Towards water resource factories

Designing and planning sustainable circular wastewater treatment processes

Kehrein, P.A.

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Towards Water Resource Factories

Philipp Kehrein

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Designing and planning sustainable circular wastewater treatment processes

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Designing and planning sustainable circular wastewater treatment processes

Dissertation

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by

Philipp Alexander KEHREIN
M.Sc. in Environmental Science - Soil Water, Biodiversity (EnvEuro double-degree),
University of Hohenheim, Germany & University of Copenhagen, Denmark
born in Wuppertal, Germany

This dissertation has been approved by the promotor.

Composition of the doctoral committee:

Rector Magnificus,	Chairperson	
Prof. Dr. P. Osseweijer	Delft University of Technology	Promotor
Prof. Dr. Ir. J. Dewulf	Ghent University	Promotor
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Reserve member:

Prof. Dr. Ir. M.C.M van Loosdrecht	Delft University of Technology

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List of abbreviations

AC	Activated carbon
AD	Anaerobic digestion
AGS	Aerobic granular sludge
AHP	Analytical hierarchy process
AnMBR	Anaerobic membrane bioreactor
AOP	Advanced oxidation process
bCOD	Biodegradable chemical oxygen demand
BES	Bioelectrochemical systems
BOD	Biological oxygen demand
Capex	Capital expenditures
CAS	Conventional activated sludge
CBA	Cost and benefit analysis
CEB	Chemically enhanced backwash
CEC	Contaminants of emerging concern
CEPT	Chemically enhanced primary treatment
CFC	Chlorofluorocarbon
CHP	Combined heat and power
CIP	Chemical cleaning in place
COD	Chemical oxygen demand
CPR	Chemical phosphorous removal
ct	Euro cent
Demi water	Demineralized water
DO	Dissolved oxygen
DS	Dry solids
EBPR	Enhanced biological phosphorous removal
EPS	Extracellular polymeric substances
GAC	Granular activated carbon
GHG	Greenhouse gas
GWP	Global warming potential
H_2O_2	Hydrogen peroxide
HL-MBR	High loaded membrane bioreactor
HM	Heavy metal
HRT	Hydraulic retention time
IEX	Ion exchange mixed bed
LHV	Lower heating value
LMH	Membrane flux in litre per m2 per hour
LRV	Log removal value
MATP	Membrane-based advanced treatment process
MBR	Membrane bioreactor
MEB	Mass and energy balance
MEC	Microbial electrolysis cell
MF	Microfiltration
MFC	Microbial fuel cell
MP	Micro-pollutant
MW	Megawatt
N	Nitrogen
NPV	Net present value
Opex	Operational expenditures
P	Phosphorous
PAC	Powder activated carbon
PCP	Polychlorinated biphenyl
PHA	Polyhydroxyalkanoates
PJ	Petajoule
PPP	Public-private partnership
PV	Photovoltaic
Remin.	Remineralisation
RO	Reverse osmosis
RRR	Resource recovery route
SAC	Strongly acidic cation exchange resins
SBA	Strongly basic anion exchange resins
SCP	Single cell protein
SDG	Sustainable development goal
SI	Supplementary information
SRT	Solids retention time
STOWA	Dutch Foundation for Applied Water Research
TJ	Terajoule
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
TSS	Total suspended solids
UF	Ultrafiltration
UV	Ultraviolet light
VFA	Volatile fatty acids
WAVE	Water Application Value Engine software
WMU	Water management utility
WRF	Water resource factory
WRRF	Water resource recovery facility
WWTP	Wastewater treatment plant

Summary

In the current linear take-make-waste pattern the production of goods starts with raw material extraction followed by industrial conversion into products that are used and finally wasted. This linear system accelerates resource depletion and therefore hinders the development of sustainable societies. This is also valid for the use of water and the implied production of wastewater. The ongoing rapid urbanisation in many areas of the world including Europe has led to high increases in wastewater since the beginning of the 20th century. The initial goal of wastewater treatment was to protect surface water users from health risks due to pollution. Then, during the last decades the protection of the environment itself from nutrient pollution has been enforced by implementing stricter legal wastewater treatment plant (WWTP) effluent standards. The conventional activated sludge process is the currently most widely applied wastewater treatment technology in these plants. It succeeds in reaching legal standards for chemical oxygen demand (COD), nitrogen, and phosphorous effluent concentrations but in its currently applied form, it is considered unsustainable due to its low resource recovery potential and cost-effectiveness on the one hand and its high energy demand and environmental footprint on the other. To adapt wastewater treatment practices to urgent requirements for more sustainable urban environments, a paradigm shift has been proposed by academia since over a decade. It recognises the potential of wastewater as a resource and demands to perceive it as such instead of a waste stream.

Numerous technologies have been developed since then to recover water, energy, fertilizers and other products from municipal wastewater. By reusing resources contained in municipal wastewater, we could tackle water scarcity problems, lower fossil energy consumption and address global nutrient needs. In addition, it should not be forgotten that a variety of other products (e.g. biopolymers) can be recovered from wastewater. Unfortunately the implementation of innovative recovery technologies into treatment processes does not match with their rapid development which leads to an increasing number of readily available technologies that are waiting for implementation. To achieve that, a solid analysis prior to the implementation of recovery technologies into treatment processes is required to understand how to optimally integrate and design processes form a circular economy perspective. The focus of this dissertation is to solve uncertainties about which of the innovative technologies are most effective and how to optimally combine them to design and successfully implement water resource factories instead of wastewater treatment plants in the future. A water resource factory is a process that not only treats wastewater in a robust way to meet legal effluent qualities but also recovers various marketable resources, is technically feasible, cost efficient, and shows low environmental impacts. To contribute to this ongoing transition this dissertation investigates not only the technical design space of water resource factories but also the non-technical bottlenecks that may hinder their successful implementation. Consequently, a holistic and multidisciplinary research approach is followed to consider the various bottlenecks to be solved to innovate wastewater treatment and integrate it into a circular economy. The dissertation gradually evolves from understanding the potential of innovative resource recovery technologies towards exploring process design and system integration possibilities and finally providing guidance for decision makers to holistically plan and practically implement water resource factories.

Chapter 1 gives a general introduction about the research field of wastewater resource recovery and puts it into context to the circular economy concept. Also the outline of this dissertation and the research questions it aims to answer are presented.

The critical literature review in **chapter 2** aims to inform decision-makers in water management utilities and elsewhere about (i) the vast technical possibilities, (ii) the market supply potentials, and (iii) the diverse bottlenecks, related to the successful recovery of various resources form municipal wastewater treatment plants. 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 11 resources to supply national demands. Depending on the resource, the supply potential may vary

greatly. Secondly, resource recovery technologies developed in academia are critically reviewed regarding their technological readiness and shortcomings. The third section of the review identifies and classifies nine non-technical bottlenecks that have to be overcome to successfully implement these technologies into wastewater treatment processes. The bottlenecks are related to economics and value chain development, environment and health, and society and policy issues.

In **chapter 3** the mass and energy flows of an existing aerobic granular sludge treatment plant that currently recovers no resources are modelled. Mass and energy balances are an excellent method to model resource recovery potentials of innovative process designs at an early design stage because they allow to quantify recoverable resources as well as trade-offs between possible recovery technology choices. In total, five different process designs that would recover chemical oxygen demand (COD) and phosphorous (P) are investigated. The integration of anaerobic sludge digestion for subsequent electricity and heat generation from methane is modelled including chemically enhanced primary treatment for maximised energy recovery. Moreover, COD recovery as biopolymers is modelled for different process designs to reveal the trade-offs of COD recovery as energy and as biopolymers. In addition, the recovery of struvite is compared to phosphorous recovery from sludge incineration ashes. The trade-offs between P recovery as fertilizer and as biopolymers are also quantified. It is revealed how the integration of different recovery technologies may limit or complement each other. The study helps therefore to understand how aerobic granular sludge based treatment processes can be designed as water resource factories in the future.

Chapter 4 aims to contribute to the question of how municipal wastewater can alleviate water scarcity which is the geographic and temporal mismatch between freshwater demand and availability. Membrane-based advanced treatment processes are applied worldwide as the centre technology for wastewater reuse because they are scalable and provide a physical barrier that retains pollutants of almost all kind. Membrane-based processes can be designed to specifically meet distinct water qualities demanded by industrial, domestic or agricultural consumers. To decide which one of these three possible reuse types is preferable from a reclamation process perspective, process recovery rates, energy consumption and net costs need to be compared. Therefore, mass and energy balances are conducted to estimate the recovery rates and energy requirements of advanced treatment processes combining different process modelling tools and assuming Dutch principles. Then cost and benefit analyses are conducted to reveal the economic performance of the studied process designs under Dutch market conditions. Furthermore, the pros and cons of designing a "fit for multi-purpose" process are discussed. Finally, a process optimization concept that would increase the performance of reverse osmosis processes is shown and renewable energy integration possibilities are presented.

Chapter 5 converts the insights gained in the previous chapters into a novel framework for conceptually designing and strategically planning water resource factories. A multidimensional and multidisciplinary approach is followed to design wastewater treatment plants from a resource recovery perspective which leads to true water resource factory design. The designed process needs to be technically feasible, cost effective, show low environmental impacts, and recover various resources that can be successfully marketed. To achieve that, the traditional WWTP design space is opened up for a variety of other expertise that complement the traditional wastewater engineering domain. The framework combines insights and methodologies from different fields and disciplines, like e.g. circular economy, industrial process engineering, project management, value chain development, and environmental impact assessment. It structures possible resource recovery activities according to multiple assessable criteria that also allow to assess the marketability and value chain development potential of recoverable resources.

Chapter 6 concludes the main findings of this dissertation and provides an outlook for future actions needed within the wastewater domain to successfully plan, design and implement water resource factories as an integral part of a circular economy.

Samenvatting

In het huidige lineaire "take-make-waste" patroon begint de productie van goederen met de winning van grondstoffen, gevolgd door industriële omzetting in producten die worden gebruikt en uiteindelijk worden verspild. Dit lineaire systeem versnelt de uitputting van grondstoffen en belemmert daarom de ontwikkeling van duurzame samenlevingen. Dit geldt ook voor het gebruik van water en de impliciete productie van afvalwater. De aanhoudende snelle verstedelijking in veel delen van de wereld, waaronder Europa, heeft sinds het begin van de 20e eeuw geleid tot een sterke toename van het afvalwater. Het oorspronkelijke doel van afvalwaterzuivering was om gebruikers te beschermen tegen gezondheidsrisico's als gevolg van vervuiling van oppervlaktewater . Vervolgens is in de afgelopen decennia de bescherming van het milieu tegen nutriëntenverontreiniging zelf afgedwongen door de invoering van strengere wettelijke normen voor afvalwaterzuiveringsinstallaties (AWZI). Het conventionele actiefslibproces is momenteel de meest toegepaste afvalwaterbehandelingstechnologie in deze installaties. Het slaagt erin de wettelijke normen te halen voor de concentraties van chemisch zuurstofverbruik (CZV), stikstof en fosforafvalwater, maar in zijn huidige vorm wordt het als onhoudbaar beschouwd vanwege het lage potentieel voor het terugwinnen van hulpbronnen en de kosteneffectiviteit enerzijds en het hoge energieverbruik, en de vraag en ecologische voetafdruk anderzijds. Om afvalwaterzuiveringspraktijken aan te passen aan de dringende eisen voor duurzamere stedelijke omgevingen, stelt de academische wereld al meer dan een decennium een paradigmaverschuiving voor. Het heeft het potentieel van afvalwater erkent als een hulpbron en vraagt daarom om het als zodanig te zien in plaats van als een afvalstroom.

Sindsdien zijn er tal van technologieën ontwikkeld om water, energie, meststoffen en andere producten uit gemeentelijk afvalwater terug te winnen. Door grondstoffen aanwezig in gemeentelijk afvalwater te hergebruiken kunnen we de problemen met waterschaarste aanpakken, het verbruik van fossiele energie verminderen en de wereldwijde behoefte aan voedingsstoffen aanpakken. Bovendien mag niet worden vergeten dat uit het afvalwater een verscheidenheid aan andere producten kan worden teruggewonnen. Helaas komt de implementatie van innovatieve terugwinningstechnologieën in behandelingsprocessen niet overeen met hun snelle ontwikkeling, wat leidt tot een toenemend aantal gebruiksklare technologieën die wachten op implementatie. Om dat wel tot stand te brengen is een gedegen analyse vereist voorafgaand aan de implementatie van hersteltechnologieën in behandelingsprocessen om te begrijpen hoe processen optimaal kunnen worden geïntegreerd en ontworpen vanuit het perspectief van de circulaire economie. De focus van dit proefschrift is onzekerheden oplossen over welke van de innovatieve technologieën het meest effectief zijn en hoe deze optimaal kunnen worden gecombineerd om in de toekomst watergrondstoffabrieken te ontwerpen en met succes te implementeren in plaats van afvalwaterzuiveringsinstallaties. Om bij te dragen aan deze voortdurende transitie, onderzoekt dit proefschrift niet alleen de technische ontwerpruimte van watergrondstoffabrieken, maar ook de niet-technische knelpunten die hun succesvolle implementatie kunnen belemmeren. Als gevolg daarvan wordt een holistische en multidisciplinaire onderzoek benadering gevolgd om de verschillende knelpunten te overwegen die moeten worden opgelost om afvalwaterzuivering te innoveren en te integreren in een circulaire economie. Het proefschrift ontwikkeld zich geleidelijk van het begrijpen van het potentieel van innovatieve technologieën voor het terugwinnen van hulpbronnen naar het verkennen van procesontwerp en systeemintegratiemogelijkheden, en uiteindelijk het bieden van begeleiding aan besluitvormers om holistisch te plannen en praktisch te implementeren voor watergrondstoffabrieken.

