to invest primarily in those with the potential to diminish conventional resource exploitation most substantially. RRRs that contribute significantly to meeting overall societal resource needs are likely to attract more interest from public funding bodies or policy incentive schemes than those with lesser potential in this respect.

Although numerous technologies for the recovery of water, energy, nutrients and products from wastewater have been explored in the academic arena, few of these have ever been applied on large scale due to technical immaturity and/or non-technical bottlenecks. In all, we have identified nine such bottlenecks mentioned in scientific literature, which may hinder the successful integration of resource recovery routes into WWTPs (Table 2). Six of these are related to economics and value-chain development (process costs, resource quantities, resource quality, market value, application and distribution), two to environmental (emissions) and health (contamination) risks and another two to social (acceptance) and policy issues. It is unlikely that WMUs can influence the resolution of all these bottleneck to an equal extent. We hypothesise that those related to issues other than the technical process itself are currently difficult for WMUs to solve. This is due to their rather narrow management focus on wastewater treatment rather than resource recovery. Implementing RRRs successfully will require WMUs to extend their engineering expertise and to become market participants actively engaged with all aspects relevant to the creation of value chains for recovered resources, without losing sight of their primary focus on treating wastewater to legal effluent standards.

To implement future water-resource recovery facilities (WRRFs) that recover multiple resources, WMUs need to perceive themselves as market actors producing goods rather than as utilities managing a fixed budget for cost-effective treatment-plant operations. The challenge is to leave behind the paradigm of merely operating existing WWTPs and instead to start perceiving wastewater as a resource that requires management at different levels and investments in research and development in order to reintroduce resources successfully into markets for societal consumption. Value can be created if the interests of all stakeholders, including business partners, end users and policy makers, are integrated into the planning process. If a WMU plans the implementation of a technically feasible resource recovery technology, it is recommendable that it analyse in advance whether any of the non-technical bottlenecks presented in this review still need to be tackled. In the future, WMUs could co-operate to develop a common recovery strategy that co-ordinates efforts to exploit synergies and the advantages of economies of scale. If several recover the same resource, value-chain development could be facilitated by acting as one supplier and so increasing their collective market power . This idea has already been put into practice in the Netherlands, where water boards have set up the so-called 'Energy and Raw Materials Factory (Energie en Grondstoffen Fabriek) to act as a collaborative network organisation co-ordinating recovery efforts by several WMUs.

The most precious resource contained in municipal wastewater is water. Unlike energy, after all, which can be obtained from multiple sources, it has is no alternative origin. Wastewater reuse can provide an important alternative source of fresh water in regions that expect lasting shortages in the future. Preferably, it should also be promoted where it is less energy and resource-demanding than conventional fresh-water treatment and distribution. In the future, it is possible that stricter effluent-quality regulations will require the elimination of emerging pollutants. For this reason, advanced energy-intensive treatment steps could become necessary anyway (Høibye et al. 2008). The resulting higher effluent quality would also increase water-reuse opportunities.

Anaerobic digestion as a bioenergy production system will only become economically viable if subsidies are available to ensure its competitiveness with commercial natural-gas supplies (Kleerebezem et al., 2015). This counteracts the development of potentially more sustainable solutions, like the recovery of COD as biomaterials. In addition to the recovery of chemical energy stored in the COD, municipal WWTP effluents contain thermal energy that could provide ten times more heat than the CH_4-CHP route and should therefore be considered more prominently in wastewater resource recovery planning.

Nutrient-recovery technologies should aim for the capture of most nutrients. For P recovery, that could mean that it is beneficial to place the recovery unit at the end of the treatment process, as is already the case with sludge incineration ash. In livestock-intensive regions, however, P recovery strategies should focus on manure before municipal wastewater due to the recoverable quantities as shown in Figure 4. Ammonium recovery is only recommendable if the process consumes less energy than conventional ammonium production. Informing and

educating the public, and involving it in the process of planning future resource-oriented WWTPs, can help to increase acceptance of resources recovered from wastewater.
The supply potentials and bottlenecks presented in this paper should be perceived as challenges rather than as obstacles. We believe that successfully implementing wastewater resource recovery requires pro-active management of potential bottlenecks and sharing the risks associated with being a pioneer. To achieve the transition from WWTPs to WRRFs, resource recovery needs to be considered a strategic goal from the earliest process design and planning stages. Implementing a WRRF requires decisions in fields far beyond the traditional responsibilities of WMUs. The scientific community should therefore elaborate the insights into process integration and the decision-support tools needed to help WMUs strategically plan and design WRRFs to exploit their vast technological potential and to overcome non-technological bottlenecks.

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