Hoofdstuk 1 geeft een algemene inleiding over het onderzoeksveld van de terugwinning van grondstoffen uit afvalwater en plaatst dit in de context van het concept van de circulaire economie. Ook worden de opzet van dit proefschrift en de onderzoeksvragen die het beoogt te beantwoorden gepresenteerd.

De kritische beoordeling in **hoofdstuk 2** heeft tot doel besluitvormers in waterbeheerbedrijven en elders te informeren over (i) de enorme technische mogelijkheden, (ii) het marktaanbodpotentieel, en

(iii) de diverse knelpunten, gerelateerd aan het succesvolle terugwinning van verschillende grondstoffen uit gemeentelijke afvalwaterzuiveringsinstallaties. Informatie en data zijn uit literatuur onttrokken om een holistisch overzicht te geven van dit groeiende onderzoeksveld. Ten eerste wordt deze data gebruikt om het potentieel van 11 middelen te berekenen om aan de nationale vraag te voldoen. Afhankelijk van de grondstof kan het leveringspotentieel sterk variëren. Ten tweede worden technologieën voor het terugwinnen van grondstoffen die in de academische wereld zijn ontwikkeld kritisch beoordeeld op hun technologische paraatheid en tekortkomingen. Het derde deel van de review identificeert negen niet-technische knelpunten die moeten worden overwonnen om deze technologieën met succes in afvalwaterzuiveringsprocessen te implementeren. De knelpunten zijn gerelateerd aan economie en waardeketenontwikkeling, milieu en gezondheid, en maatschappelijke en beleidsvraagstukken.

In **hoofdstuk 3** worden de massa- en energiestromen van een bestaande aërobe korrelslibbehandelingsinstallatie gemodelleerd die momenteel geen grondstoffen terugwint. Massa- en energiebalansen zijn een uitstekende methode om het potentieel voor het terugwinnen van grondstoffen van innovatieve procesontwerpen in een vroege ontwerpfase te modelleren, omdat ze het mogelijk maken om herwinbare grondstoffen te kwantificeren, en om afwegingen te maken tussen mogelijke keuzes voor terugwinningstechnologie. In totaal worden vijf verschillende procesontwerpen onderzocht die de chemische zuurstofbehoefte (CZV) en fosfor (P) terugwinnen. De integratie van anaërobe slibgisting voor de daaropvolgende opwekking van elektriciteit en warmte uit methaan is gemodelleerd inclusief chemisch verbeterde primaire behandeling voor maximale energieterugwinning. Verder wordt CZV-terugwinning als biopolymeren gemodelleerd voor verschillende procesontwerpen om de wisselwerking met CZV-terugwinning als energie aan het licht te brengen. Daarnaast wordt de terugwinning van struviet vergeleken met de terugwinning van fosfor uit slibverbrandingsassen. Ook de afwegingen tussen P-terugwinning als meststof en als biopolymeer worden gekwantificeerd. Er wordt onthuld hoe de integratie van verschillende terugwinningstechnologieën elkaar kunnen beperken of aanvullen. De studie helpt daarom te begrijpen hoe aëroob korrelslib-gebaseerde behandelingsprocessen in de toekomst kunnen worden ontworpen als fabrieken voor grondstoffen terug te winnen uit water.

Hoofdstuk 4 beoogt bij te dragen aan de vraag hoe gemeentelijk afvalwater waterschaarste kan verminderen, wat de geografische en temporele mismatch is tussen de vraag naar zoet water en de beschikbaarheid. Geavanceerde behandelingsprocessen op basis van membranen worden wereldwijd toegepast als de centrumtechnologie voor hergebruik van afvalwater omdat ze schaalbaar zijn en een fysieke barrière vormen die vervuilende stoffen van bijna alle soorten tegenhoudt. Processen gebaseerd op membranen kunnen worden ontworpen om specifiek te voldoen aan de verschillende waterkwaliteiten die vereist zijn voor alle soorten waterhergebruik die worden gevraagd door industriële, huishoudelijke of agrarische gebruikers. Om te beslissen welke van deze drie mogelijke soorten hergebruik de voorkeur verdient vanuit het perspectief van het terugwinningsproces, moeten de terugwinningspercentages, het energieverbruik en de netto kosten van het proces worden vergeleken. Massa- en energiebalansen worden uitgevoerd om de terugwinningspercentages en energiebehoeften van geavanceerde behandelingsprocessen in te schatten, waarbij verschillende procesmodelleringstools worden gecombineerd en uitgaande van Nederlandse principes. Vervolgens worden kosten- en batenanalyses uitgevoerd om de verschillende economische prestaties van de bestudeerde procesontwerpen onder Nederlandse marktomstandigheden zichtbaar te maken. Verder worden de voor- en nadelen van het ontwerpen van een "fit for multi-purpose" proces besproken. Ten slotte wordt een procesoptimalisatieconcept getoond dat de prestaties van omgekeerde osmoseprocessen zou verbeteren en worden mogelijkheden voor integratie van hernieuwbare energie gepresenteerd.

Hoofdstuk 5 zet de inzichten die in de voorgaande hoofdstukken zijn opgedaan om in een nieuw raamwerk voor het conceptueel ontwerpen en strategisch plannen van watergrondstoffabrieken. Een multidimensionale en multidisciplinaire benadering wordt gevolgd om afvalwaterzuiveringsinstallaties te ontwerpen vanuit het perspectief van terugwinning van grondstoffen, wat zou leiden tot daadwerkelijke watergrondstoffabrieken. Die moeten technisch haalbaar en kosteneffectief zijn, een lage milieu-impact hebben en verschillende grondstoffen terugwinnen die met succes op de markt kunnen worden gebracht. De traditionele AWZI-ontwerpruimte wordt opengesteld voor een verscheidenheid aan andere expertises die het traditionele domein van afvalwatertechniek aanvult. Het raamwerk combineert inzichten en methodologieën uit verschillende velden en disciplines, zoals b.v. circulaire economie, industriële procestechniek, projectmanagement, ontwikkeling van de waardeketen en milieueffectrapportage. Het structureert mogelijke activiteiten voor het terugwinnen van hulpbronnen volgens meerdere beoordeelbare criteria die het ook mogelijk maken om de verhandelbaarheid en het ontwikkelingspotentieel van de waardeketen van terugwinbare hulpbronnen te beoordelen.

Hoofdstuk 6 omvat de belangrijkste bevindingen van dit proefschrift en biedt een vooruitzicht voor toekomstige acties die nodig zijn binnen het afvalwaterdomein om met succes watervoorgrondstoffabrieken te plannen, ontwerpen en implementeren als integraal onderdeel van een circulaire economie.

1

General Introduction

“Until you dig a hole, you plant a tree, you water it and make it survive, you haven't done a thing. You are just talking”.

Wangari Maathai

1.1. Introduction and motivation

The quote above is from the famous Kenyan politician and environmental and human rights activist Wangari Maathai who won the Nobel Peace Prize in 2004. I comprehend it as a reminder that scientific work is a crucial but theoretical exercise that cannot substitute action towards real change. In that sense, although this dissertation is a bundle of words I hope it lays the necessary foundation to make right decisions during the implementation of water resource factories in the future. The cover of this book shows the painting "Waterfall" by the Dutch artist M.C. Escher. I chose it not only because Escher lived in Den Haag, like myself during my PhD, but because he paints an eternal water cycle in an urban setting that also includes energy recovery by a water wheel. In this dissertation I want to show that the creation of circular water systems is not only possible by using optical illusions, like Escher did but that they can be strategically planned and engineered.

The term "sustainable development" was first announced by the World Commission on Environment and Development in 1987. It describes a development that is capable to cover today's needs for an intact environment, social justice and economic prosperity without limiting those needs for future generations (Finkbeiner et al. 2010). The European Commission has followed this concept politically by manifesting sustainable development as a major strategic goal. The transition to a more circular economy, where the value of products, materials and resources is maintained in the economy for as long as possible and the generation of waste is minimised, is seen as an essential necessity to develop a sustainable, low carbon, resource efficient and competitive economy (European Commission 2015a). Since the industrial revolution, the production of goods starts with raw material extraction followed by industrial conversion into products that are wasted after their use. This linear "take, make, waste" pattern is the root cause of the current unsustainable resource consumption (Daigger 2008). It accelerates resource depletion and therefore restricts the development of sustainable economies (The Worldwatch Institute 2008).

Worldwide, cities play a key role in achieving global sustainability targets. In the immediate future, millions of people will move from rural areas into cities to seek for better economic opportunities. Considering this ongoing urbanization, sustainable urban development becomes an important pillar for a more sustainable future in general (Shen et al. 2011). Since resource flows are especially directed towards urban areas, especially cities face the urgency for implementing strategies that lead to a more circular management of consumed resources (Agudelo-Vera et al. 2012). If resource recovery systems can be implemented, the concentration of resources in cities provides great chances to achieve a more sustainable resource use in the future. For example, close to 100% of phosphorous eaten in food is excreted and therefore urban wastewater flows have been labelled as phosphorous "hotspots" (Cordell et al. 2009).

It is clear that resource demands will continue to grow on a global scale due to a steady population growth and a global market economy model that assumes perpetual growth. This means that many resources will get rapidly more limiting if a circular economy will not be achieved. Regarding water, the global demand is expected to grow about 50% by 2050 compared to today which will imply large increases in urban water demand and wastewater generation (WWAP 2017). In this context, wastewater management plays globally a central role for sustainable urban development (Corcoran et al. 2010). In addition to water, municipal wastewater contains the detritus of our daily lives, like feces, fat, food scraps, detergents and pharmaceuticals that are flushed out of households (Li et al. 2015). In addition, it contains wastewater from commercial establishments and industries, as well as storm water and surface runoff. It may contain high concentrations of organic and inorganic pollutants, pathogenic microorganisms, as well as toxic chemicals (Riffat 2013).

In most western countries, municipal wastewater is conveyed by sewer systems to centralized wastewater treatment plants (WWTPs) that are operated by public, private, or public-private water management utilities responsible for meeting legal effluent qualities and managing treatment costs effectively. The current practices in wastewater treatment are based on rather outdated engineering solutions established in the early 20th century and need innovation to meet future challenges (Daigger

2009). Initially, the goal of wastewater treatment was to protect downstream users from health risks. Then in the last decades, stringent legal effluent standards were introduced to prevent nutrient pollution in surface waters. As a consequence, wastewater treatment plants started to implement nutrient removal technologies (Verstraete et al. 2009). The currently most widely used wastewater treatment technology is the conventional activated sludge process (CAS) in which aerobic microorganisms metabolise the organic fraction of the wastewater under constant oxygen supply (Oh et al. 2010). Although the CAS process succeeds in reaching legal effluent requirements, it is considered unsustainable due to its low resource recovery potential and cost-effectiveness on the one hand, and its high energy demand and environmental footprint on the other hand (Verstraete and Vlaeminck 2011).

The urge for a circular economy and the resource inefficiency of current wastewater treatment practices drives a paradigm shift on wastewater treatment from pollutant removal towards recovery of water, energy, fertilizer, and other products. Consequently, wastewater has been recognised in the scientific discussion as a resource rather than a waste stream since already a decade (Guest et al. 2009; Ma et al. 2013; van Loosdrecht and Brdjanovic 2014). Today, several specifications exist within the wider research field of wastewater resource recovery. They address in depth the potential of a particular resource, a certain end-product, or a single recovery technology. Examples include the specifications on water reclamation processes for different reuse types (Eslamian 2016), biological resource recovery pathways (Puyol et al. 2017), energy and product recovery pathways from waste sludge (Tyagi and Lo 2016), or different phosphorous recovery pathways from domestic wastewater (Le Corre et al. 2009; Rittmann et al. 2011). The online search engine "Google Scholar" finds over 145 thousand scientific articles for the term "municipal wastewater resource recovery technology", excluding citations and patents. It can therefore be argued that enough technological innovation has been developed already to implement more sustainable wastewater treatment processes which are based on the circular economy principle.

Since a key characteristic of the growing scientific resource recovery field is a broad range of technical options (Batstone et al. 2015), the question arises on which of them focus should become laid in WWTP process design. Uncertainties about which techniques are most effective and how to combine innovative technologies stands in the way of creating "water resource factories" that finally replace the old wastewater treatment plant concept (Li et al. 2014). Therefore the general research question that this dissertation aims to answer is, how can innovative resource recovery technologies become effectively and efficiently implemented into municipal wastewater treatment processes to design water resource factories in the future? A water resource factory is a process that not only treats wastewater in a robust way to meet legal effluent qualities but also recovers various marketable resources, is technically feasible, cost efficient, and shows low environmental impacts. Although the urgencies mentioned above push natural resource use towards limits beyond sustainability and hence make the development of circular urban wastewater management practices inevitable, changing the current wastewater handling system seems difficult because it implies distractions (Daigger 2009). It is clear that the effective treatment of wastewater for safe and environmentally friendly discharge has to remain also within the water resource factory concept the primary objective in process design and planning, but it is time to improve the sustainability performance of centralised wastewater treatment processes by implementing innovative resource recovery technologies (Bdour et al. 2009; Li et al. 2014).

To achieve that, innovative technologies but also integrative and multidisciplinary strategical planning, process design and management methods are required. To design innovative processes from a resource recovery perspective requires to explore and understand their techno-economic performance, quantitative recovery potential, energy consumption and trade-offs implied by choosing a certain recovery technology over another one. In addition to the exploration of the technical process design space and performance, water resource factories need to be designed to fulfil several other objectives that lie outside the scope of traditional wastewater treatment plant engineering efforts. Important general requirements for water resource factories are:

(i) The fulfilment of different effluent quality regulations that allow safe reuse of reclaimed water (Li et al. 2014);

(ii) The economic feasibility and cost-effectiveness of the process;

(iii) The decrease of greenhouse gas emissions by increasing the in-plant energy efficiency through energy recovery while simultaneously minimizing the in-plant energy consumption (Wan et al. 2016);

(iv) Value chain development that leads to the recovery of valuable products which can be produced in quantities, qualities and at costs that match the current market demand and prices (van Loosdrecht and Brdjanovic 2014);

(v) The fulfilment or emplacement of legal and policy guidelines (van der Hoek et al. 2016);

(vi) The change of people's perceptions on the re-consumption of wastewater derived products that may evoke reluctance to accept this approach (Yi et al. 2011).

These diverse requirements indicate the need for a new, holistic, integrative and multidisciplinary process design and planning approach for water resource factories. To fulfil that, this dissertation aims to provide insights not only about single technological resource recovery solutions but more on process integration possibilities. It aims to improve the understanding of the implications that come with the integration of several recovery technologies into a treatment process. Studying the trade-offs between possible recovery technologies and their quantitative resource recovery potentials, as well as the techno-economic performance and energy consumption of processes, contributes to better informed decision making in water resource factory design. Beyond that, designing processes for successful implementation requires the consideration of factors that lie outside the technical design space. Therefore, the marketability, policies and people's perception towards recovered resources are considered as well. This holistic research approach leads to the following research questions and structure of this dissertation.

1.2. Outline and research questions

Covering the vast research field of resource recovery from municipal wastewater from a holistic perspective requires to understand the state of the art in recovery technologies and their potentials and bottlenecks. Therefore, in **chapter 2** data and information from literature was collected to conduct a critical and extensive review over the research field. During the review process, three main questions arose that had not been tackled yet in an inclusive manner elsewhere.

(1) What is the quantitative potential of municipal wastewater to supply resources for a society?

(2) Which technologies have already been widely applied to recover water, energy, fertilizers, and other products from municipal wastewater treatment plants? Which recovery technologies are ready and could be applied but are not? Which technologies are currently under development?

(3) What are the bottlenecks for the successful implementation of resource recovery technologies? How to categorize these bottlenecks and how do they relate to water management utilities that are responsible for wastewater treatment innovation?

To answer these questions, the recoverable quantities of 11 resources contained in municipal wastewater and the overall market sizes of these resources have been calculated for a certain geographical scope to reveal their market supply potential. Then scientific articles and books have been screened to identify technologies for water, energy, fertilizer and other product recovery from municipal wastewater treatment plants. Each technology is critically reviewed regarding its potentials and shortcomings to become successfully implemented. Finally, a detailed overview of bottlenecks mentioned in scientific literature that may hinder the successful implementation of resource recovery technologies is presented. The bottlenecks are categorized into the three categories of (i) economics and value chain development, (ii) environment and health, and (iii) society and policy. Finally, the power of a water management utility to influence revealed bottlenecks is discussed.

Chapter 3 explores process integration possibilities by analysing how a water resource factory could be designed that is based on the aerobic granular sludge process. Since the organic matter and phosphorus contained in the influent can only be recovered once and numerous process designs are possible to reach that, the following questions arise.

(1) How much energy and materials can be recovered by different possible process designs?

(2) Which trade-offs regarding recoverable resource quantities exist? How much COD can be recovered as energy and/or biopolymers? How much phosphorous can be recovered as struvite and/or from sludge incineration ash?

(3) Among the possible alternatives, what are the preferable process design choices when resource recovery rates are the major design criteria and why?

To answer this, five different possible process designs that would recover chemical oxygen demand as energy and/or extracellular polymeric substances, and phosphorous as struvite have been modelled with mass and energy balances. Mass and energy balances are an excellent tool to explore potential resource recovery possibilities in early stage water resource factory design. Estimating quantities, trade-offs and synergies between certain recovery technology integrations provides a basis for better informed decision making and objective formulation.

Since water is the most precious resource contained in municipal wastewater, **chapter 4** explores water reclamation processes. The reclamation of water is more critical than energy because for the latter many alternative sources exist whereas for water there are less choices (Ma et al. 2015). Water scarcity is the geographic and temporal mismatch between freshwater demand and availability and increases in many regions worldwide (Mekonnen and Hoekstra 2016). The reclamation of water from municipal wastewater has been widely recognized as a practical alleviation of regional water scarcity and is therefore promoted politically by the European Commission (European Commission 2018). Water is with ca. 99% the most abundant resource in municipal wastewater which makes it also the most economically interesting resource to recover (Verstraete et al. 2009). Pressure-driven membrane-based advanced treatment processes (MATPs), especially those including ultrafiltration (UF) and reverse osmosis (RO), are the centre technology to reclaim water for industrial, potable, and agricultural reuse. However, to make choices in WWTP effluent reclamation process design knowledge is required about:

(1) Which reuse type (i.e. industrial, potable, agricultural) is preferable from a techno-economic reclamation process perspective?

(2) What are the specific recovery rates, energy consumption and net costs of processes that reclaim water for these different reuse types?

(3) Is it useful to design a process that can reclaim WWTP effluents with different qualities to target multiple reuse types simultaneously?

(4) Which renewable energy solutions (i.e. electricity recovery via anaerobic sludge digestion and photovoltaic) could minimize the fossil energy consumption of membrane-based effluent reclamation processes?

(5) How can recovery rates, energy consumption and costs become optimized for reclamation processes that rely on reverse osmosis (RO) to improve the sustainability performance of this key technology?

To answer these questions the differences in recovery rates, energy consumption and net costs of effluent reclamation processes are revealed by modelling four different membrane-based WWTP effluent reclamation processes. One reclaims demineralised water for industrial applications, another reclaims potable water, and two processes reclaim irrigation water of different quality. Mass and energy balances are conducted to estimate the recovery rates and energy requirements of modelled processes while a cost and benefit analysis reveals their economic performance. In addition, a fit for multi-purpose process design concept is discussed. Finally, an optimized process that integrates a softening and a bio-stabilizing process prior to the RO unit is investigated and renewable energy integration possibilities are presented.

Chapter 5 follows a systems thinking approach and combines the insights of previous chapters into an integrative and holistic framework for water resource factory design and planning which has been named "SPPD-WRF Framework". It looks beyond the technical feasibility and also includes important aspects that lie outside the technical process but may determine its successful implementation. To achieve that, several questions need to be answered:

(1) How to strategically plan the design and implementation of water resource factories step by step considering unique site specific circumstances?

(2) Since various process designs are possible due to the diverse technological options, what are useful criteria to assess water resource factory processes from a holistic perspective measuring a process's technical feasibility, its economic performance and environmental impacts?

(3) Which criteria are useful to assess the marketability and value chain development potential of recoverable resources?

To answer these questions, the wastewater treatment plant design space was opened up for a variety of expertise that complements the traditional wastewater engineering domain. Useful criteria to assess the technical feasibility, the environmental impacts and the economic performance are integrated into the SPPD-WRF Framework. Moreover, criteria to assess the marketability and value chain development potential for recoverable resources are introduced. In addition, the inclusion of stakeholders in the design and decision making process is considered. Due to the multidimensional and multidisciplinary approach of the framework, it can help decision makers to design processes from a circular economy perspective and assess their sustainability at an early design stage. This extends the rather narrow focus of traditional wastewater treatment plant design strategies that often merely consider robust treatment performance and process costs.

Chapter 6 concludes the main findings from this dissertation and provides a short outlook for future research activities.

2

A critical review of resource recovery from municipal wastewater treatment plants

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“The proper use of science is not to conquer nature but to live in it”.

Barry Commoner

2.1. Introduction

Although wastewater resource recovery technologies have been extensively elaborated by the scientific community in recent decades, their large-scale implementation in municipal wastewater treatment plants (WWTPs) is still poor. This can primarily be explained by various non-technical reasons, as well as by 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 (CAS) process, in which aerobic microorganisms metabolise the organic fraction present in the wastewater under constant oxygen supply (Oh et al. 2010). Although the CAS process succeeds in meeting 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 a 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 other products – their redesign from treatment facilities into water resource factories (WRFs) provides possibilities to contribute to a more circular economy. Within academia, it seems clear that current wastewater treatment practices are based on outdated concepts established in the early 20th century. It seems inevitable that we will have to develop new practices 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 sector 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 consumes resources (Daigger 2009). Due to increasing numbers of available resource recovery technologies, WWTP process design is no longer a simple technical problem, but a complex issue that requires an integrated approach in order to make effective decisions (Bozkurt et al. 2017). The question 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 WRFs (Li et al. 2015).

In addition to technical uncertainties that are 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 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. 2017), 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, as yet there is no review available that provides a holistic overview of the field.

This paper seeks to fill that gap by providing a holistic overview of resource recovery from municipal WWTPs. Data to calculate the potential of 11 resources recoverable in municipal WWTPs to supply markets in the Netherlands and Flanders (Belgium) was derived from a literature review. Resource recovery technologies investigated in academia were then comprehensively and critically reviewed. Finally, bottlenecks discussed in the reviewed literature that have to be overcome to successfully implement these technologies into WWTPs were categorised and analysed. By covering the market supply potential, the vast technical possibilities and the bottlenecks, this paper can inform 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, it is time to improve these plants' sustainability performance by integrating innovative resource recovery technologies into treatment-process designs (Bdour et al. 2009).

2.2. Market supply potentials of recovered resources

The market supply potential of resources recovered from municipal wastewater is shown in Table 2.1. It indicates what role municipal WWTPs could potentially play in a circular economy if resource recovery routes (RRRs) were implemented nationwide. The supply potential for each resource is calculated on the one hand from the quantities that 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 was 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 it 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 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.

Table 2.1 Calculated market supply potentials of water, energy, fertilizer and other products recoverable from municipal WWTPs in the Netherlands or Flanders.

Resource demand		**Potential resource recovery from WWTPs**		**Market supply potential %**
Water demand	***Netherlands***	**Water recovery**	***Netherlands***	**Water**
Water abstraction [a]	9482m m^3/a	Effluents [a1]	1909m m^3/a	20
		Treated by MF-UF [a2]	1622m m^3/a	17
		Treated by MF-UF/RO [a3]	1217m m^3/a	13
Energy demand	***Netherlands***	**Energy recovery**	***Netherlands***	**Energy**
Natural gas [b]	1227 PJ/a	CH_4 from COD (anaerobic) [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) [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) [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
Imported 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) [j1]	1,9 PJ/a	0,2
		Electricity CH4 (CHP) [k1]	0,7 PJ/a	0,2
		Electricity (cellulose combustion) [k2]	0,7 PJ/a	0,2
		Heat CH_4 (CHP) [l1]	88 PJ/a	1

		Heat (cellulose combustion) [l2]	1,2 PJ/a	1
VFA demand	***Global***	**VFA recovery**	***Netherlands***	**VFA**
Acetate [m]	16000 kt/a	Acetate recovery [m1]	142 kt/a	1
Propionate [m]	380 kt/a	Propionate recovery [m2]	64 kt/a	17
Butyrate [m]	500 kt/a	Butyrate recovery [m3]	29 kt/a	6
Alginate demand	***Global***	**EPS recovery**	***Netherlands***	**EPS**
Production [o]	30 kt/a	Potential EPS production [o1]	76 kt/a	252
Fodder demand	***Flanders***	**SCP recovery**	***Flanders***	**SCP**
Fodder N consumed [p]	149 kt/a	Influent N [p1]	24 kt/a	16
		SCP from anaerobic digestate [p2]	4,8 kt/a	3
CO_2 demand	***Netherlands***	**CO_2 recovery**	***Netherlands***	**CO_2**
Industrial CO_2 [q]	1239 kt/a	CO_2 from biogas in WWTPs [q1]	53 kt/a	4

Footnotes:

[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 (199). [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 the full influent cellulose fraction is sieved out (Ruiken 2010).

[j1]Total COD into Dutch WWTPs per year = 946000 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 = 180000 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 = 180000 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 = 946000 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).

[o]Global conventional alginate production (Pawar and Edgar 2012). [o1]EPS recovery: total COD into Dutch WWTPs per year = 946000 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] CO_2 contained in biogas recovered in Dutch WWTPs in the year 2012 (Hogendoorn et al. 2014).

2.2.1. 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 of 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 80000 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. Therefore, even if a large fraction of WWTP influent is reclaimed, it can only partly satisfy total regional demand for fresh water. As shown in the examples in Table 2.1, 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 could reduce it by 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.

2.2.2. Energy supply potential

A municipal WWTP can provide a significant share of the total energy consumption of 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 2.1, 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 were converted into electricity and heat in a combined heat and power (CHP) unit of typical efficiency (ca. 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 were dewatered and then co-combusted in coal-fired power plants, the amount of electricity obtained would be a negligible 0.1% of overall consumption. The main reason for the low energy-recovery potential of sludge incineration is that a considerable amount of energy is required to evaporate its water content, as sludge is often 80% water 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 were installed to recover heat energy of 5 ºC, 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 than 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 2.1).

2.2.3. Fertilizer 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 2.1 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 2.1, 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.

2.2.4. Supply potential of other products

As exemplified for the Dutch case, in addition to fertilizers, multiple products – for example, cellulose, volatile fatty acids (VFAs), extracellular polymeric substances (EPS), single-cell protein (SCP) and CO_2 – can be recovered from wastewater. In principle, more products can be recovered from wastewater, but data on such routes is still limited, which gives rise to 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 mentioned above, namely 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 2.1, if the cellulose fibres were used as raw material for the Dutch paper and paper board industry, they would have the potential to satisfy 7% of demand from this sector. In all, 180000 t of toilet paper are flushed down Dutch toilets every year. As this represents approximately 180000 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 were the fibres to be 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 figures on the global production volumes of the three main VFAs differ 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 were converted into PHA, it is likely that a significant share of European PHA production could be supplied by the combined Dutch WWTPs. However, estimates of annual PHA market sizes vary greatly from almost 150000 tonnes European market size (de Jong et al. 2012) to 100000 tonnes global market size (Pratt et al. 2019), which makes it difficult to estimate a reliable supply potential.

If Dutch influents were invariably treated using aerobic granular sludge processes, and 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 much more environmentally efficient than the production of conventional fodder. Its potential to satisfy Flemish demand for fodder, however, is rather limited: it could substitute only 3% of conventional fodder N because 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 substantially towards achieving the greenhouse-gas emission-reduction target of the Dutch water boards. It could also 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).

2.3. 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 other products can be recovered from wastewater (van Loosdrecht and Brdjanovic 2014). This section critically discusses RRRs for these four resource categories. 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.

2.3.1. 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, such as 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 1.1).

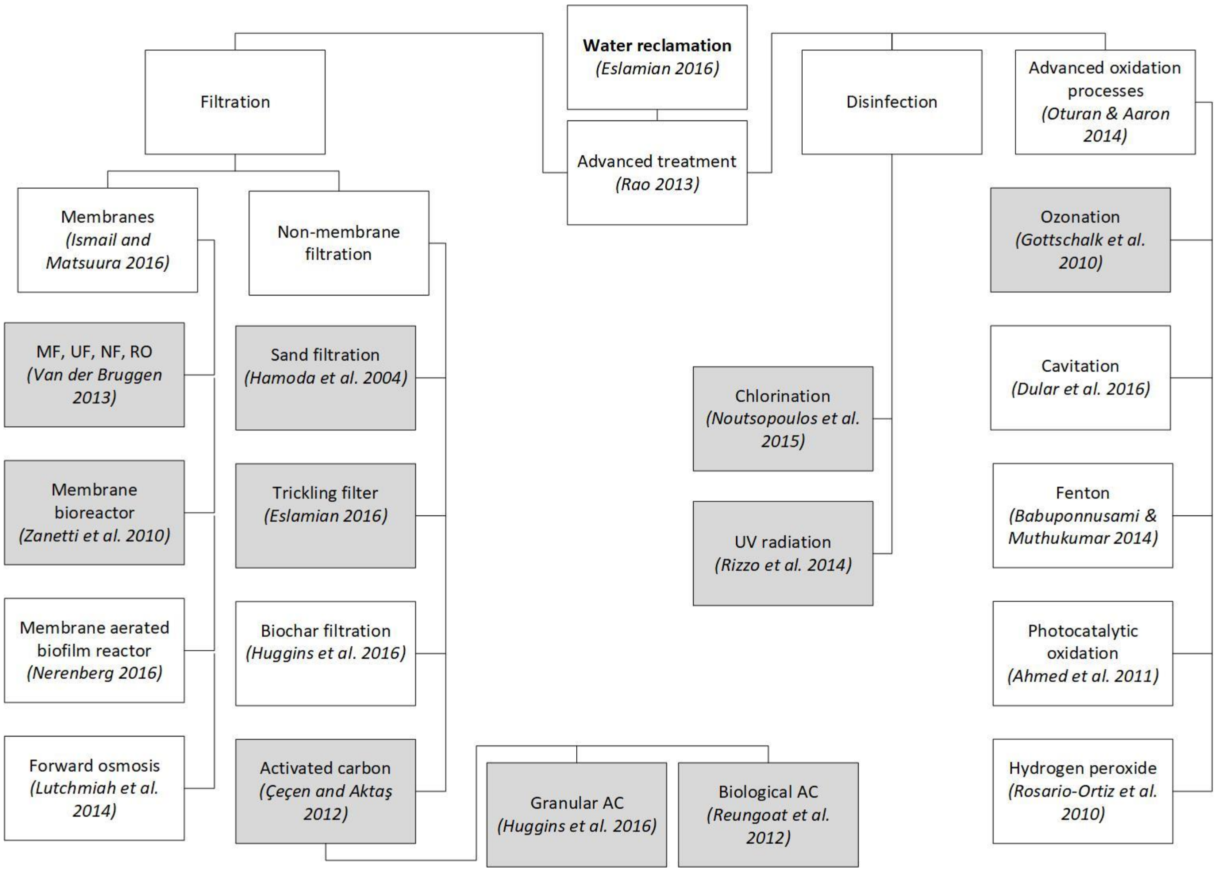

Figure 2.1 Examples of technologies to reclaim water from municipal WWTPs. Since a detailed presentation and discussion of these technologies is beyond the scope of this paper, scientific publications that explain or review them are referenced. Grey shading indicates techniques that have been applied on a large scale in municipal WWTPs. Unshaded boxes show technologies that are not widely applied.

2.3.1.1. 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, being 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 the use of NF/RO membrane technology to recover water from wastewater for indirect potable reuse can be found in Singapore, as part of the NEWater project. The process 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 frequently applied, 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 solids concentrations, which allow for smaller reactors. 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, higher aeration requirements (Judd et al. 2008).

Although membrane technologies can provide very high quality effluent, 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 increases capital costs, as more membrane units are necessary (Pearce 2008). To decrease potential fouling and clogging, effective operation requires extensive pre-treatment of secondary effluents (Wintgens et al. 2005). An additional cost factor for efficient large-scale membrane-technology application for wastewater reuse arises from disposing of the complex 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).

2.3.1.2. Activated carbon filtration

Activated carbon (AC) filtration as an advanced treatment process can produce higher quality effluent that is 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 0.07 mm 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 various types of pesticides during the production of drinking water. 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 pre-treatment 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).

2.3.1.3. Advanced oxidation processes

The removal of emerging pollutants like pharmaceuticals is a growing concern in wastewater treatment (Ranade and Bhandari 2014) and certainly needs 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 the final polishing and disinfection step after biological treatment, but can also be used as a pre-treatment 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 and concentration and the required effluent quality. Besides the sequential application of various AOPs to enhance the 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 (Petrovic et al. 2011; Oller et al. 2011; Wang and Xu 2012; Oturan and Aaron 2014). 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 limits 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 that contains 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 (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). Microorganisms 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 microorganisms present in the wastewater. Moreover, the physiochemical parameters of the treated effluent, such as turbidity, hardness, suspended solids, iron, manganese and humic acids content, can be disruptive factors preventing UV light waves from reaching all microorganisms (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 is accompanied by potential risks. Harmless substances can react with the disinfectant and form harmful molecules, 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).

2.3.1.4. Summary: water reclamation and reuse

Successful wastewater reclamation and reuse is hindered not only by technology-related bottlenecks but also by more general ones. Taken together, these indicate that such reuse might be a valid option only 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 greater 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 the demand for and 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 being 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 there eventually had to be scrapped due to public safety concerns. Similar cases are reported from 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 increases 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 that are cost effective (Bdour et al. 2009).

The use of reclaimed water for the irrigation of crops also entails 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 cooperation 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 lack 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 commendable that the European Commission established the 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).

2.3.2. 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 by a similar amount (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. The treatment of municipal wastewater currently accounts for about 4% of the national electricity consumption in both the United States (Wang et al. 2015a) and the United Kingdom (Oh et al. 2010). As shown in Figure 2.2, the recovery of fuels from wastewater is achievable through the application of different technologies. The chemical energy in typical municipal wastewater is 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), although in the latter process 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 is usually in the range of 30–50% (Wan et al. 2016), depending on the country concerned.

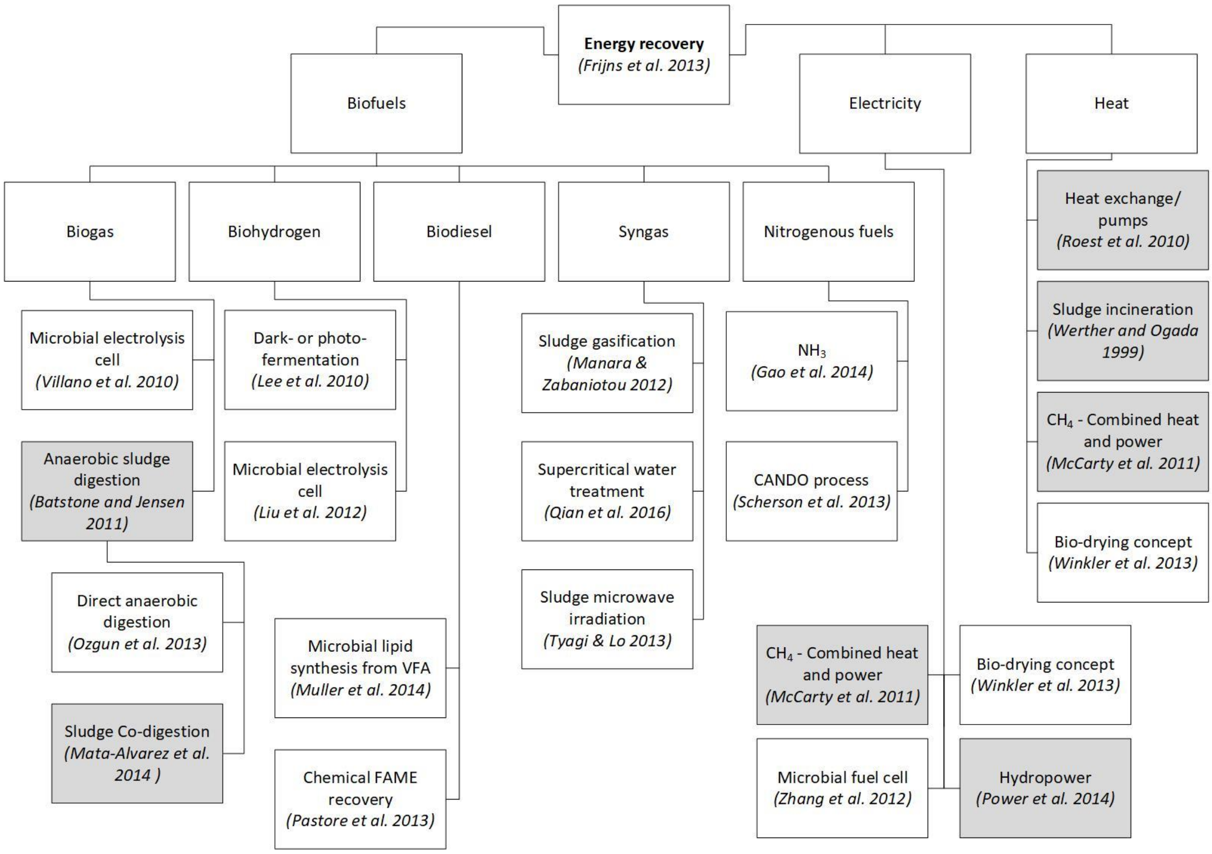

Figure 2.2 Examples of technologies to recover energy from municipal WWTPs. Since a detailed presentation and discussion of these technologies is beyond the scope of this paper, scientific publications that explain or review them are referenced. Grey shading indicates techniques that have been applied on a large scale in municipal WWTPs. Unshaded boxes show technologies that are not widely applied.

2.3.2.1. Methane

The production of biogas by anaerobic sludge digestion is currently the most widely used energy recovery method, and it is applied worldwide on different scales (Rulkens 2008a). 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 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 of 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, that is, anaerobically, for example in anaerobic membrane bioreactors (AnMBRs) or up-flow anaerobic sludge blanket (UASB) reactors. 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 used in large conventional plants for treatment of the sludge line, not the water line (Logan and Rabaey 2012).

2.3.2.2. 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 the fast gasification of wet sewage sludge (Manara and Zabaniotou 2012) – a thermal conversion process that 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 been reported and reviewed elsewhere (Qian et al. 2016).

Hydrogen can also be recovered from wastewater by biological means, namely in a two-step anaerobic sludge treatment process limited to hydrolysis and acidogenic fermentation by phototropic and/or lithotrophic microorganisms. 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 the quite low yields (Lee et al. 2010).

Biodiesel is another fuel that can be derived from sludge. Lipids can represent a significant proportion of the organic fraction in municipal wastewater and specialised microorganisms 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. 2017). One major bottleneck, however, is that the performance of phototrophic organisms depends on climatic conditions that are not available all year round in countries that have 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 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, for example by stripping,. 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. However, processes that recover ammonia for fuel generally consume more energy than they recover, which makes them economically unfeasible. Another major problem with these routes is the low N concentrations in municipal wastewater. Thus, recovering ammonia as fertiliser instead of as an energy carrier seems preferable (Gao et al. 2014).

2.3.2.3. Sludge incineration

When sewage sludge is incinerated, complete oxidation of its organic content is achieved, thus forming CO_2, water and inert material (ash), all of which have to be disposed of. The ash can be used, for instance, 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 are 25–30%. 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 increase 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 achieve a positive energy balance from combustion, the water content needs to be reduced to below 30% – which usually requires energy and therefore creates costs (McCarty et al. 2011). The actual energy recovery 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 in the plant.

2.3.2.4. Bioelectrochemical systems

In bioelectrochemical systems (BESs), COD is oxidised by microorganisms 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 feedstock 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 microorganisms 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 that are 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. These product groups are:

- 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, the 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, BESs 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 a practical solution for resource recovery in WWTPs.

2.3.2.5. 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 that is 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 in the influent temperature caused 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 3–4 units of heat energy per unit of electrical energy consumed (Mo and Zhang 2013). In addition to heating or cooling buildings, a 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, such as aquifers (van der Hoek et al. 2016). Selling surplus heat to nearby consumers is recommendable, but especially in 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, with capacities of 10–20 MW, were already operational (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, the city government achieved energy savings of 20–30% by 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).

2.3.2.6. 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 deliver reliable performance when applied to an effluent flow. However, if such technologies are applied to untreated wastewater, they must be made from stainless steel 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 rise. Economic viability is therefore always site specific and depends not only on physical circumstances, such as the technology selected, but also on both present and future market conditions (Power et al. 2014). Although individual large-scale applications in Australia, the UK and Ireland have proven the economic viability of hydropower technologies in WWTPs, most scientific case studies lack a detailed analysis of this factor. 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).

2.3.2.7. 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 water industry's strong focus 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 be recovered from COD. This aspect is gaining increasing attention, as discussed below.

2.3.3. 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 emissions of anthropogenic greenhouse gas (GHG) and demand for energy. Over 90% of these emissions are related to the production of ammonium fertiliser (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 in 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). Mining P from rocks has a huge environmental impact because it generates by-products like gypsum, which are often contaminated with radioactive elements and heavy metals and are not disposed of in an environmentally 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 widely studied and a variety of solutions have been developed (see Figure 2.3). 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 biological, chemical or physical methods.
2. Release of nutrients by biological, chemical or thermal methods.
3. Nutrient extraction and recovery in the form of concentrated fertilizer, by chemical or physical methods.

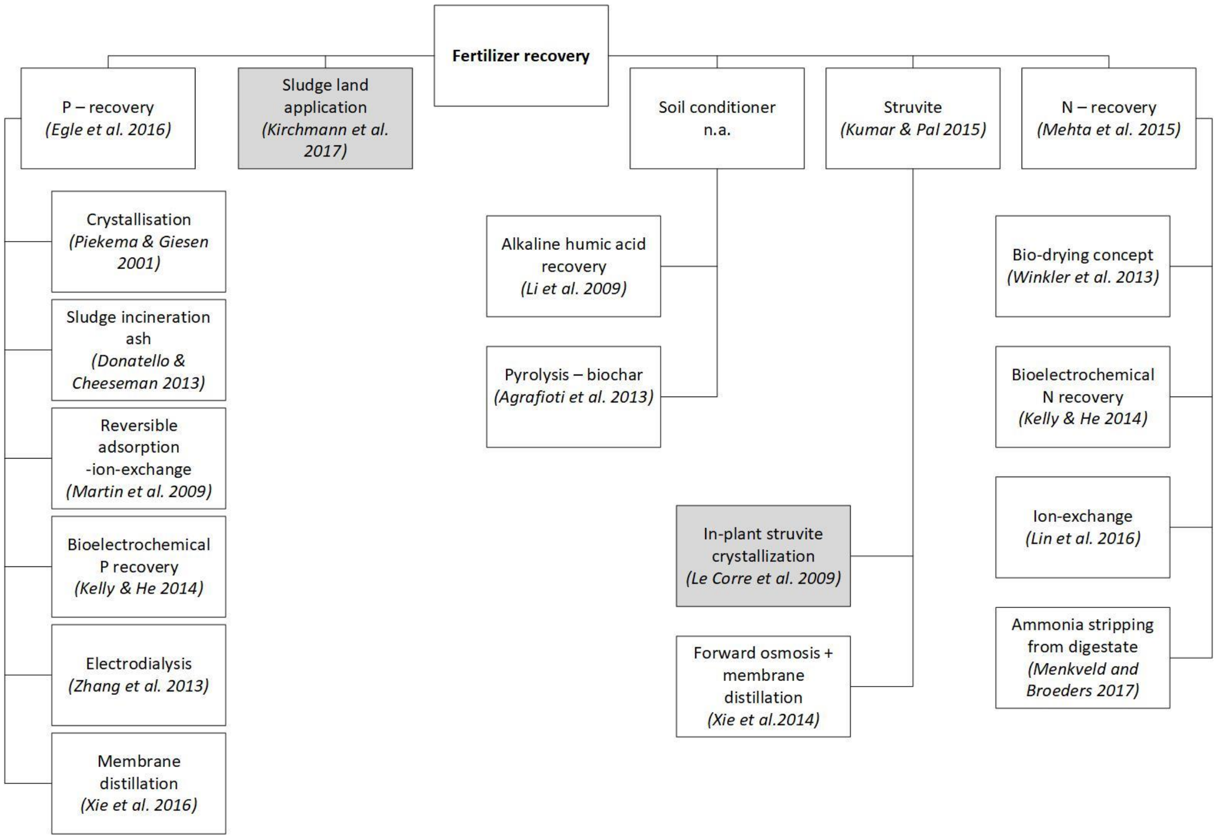

Figure 2.3 Examples of technologies to recover fertiliser from municipal WWTPs. Since a detailed presentation and discussion of these technologies is beyond the scope of this paper, scientific publications that explain or review them are referenced. Grey shading indicates techniques that have been applied on a large scale in municipal WWTPs. Unshaded boxes show technologies that are not widely applied.

2.3.3.1. Land application of sludge

Wastewater fertiliser recovery currently 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). 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 a soil's organic carbon content. The transport 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).

2.3.3.2. Struvite

Struvite precipitation as a recovery route for ammonia and phosphate has gained a lot of interest among researchers in recent decades, and it is applicable on a large scale (Le Corre et al. 2009). Struvite is magnesium ammonium phosphate ($MgNH_4PO_4{\cdot}6H_2O$), a mineral commonly formed in WWTPs through spontaneous precipitation if Mg concentrations are high enough, although this is often not the case. The formation and growth of struvite crystals in 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 struvite usable as a slow-release commercial fertiliser suitable for soils with a 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 100 mg/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 P concentrations (Zhang et al. 2013; Xie et al. 2016). Nutrient enrichment is usually required prior to struvite precipitation and recovery from side streams in 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 that preclude struvite formation (Wilfert et al. 2015). Due to those wastewater P fractions that are fixed in biomass or bound to metals like Fe and are 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 by continuously removing biomass (Sheik et al. 2014), the recoverable amounts are rather low and unlikely to exceed 1 kg of struvite per 100 m^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 be as high as €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 be included when assessing the cost effectiveness of struvite recovery. 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 thus limit its safe agricultural application (Xie et al. 2016). For example, it has been revealed that recovered struvite crystals can contain arsenic concentrations of up to 570 mg/kg^{-1} (Lin et al. 2013). Successful struvite recovery can also be hindered by a lack of legal regulation. It was first successfully recovered in the Netherlands in 2006, but 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). Despite the change in the law, however, no breakthrough in the implementation of struvite recovery seems to have occurred. It must therefore be questionable how severely that legislative bottleneck actually impacted the use of the technique.

2.3.3.3. Sludge incineration ash

Technologies that recover P from sludge incineration ash are currently in focus because they promise high influent P recovery rates. In order to achieve high recovery efficiencies, however, they require special incinerators 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 in 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 pre-treatment. But this does not involve P recovery (Mehta et al. 2015).

2.3.3.4. Soil conditioner

Used alongside mechanical and thermal methods, alkaline treatment is a simple and highly efficient chemical means of disintegrating sludge. In addition to reducing the volume of 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 that is more difficult in the case of humic acids 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 45 µm 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 can 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, such as 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 both hard to predict and very site specific (Jeffery et al. 2011).

2.3.3.5. Membrane-based nutrient recovery

Electrodialysis, membrane distillation and forward osmosis are emerging nutrient-recovery technologies that have been 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 chemicals 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 available. These technologies therefore remain a fairly theoretical option, still a long way from practical application in large-scale wastewater treatment facilities (Xie et al. 2016).

2.3.3.6. 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, which gives this route a competitive disadvantage (Khiewwijit et al. 2016). Numerous new P recovery technologies have been developed for various access points in WWTPs, and in recent years some of them have been implemented at full scale. 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 several times over, as shown by (Cornel and Schaum 2009) for German market conditions. As well as conventional fertiliser, manure from livestock production also competes with fertilizer recovered from wastewater. (Coppens et al. 2016) show that, in Flanders, the P entering WWTPs could fulfil 14% of total local fertiliser P demand, while the 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 2.4.

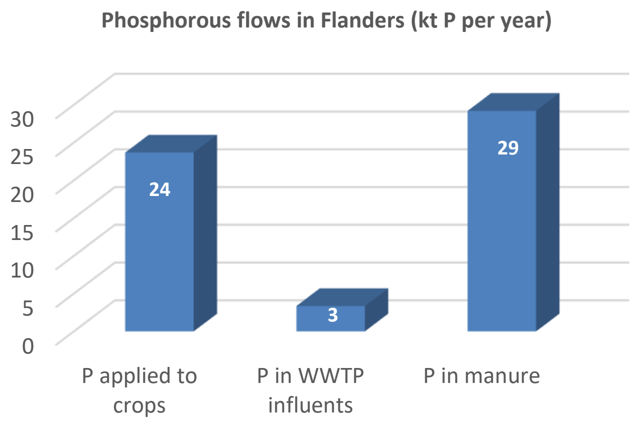

Figure 2.4 Phosphorous flows (kt/yr^{-1}) in the livestock-intensive region of Flanders (Belgium) (Coppens et al. 2016).

However, P can be recovered in a WWTP at different stages of 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 CPR, or both, 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}, physico-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 that is released into the atmosphere. The 25% organic influent N consists partly of urea and hydrolysed proteins, both of which are 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).

2.3.3.7. Other product recovery technologies

Besides fertilizers, various other products can be recovered from wastewater, as shown in Figure 2.5. A number of publications point out the potential contribution towards sustainable development that is achievable by applying product recovery technologies in 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 application, none is yet reported as being widely used.

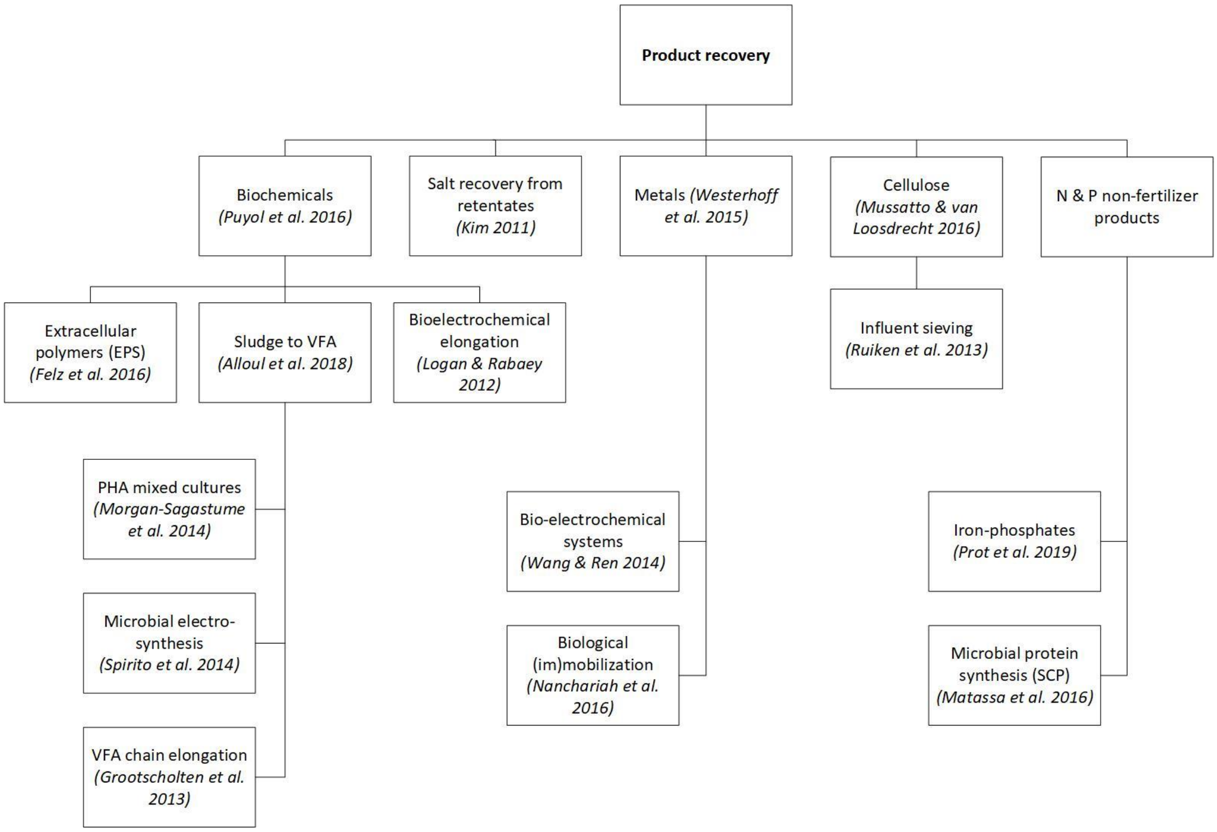

Figure 2.5 Examples of technologies to recover other products from municipal WWTPs. Since a detailed presentation and discussion of these technologies is beyond the scope of this paper, scientific publications that explain or review them are referenced.

2.3.3.8. 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 do so, 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 using a surfactant like sodium dodecylbenzene sulphonate and 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 that are 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 the associated disposal costs (Jie et al. 2014).

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 (Kleerebezem et al. 2015). Another bottleneck is the solubility of VFAs, which 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 that is then separated from the liquid (Kleerebezem et al. 2015). However, very few studies have examined all pertinent parameters of VFA production routes from waste streams and most of the variables have yet to be examined satisfactorily. Such uncertainties contribute to the fact that most waste-based VFA production concepts are still confined to the laboratory (Lee et al. 2014).

Although higher added-value products can be derived from VFAs, this does not imply that waste-based VFA production is economically preferable to 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 well at low COD concentrations is required (Alloul et al. 2018).

2.3.3.9. Polyhydroxyalkanoates (PHAs)

One possibility for the refining of VFAs is to convert them into PHAs, which are fully biodegradable biopolyesters that are able to substitute fossil-fuel derived polymers. Due to their comparable properties, 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 cells (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.
- 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).

An alternative procedure was trialled in 2015 by Dutch water utilities, which collaborated in a one-year pilot study on activated sludge in 15 WWTPs using biological nutrient removal (BNR). BNR plants with pre-denitrification exert an anoxic feast period on the biomass, and this promotes PHA-storing biomass. Up to 50% gPHA/gVFS have been harvested from these WWTPs without changing their infrastructure (Pratt et al. 2019).

However, the PHA yield on the substrate and the efficiency of the downstream processing lead to costs that are 20–80% higher than those for petrochemical polymers of a similar quality (Fernández-Dacosta et al. 2015). Value creation from wastewater-derived PHA depends on the security of polymer supply in terms of both quantity and quality, but until recently no study had provided a clear answer to the question whether a mixed culture process could fulfil these criteria (Pratt et al. 2019). 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).

2.3.3.10. 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 that have a 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 have 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 such shaping is successful, how to construct a stable and resilient system suitable for industrial-scale application need further study. In addition, improved extraction technologies are needed, 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).

2.3.3.11. Extracellular polymeric substances (EPSs)

In recent years, the aerobic granular sludge (AGS) process – which is also known as the NEREDA process has been applied successfully in 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). EPSs 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 EPSs from AGS is a potential future product recovery route that can yield a high-value product. In the Netherlands, plans have been drawn up for two full-scale demonstration systems for commercially viable and sustainable EPS recovery (van der Roest et al. 2015).

A method using sodium carbonate (Na_2CO_3) and calcium ions (Ca^{2+}) extracts EPSs 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 EPSs 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 EPSs. 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 with these materials and their composites. (Tseggai 2016) indicates that the range of possible applications for EPSs, both as a composite and as a raw material, is extensive. If alginate is to be substituted with wastewater-derived EPSs, however, they must be produced more cheaply than conventional alginate, not least because the current level of production – namely 30000 tonnes annually – is estimated to be only 10% of the alginate-like material potentially obtainable from wastewater. This indicates a high unexploited potential for conventional production, which 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).

2.3.3.12. 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 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 to be 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 integrated into the CAS process, however, it would recover only the influent N that ends 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.

2.3.3.13. Iron phosphate

Significant iron (Fe) loads can enter a WWTP via Fe-rich industrial wastewater, groundwater infiltration and 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 in 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. 2016). 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 requires more knowledge about the factors that determine 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 a plant's different units, 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, a better understanding of these mechanisms is needed (Wilfert et al. 2015). Nevertheless, an innovative pilot system ('ViviMag') using magnetic separation to recover vivianite from digested sewage sludge is under construction in the Netherlands (Wetsul.nl 2019; Prot et al. 2019).

2.3.3.14. 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 more than 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 (<0.5 mm) 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 (e.g. 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).

2.3.3.15. Summary: product recovery

Initial findings concerning some of the product recovery routes reviewed above show promising results in terms of quantities and market prices (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 choosing a specific route. This is the case with, for example, the recovery of EPS and PHA (van der Hoek et al. 2016). As mentioned, however, the consumer's association of wastewater-derived products with faecal matter is a major 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 are 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).

2.4. Bottlenecks in wastewater resource recovery

As discussed above and presented in Table 2.2, a variety of issues that may hinder the successful implementation of RRRs are mentioned in the scientific literature. These relate to nine bottlenecks that can be grouped into three categories (A, B, C).

<u>Economics and value chain (A)</u>

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

<u>Environment and health (B)</u>

7. Emissions and health risks.

<u>Society and policy (C)</u>

8. Acceptance.
9. Policy.

Most of the bottlenecks are in the economics and value-chain development category. This reflects the findings of (van der Hoek et al. 2016), who state that particularly market potential and competition introduce uncertainties in respect of successful resource recovery from wastewater. However, some of the bottlenecks presented in Table 2.2 overlap other categories and so should be perceived as interlinked rather than absolute. Moreover, rather than interpreting bottlenecks as barriers to the implementation of resource recovery routes, they should be seen 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.

However, a general policy related bottleneck for wastewater resource recovery implementation is the definition of waste. The end-of-waste concept has been manifested in the Waste Framework Directive 2000/98 to re-introduce recovered products from solid waste streams into consumption and change their definition by fulfilling certain end-of-waste criteria. The criteria shall promote product standardization and quality and safety assurance and improve harmonization and legal certainty in the recyclable material markets (Zorpas 2016). Moreover, recovering materials from waste streams in accordance with generally accepted criteria gives a positive association with the materials as they are not any longer labelled as waste but as new and safe-to-use products. End-of-waste criteria need to be developed in accordance with the following four conditions: (i) the substance or object is commonly used for a specific purpose; (ii) a market or demand exists for such a substance or object; (iii) the substance or object fulfils the technical requirements for its specific purpose and meets the existing legislation and standards applicable to products; and (iiii) the use of the substance or object will not lead to overall adverse environmental or human health impacts (Saveyn et al. 2014). However, the end-of-waste concept has been considered so far only marginally in the field of wastewater resource recovery. It has been suggested mainly for the use of sewage sludge as feedstock or co-substrate for biogas production (Saveyn et al. 2014) or as soil amendment product (Zorpas 2016; Kacprzak et al. 2017) and for nutrient recovery strategies (Dereszewska and Cytawa 2016). The bottlenecks identified in this review hint clearly that the end-of-waste concept is yet insufficiently considered for most resources recoverable from municipal wastewater. Reasons are that active support from legislators and governance is lacking because recycling is mostly governed by fragmented decision-making in regional administrations. Active regulatory support such as recovery obligation or subsidies are yet missing in many countries. For example, to facilitate the marketing of recovered P the inclusion of such materials into existing fertiliser regulations has to be focused (Hukari et al. 2016).

Table 2.2 Detailed overview of bottlenecks mentioned in scientific literature that may hinder the successful implementation of RRRs in municipal WWTPs. Bottlenecks are categorized into (A) economics and value chain, (B) environment and health, or (C) society and policy.

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. Per m^3 water reclaimed by secondary effluent treatment with ultrafiltration and reverse osmosis a cost of 0,46 € and a benefit of 0,25 € has been calculated	(Verstraete et al. 2009; Batstone et al. 2015)
			Fouling as an additional cost factor for membrane technologies. Costs vary greatly and depend on membrane characteristics, operating conditions, feedwater quality and applied cleaning techniques	(Yangali Quintanilla 2010)
			Disposal costs of membrane retentate depend on level of treatment, retentate characteristics and disposal method	(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. 2014)
			NH_3 recovery for fuel is not cost effective because energy costs of removing NH_3 often exceed the energy and value of recovered gas	(Gao et al. 2014)
		Fertilizer	P recovery costs exceed conventional P ore costs. Assuming a load of 660 g P per capita per year, recovery costs would be 3.600-8.800 € per tonne recovered P under German market conditions	(Cornel and Schaum 2009)
			Struvite recovery processes may not be cost effective which depends strongly on profits from struvite sales. Market prices vary greatly	(Le Corre et al. 2009)

			and have been estimated for e.g. Australia to lie between 180-330 € per tonne	
			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)
		Other products	PHA recovery processes can be more costly than conventional production routes. Recovery costs depend greatly on applied downstream processes and may range between 1,4 - 1,95 € per kg	(Fernández-Dacosta et al. 2015)
			CO_2 recovery from biogas is economically feasible only if a biogas upgrading unit is already present. Payback times for recovery equipment may vary between 1-12 years	(Hogendoorn et al. 2014)
			Bioelectrochemical systems may require expensive electrodes (e.g. platinum cathodes)	(Villano et al. 2010; Logan and Rabaey 2012)
			Microbial electrolysis cells using CO_2 for chemical production require extra energy input depending on the electron donor used. The potential of municipal wastewater as electron donor is not quantified yet	(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 the wastewater stream.	Energy	Combined heat and power units for recovered CH_4 have high conversion losses of ca. 60%	(Wan et al. 2016)
			COD may be too diluted for effective direct anaerobic digestion of wastewater.750 mg COD per litre is a medium concentration for municipal WWTP influents	(Logan and Rabaey 2012; Frijns et al. 2013)
			Dark fermentation of sludge shows very low H_2 yields of ca. 17%	(Lee et al. 2010)
		Fertilizer	Nutrient quantities recoverable from wastewater are low compared with industrial production rates. E.g. in Flanders (Belgium) yearly mined P imports amount of	(Kleerebezem et al. 2015; Coppens et al. 2016)

			44.100 tonnes while combined WWTP influent-P amounts only of 3.350 tonnes	
			Struvite: low P concentrations limit precipitation which requires at least 100 mg P per litre	(Zhang et al. 2013; Xie et al. 2016)
			Struvite: only soluble P fraction of side streams is recovered	(Wilfert et al. 2015)
			Low N concentrations of only 30 mg per litre NH_4-N in average Dutch wastewater may make NH_4 recovery uneconomical	(Kuntke et al. 2012; Khiewwijit et al. 2016)
		Other products	VFA concentration in wastewater and fermenter effluent may be too low for economical 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.	Fertilizer	Field application of sewage sludge: high water content (70-90%) and low nutrient content (7 kg P per tonne)	(Kirchmann et al. 2017)
			Possible contamination of struvite	(Lin et al. 2013; Xie et al. 2016)
		Other 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 and depends on pH, temperature, dilution rate, types of carbohydrates present and feeding patterns	(Kleerebezem et al. 2015)
			Uncertainty about whether mixed culture derived PHA from municipal wastewater can deliver reliable quality remains to be validated although pioneer pilot testing has been conducted with promising results	(Pratt et al. 2019)
Market value and competition	Conventional production systems potentially outcompete the RRR. This may be due to various factors, including higher product quality and quantities or lower production costs.	Energy	CH_4 has a low market value(EU-28 average 2019: 0,046 € per kWh for household consumers)	(Rabaey and Rozendal 2010; Kleerebezem et al. 2015; Eurostat 2019a)
			Electricity has a low market value (EU-28 average 2019:	(Puyol et al. 2017; Eurostat 2019b)

			0,22 € per kWh for household consumers)	
		Fertilizer	Bulk nutrients from the fertiliser industry are available cheaply (phosphate rock: 110 US$ per tonne in 2014)	(FAO 2015; Khiewwijit et al. 2016; Puyol et al. 2017)
			In livestock intensive regions P-rich manure is often abundantly available as an alternative fertiliser (see figure 2.4)	(Coppens et al. 2016)
			The market value of struvite is hard to estimate due to a lack of knowledge and trust of farmers into its fertilizing potential	(Le Corre et al. 2009)
		Other products	Petrol-based plastics may outcompete bioplastics and the latter are produced more economically from pure microbial cultures using sugar as feedstock instead from mixed microbial cultures applied to wastewater	(Tamis and van Loosdrecht 2015; van der Hoek et al. 2016)
			Finding real advantages of recovered biochemicals over fuel- or sugar-based alternatives to justify higher price of biodegradable/bio based plastics compared to conventional plastics (2,5 US$ per kg compared to 1,5 US$ per kg in 2014)	(Tamis and van Loosdrecht 2015; 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 an RRR successful.	Other products	Identifying niche markets (local or otherwise) and applications with unique selling propositions to increase competitiveness	(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. A 100 m vertical lift is as costly as a 100 km horizontal transport (0,05-0,06 US$ per m^3 in 2005)	(Zhou and Tol 2005; 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)
		Fertilizer	In-field sludge application: transport between WWTP and arable land might be too costly due to high water content	(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 due to insufficient process control.	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)
			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)
		Fertilizer	Struvite may be contaminated with emerging pollutants and heavy metals	(Lin et al. 2013; 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)

		Other 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	To be successful, RRRs need adequate policy and legal frameworks. 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 implement legislation and policies for water reuse	(Guest et al. 2009)
		Energy	Anaerobic digestion needs to be subsidised to become competitive with natural gas	(Kleerebezem et al. 2015)
		Fertilizer	Lack of legislation on in-field struvite application	(van der Hoek et al. 2016)
		Other products	Legislation forbids the use of protein produced from faecal substrate (in Europe)	(Alloul et al. 2018)

2.4.1. The role of water management utilities

Water management utilities (WMUs) could possibly ease or even eliminate the listed bottlenecks to successful RRR implementation by proactively planning resource recovery routes. However, WMUs may not be sufficiently influential to tackle all bottlenecks to the same extent (Figure 2.6). To reduce process costs, recover safe and environmentally benign products, or 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 decisions that are beyond the scope of technical process design. Similarly, the recovery of resources in competitive quantities can be managed actively. The volumes of recovered resources 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 to increase the 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 that are 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. though it is more difficult to leverage positive change at this level, a 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, thus making the most of synergies (Stanchev et al. 2017). In addition, cooperation between WMUs for example, joining forces to apply a common recovery strategy across multiple WWTPs and thus 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 therefore seek support from value-chain actors or political institutions to share the risks of innovation implementation (NSF et al. 2015).

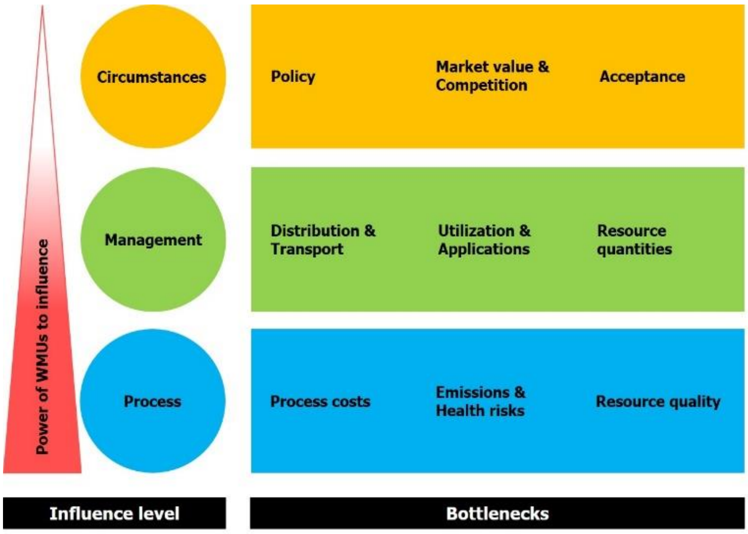

Figure 2.6 The power of water management utilities (WMUs) to influence identified bottlenecks for resource recovery technology implementation into WWTPs.

2.5. 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 2.1 shows that not all resource recovery routes (RRRs) can meet substantial shares of overall resource demands. The market potentials of recovered water, energy, fertilizer and other products depend on the volumes demanded, the quantities contained in wastewater streams 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. We hypothesise that those 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, fertilizer and other 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 that may hinder the successful integration of RRRs into wastewater treatment plants (WWTPs) (Table 2.2). 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 two to social (acceptance) and policy issues. It is unlikely that water management utilities (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 meet legal effluent standards.

To implement future water resource recovery facilities (WRFs) 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 abandon the paradigm of merely operating existing WWTPs and 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 policymakers, are integrated into the planning process, and applications with unique selling propositions are found. If a WMU plans the implementation of a technically feasible resource recovery technology, it is recommendable that it analyses in advance whether any of the non-technical bottlenecks presented in this review still need to be tackled. In the future, WMUs could develop a common recovery strategy for selected resources that coordinates efforts to exploit synergies and the advantages of economies of scale. If several WMUs recover the same resource, value-chain development could be facilitated by acting as one supplier, thus increasing their collective market power. This idea has already been put into practice in the Netherlands, where water boards have established the Energy and Raw Materials Factory (Energie en Grondstoffen Fabriek) to act as a collaborative network organisation coordinating the recovery efforts of several WMUs.

The most precious resource contained in municipal wastewater is the water itself. Unlike energy, which can be obtained from multiple sources, it has no alternative origin. Wastewater reuse can provide an important alternative source of fresh water in regions that expect lasting shortages in the future. It should preferably be promoted where it is less energy- and resource-demanding than conventional fresh-water treatment and distribution. It is possible that in the future, stricter effluent-quality regulations will require the elimination of emerging pollutants. Thus, 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 made 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 recover ten times more energy than the CH_4–CHP route and should therefore be considered more prominently in wastewater resource recovery planning to operate a plant carbon neutrally. Fertilizer 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 2.4. Ammonium recovery is only recommendable if the process consumes less energy than conventional ammonium production.

However, the supply potentials and bottlenecks presented in this paper should be perceived as challenges rather than obstacles. We believe that successfully implementing wastewater resource recovery requires the proactive management of potential bottlenecks and the finding of partners along a value chain to share the risks associated with pioneering. To achieve the transition from WWTPs to WRFs, resource recovery needs to be considered a strategic goal from the earliest process design and planning stages of new processes. Designing and implementing a WRF requires decisions in fields that are 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 WRFs to exploit their vast technological potential and to overcome non-technological bottlenecks.

3

Exploring resource recovery potentials for the aerobic granular sludge process by mass and energy balances

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“Engineers ... are not mere technicians and should not approve or lend their name to any project that does not promise to be beneficent to man and the advancement of civilization.”

John Fowler

3.1. Introduction

Wastewater has been recognised as a resource rather than a waste stream since over a decade now (Guest et al. 2009). It contains resources that can be recovered with a variety of technologies into reusable water, energy, fertilizers and other valuable products (Kehrein et al. 2020a). Recovering resources that can be produced in quantities and at costs that match the current market demand and prices (van Loosdrecht and Brdjanovic 2014) and/or tackle projected future resource scarcity (van der Hoek et al. 2016) enables the transition towards water resource factories (WRFs) instead of wastewater treatment plants (WWTPs).

The aerobic granular sludge process (AGS), also known as the NEREDA™ technology was successfully introduced globally at several full scale wastewater treatment plants in recent years and is considered more resource efficient than the conventional activated sludge process (CAS) (Pronk et al. 2015). Due to the granular sludge's excellent settling behaviour and tolerance to high MLSS concentrations the NEREDA process makes large settling tanks and low biomass concentrations in reaction tanks redundant compared to conventional biological treatment processes (Kreuk et al. 2005). The required surface area for AGS processes is therefore roughly 75% lower than for CAS processes that use flocculated sludge (van Loosdrecht and Brdjanovic 2014). Another advantage of the AGS over the CAS process is biological P removal that requires almost no additions of chemicals as phosphate precipitants. Furthermore, there is no need for energy intensive recirculation of flows between anaerobic, anoxic and aerobic tanks within the plant because these conditions are all established simultaneously in the different layers of each single granule (Van der Roest 2011). The higher efficiencies in land consumption, energy and chemical use lead to approximately 25% cost reduction of the AGS process compared to CAS processes (van Loosdrecht and Brdjanovic 2014).

The AGS process is not only promising in terms of resource and cost efficiency but offers an innovative possibility for product recovery from COD. Extracellular polymeric substances (EPS) are produced by microorganisms to form a hydrogel matrix as a dense network that gives the granular microbial structures their physical stability. EPS are a complex mixture, consisting of polysaccharides, proteins, nucleic acids, (phosphor)lipids, humic substances and some intercellular polymers. They are considered useful polymers for industrial applications as they show unique material properties especially when used as composite material (Felz et al. 2016). Biopolymer like materials cannot, in general, be derived from oil-based chemicals and hence their supply relies solely on natural resources (Seviour et al. 2019). In the Netherlands, a commercial company currently develops a value chain from EPS recovered from AGS to market the polymer under the product name "Kaumera™ Gum". A full-scale EPS recovery and down streaming plant has been opened in 2019 in the WWTP of the City of Zutphen (Knaap et al. 2019).

With growing concerns over climate change, energy saving, energy efficiency and energy substitution have become a common development principle all over the world and are manifested in the 7th UN development goal for more affordable and clean energy (United Nations 2017). The wastewater sector including academia responds to that goal with exploring energy self-sufficient WWTP designs that reduce net-energy consumption and therefore may achieve carbon neutrality and decrease operational costs (Gu et al. 2017). Biogas recoverable from primary sludge and/or waste activated sludge by anaerobic digestion (AD) and subsequent combined heat and power (CHP) generation is the most widely applied energy recovery route in WWTPs (Verstraete et al. 2009). Biogas production is an established technology for a variety of organic waste streams with growing implementation worldwide. Compared to other feedstock, sewage sludge leads with ca. 60-70% to a fairly high methane content (Lee et al. 2013; Bachmann et al. 2015; Tyagi and Lo 2016). It has been shown that combining AD with up-concentration of primary COD in a chemically enhanced primary treatment unit (CEPT) could increase methane recovery while simultaneously minimizing aeration energy consumption in aerobic treatment units, and thus, may reduce greenhouse gas (GHG) emissions and operational costs of WWTPs (Wan et al. 2016). In addition to on-site energy recovery, anaerobic sludge digestion also serves the purpose of waste sludge stabilisation (Verstraete et al. 2009) which is an important method to decrease waste sludge quantities and thus waste management costs of WWTPs (Molinos-Senante et al. 2010; Kleerebezem et al. 2015).

In addition to energy recovery, phosphorous (P) recovery is another research topic experiencing high interest in academia already since decades and numerous technological recovery routes have been developed (Egle et al. 2016). Efforts to recover P have been recently intensified as various EU member states including Germany, introduced legislation to enforce P recovery from WWTPs in the near future (Günther et al. 2018). Currently, fractions of the influent-P are predominantly recovered by two technological solutions, namely chemical P extraction from sludge incineration-ash and as struvite mineral from P-rich side streams in WWTPs that apply biological P removal (Wilfert et al. 2015). Struvite precipitation in WWTPs was discovered almost 60 years ago and its removal served initially the purpose of improving plant operations and especially clogging of pipes and equipment (Zhang et al. 2013). Because it contains both ammonia and ortho-P the mineral can be used as a slow release fertiliser applicable to crops in soils with low pH value and is therefore an interesting mineral for recovery (Xie et al. 2016). The recovery of struvite by controlled crystallization requires a side stream that contains concentrated orthophosphate and ammonia. Anaerobic sludge digestion plays therefore a key role in struvite recovery as it re-immobilizes fractions of both nutrients (Batstone and Jensen 2011).

WWTPs are usually designed according to two major criteria: treatment performance to meet legal effluent standards with a reliable robustness on the one hand and cost effectiveness on the other hand. It has been postulated that resource recovery is missing as an additional but integral part in early-stage process design decisions (Guest et al. 2009; Hamouda et al. 2009; Bozkurt et al. 2017). Although the paradigm shift from treatment towards resource recovery and the inclusion of the latter as a central objective in new process designs has been claimed in literature since a decade, little resource recovery technologies seem to have experienced wide implementation yet (NSF et al. 2015).

One important reason is the still constantly growing range of technical possibilities and therefore process design complexity increases. More and more innovative treatment and resource recovery technologies become available but little is known about how to integrate and combine them effectively (Bozkurt 2015; Batstone et al. 2015; Li et al. 2015; Khiewwijit et al. 2016). One aspect that needs enhanced understanding to design WRFs in the future is that the integration of a resource recovery technology into a process likely implies trade-offs with one or more other possible resource recovery technologies, as influent constituents, like e.g. P and COD can only be recovered once (van der Hoek et al. 2016). For example, COD recovery as biogas that is combusted to recover electricity and heat may reduce the potential to recover a non-energy carrier product from COD, like e.g. biopolymers including EPS (Kleerebezem et al. 2015). Therefore, to make rational decisions in future WRF process design it is crucial to compare a spectrum of technically possible process designs potentially applicable to a site of investigation at an early design stage (Lee et al. 2013). To identify the most feasible process design among possible alternatives requires careful analysis. The quantification of potentially recoverable resources and trade-offs between alternative recovery options enhances the understanding of resource recovery technology integration into a treatment process and therefore can support decision making in WRF design. In addition, estimating how much of a certain product could potentially be recovered on-site of a WRF is essential to create value chains and market the product successfully because too little quantities may not be competitive on relevant markets(Kleerebezem et al. 2015; Chong et al. 2016).

This can be achieved through mass and energy balances (MEBs) which allow to model how wastewater constituents are converted in a particular treatment process (Solon et al. 2019b). In comparison to each other they provide insight on how integrated recovery technologies may influence each other in terms of recoverable products from a process. Therefore, the comparison based on MEBs of different process designs potentially applicable to a municipal wastewater stream helps to find the process that is most feasible from a resource recovery perspective (Solon et al. 2017). Since MEBs reveal the concentrations of selected constituents in all streams within a process, also the effluent quality that a process produces can be estimated from the results.

However, optimal resource recovery technology integration always depends on site specific circumstances of the treatment process because influent compositions, treatment technologies, or local market conditions for recovered resources may vary greatly (Wang et al. 2015a; Solon et al. 2019b). Therefore, this analysis exemplifies selected resource recovery potentials and trade-offs along the case of the Utrecht WWTP in The Netherlands. The plant uses the AGS treatment process and was designed without any resource recovery technology integration. We conducted a MEB for the process and compared it to 5 different theoretical process designs that would recover COD and P on-site at this plant. COD would be recovered either as energy derived from biogas (electricity/heat from methane combustion) or as EPS. P would be recovered on-site the plant as struvite mineral. This way recoverable quantities of these products and trade-offs between certain design choices are revealed. Consequently, the study at hand contributes to the transition towards WRFs by providing insights on potentials of COD and P recovery integration into AGS-based WWTPs. It aims to answer the question: how to integrate COD and P recovery technologies into an AGS treatment process to improve decision making in AGS based WRF design in the future.

3.2. Materials and methods

3.2.1. Current process and general assumptions

The current process is based on a WWTP in Utrecht (The Netherlands) which operates six aerobic granular sludge reactors in parallel, each with a capacity of 12 thousand m^3 to treat wastewater of 308 thousand person equivalents in total. The plant recovers no resources on-site, but only off-site at an external sludge incinerator that recovers COD as electricity and P from the leftover ash fraction (ash-P). In the first treatment step, the influent is screened for coarse materials that may cause operational problems before it enters the biological treatment stage. During the following AGS treatment aerobic and anaerobic conversions of constituents take place simultaneously in a three-step cycle of anaerobic fill and draw, aeration and settling. For operational details of an AGS treatment plant we refer to (Pronk et al. 2015). After the biological treatment, the surplus granular sludge is thickened by gravity. Data to model the plant wide conversions of measured influent constituents in different process unit operations have been obtained from literature. A detailed list of parameters applied at each operational unit is provided in the appendices. Modelled process designs and corresponding COD and P flows are depicted by Sankey diagrams where flow sizes are proportional to the influent concentration.

For simplification reasons, several general assumptions have been made for different unit operations and we want to highlight the most important ones: Influent COD fractionation values for biodegradable (easily, slowly, inert) and soluble and particulate COD fractions vary considerably in literature and are site specific (Roeleveld and van Loosdrecht 2002; Pasztor et al. 2009; Hartley 2013). They have not been measured for the influent and therefore COD conversions are modelled using the total COD value. The energy content of COD is assumed to be 17,8 kJ g^{-1} (Heidrich et al. 2011). The coarse screen, also known as grit removal, is modelled to have no impact on COD and P flows although it has been shown in literature that COD containing solids may be removed (Riffat 2013). The same applies for the gravity thickening of surplus sludge. Although the excess water flow from sludge thickening is usually redirected into the process (Appels et al. 2008) and likely contains minor very minor COD and nutrient fractions, those have not been modelled. The energy conversion efficiency of the sludge incineration unit is assumed to be 40% (electric efficiency) (Faaij 2006). Moreover, it is assumed that waste sludge arrives at the incinerator with a dry solids (DS) content of 22% which represents the Dutch average. This implies that over 70% of the sludge COD energy content is needed to evaporate the water until a sludge DS content is reached that is energy positive (Frijns et al. 2013).

Oxygen requirements in the NEREDA reactor are determined by COD oxidation and nitrification. The aeration phase is modelled with a 60% COD oxidation where organics are oxidized to CO_2 (Winkler et al. 2013). Total Kjeldahl nitrogen (TKN) inflows into the AGS reactor likely consists to ca. 70% of ammonia while the rest is organic N (Hartley 2013). The latter will almost entirely be converted into ammonia during the treatment process (Makinia et al. 2011). Consequently the TKN load to the NEREDA reactor consists almost entirely of ammonia which is removed with a rate of over 90% (de Kreuk et al. 2005). Ca. 20% of the removed TKN ends up in the sludge and the rest is converted to nitrogen gas via nitrification-denitrification. The nitrification needs to be accounted for in the estimation of oxygen requirements. The denitrification replaces oxygen for COD removal with a stoichiometry a 2,86 g oxygen per g NO_3-N which lowers the total oxygen requirements.

The recoverable resource quantities are sensitive to the various assumptions made for the different parameters applied in the mass and energy balances. Therefore, a sensitivity analysis has been conducted on those parameters that are reported in literature with a certain degree of uncertainty. It reveals which unit operations need to be optimized to increase yields of a resource recovery pathway and influence its trade-off to another pathway. Parameters most sensitive to recoverable resource quantities are discussed in the results section whereas the detailed results of the sensitivity analysis are available in the supplementary information.

3.2.2. Modelled process designs for on-site resource recovery

As explained above, the current process does not recover any resources on site but only off-site from sludge incineration where COD is recovered as electricity and P from incineration ashes. To explore how COD and P could be recovered on-site the treatment plant, the current process was theoretically re-designed into five different configurations table 3.1.

Table 3.1 Overview of modelled process designs analysed by mass and energy balances.

Site of recovery		***On-site recovery***		
Resource		***COD***		***P***
Recovered product		***Electricity/heat***	***EPS***	***Struvite***
Process unit		***AD/CHP***	***Chemical extraction***	***Crystallization***
Designs	***Current***	·	·	·
	AD	✓	·	✓
	AD/CEPT	✓	·	✓
	EPS	·	✓	·
	AD+EPS	✓	✓	✓
	AD/CEPT+EPS	✓	✓	✓

On-site COD recovery is integrated by anaerobic digestion (AD) and subsequent combined heat and power (CHP) recovery from obtained methane combustion. The total-COD into biogas conversion rate is assumed to be 50% as measured by (Khiewwijit et al. 2016) under mesophilic conditions. The digestate is handled in a decanter centrifuge from which a liquid supernatant stream is redirected into the AGS reactor as commonly seen in WWTPs (Sode et al. 2013). AD has been modelled in combination with/and without integration of polymer based CEPT which diverts primary COD into AD and therefore represents a process for maximized on-site energy recovery. In addition to AD and CHP integration for energy recovery, the second on-site COD recovery integration is EPS extraction from surplus granular sludge. The granular sludge harvested from the AGS reactor consists to 20% of EPS (van der Roest et al. 2015) and the EPS downstream processing is assumed to have no losses.

In addition to COD, also P can be recovered on-site the plant as struvite which can be precipitated from the liquid supernatant that is produced during sludge centrifugation. It is assumed that total-P (TP) in the influent consists of two thirds of solubilized ortho-P and one third of P bound to organics (Henze 2008). Most organic-P is modelled to be converted into ortho-P during the biological treatment (Krishnaswamy et al. 2011) and therefore can be potentially recovered as struvite afterwards. Ortho-P is reactive and may precipitate due to the presence of Fe, Ca, or Al in the influent. Ortho-P binding to inorganic substances will strongly depend on local conditions, like e.g. pH, temperature, mineral and P concentrations present in treatment reactors (Kreuk et al. 2005). For simplification reasons, the model accounts only for ortho-P bound to Fe because it has been shown that it is likely to be present in Dutch influents with an average concentration of 1 mg l^{-1}. Furthermore, is expected that all of this Fe is divalent and therefore leads to vivianite ($Fe^{2+}_3(PO_4)_2 \cdot 8H_2O$) formation. The ortho-P fraction precipitated as vivianite is therefore not recoverable as struvite later in the process (Wilfert et al. 2015). Following these assumptions and stoichiometry, 1 mg of divalent Fe present in the influent would bind 1,1 mg of ortho-P that is not available for recovery anymore except as vivianite. However, most ortho-P is accumulated in biomass during biological treatment before it is re-mobilized during AD (Batstone and Jensen 2011) and therefore, the liquid supernatant produced in the decanter centrifuge for digestate handling is rich in solubilized P and ideal for struvite crystallization (Mehta et al. 2015).

3.3. Results and discussion

In the following paragraphs the results of modelled process designs are discussed regarding their resource recovery potentials and trade-offs between resource recovery technology integrations. Despite the growing importance of resource recovery in wastewater treatment the production of clean water for environmentally safe discharge remains also the major objective of WRFs. Therefore it is important to state that all modelled process designs would meet Dutch legal effluent concentrations of COD, P-total, and TKN (see also appendice). Table 3.2 summarizes the results of modelled process designs expressed as influent-COD and influent-P recovery rates.

Table 3.2 Summary of influent-COD and influent-P recovery rates of modelled process designs in %.

Influent-COD recovery rates %						
	Current	**AD**	**AD/CEPT**	**EPS**	**AD+EPS**	**AD/CEPT+EPS**
Electricity (on-site)	-	6,9	14,7	-	5,1	13,9
Heat (on-site)	-	7,3	15,5	-	5,4	14,6
EPS	-	-	-	8,8	9,0	4,1
Electricity (off-site)	4,1	1,7	3,5	3,0	1,2	3,3
Total	***4,1***	***15,9***	***33,7***	***11,8***	***20,8***	***35,9***
Influent-P recovery rates %						
Ash-P	73,0	65,0	67,5	58,8	52,0	61,5
Struvite-P	-	9,3	9,6	-	7,3	8,7
Total	***73,0***	***74,2***	***77,1***	***58,8***	***59,3***	***70,3***

3.3.1. On-site COD recovery as energy

Figure 3.1 (A) shows COD flows in the current process with an off-site electricity recovery potential from sludge incineration above 9 MWh d^{-1}. The integration of AD and CHP allows to recover a fraction of the influent-COD on-site as electricity and heat. The recovery of energy from COD on-site implies a trade-off with off-site electricity recovery of -60% (figure 3.1 (B)). The recovered electricity can directly be consumed on-site the plant to reduce its energy consumption from the grid or can be supplied to it for off-site usage which would imply some losses through e.g. the Joule effect in transformers and power lines (Gu et al. 2017). The recovered heat can be used on-site for different purposes, like e.g. for heating of the anaerobic digester or for waste sludge drying (Hao et al. 2019). In addition to on-site energy recovery, the integration of AD serves also the purpose of waste sludge volume reduction which can be expected to be in the range of 30-50% (Lee et al. 2011). Therefore AD integration likely decreases costs for sludge transport to the incinerator. If a CEPT unit is integrated additionally the on-site energy recovery can be more than doubled compared to only AD integration (figure 3.1(C)). To ensure a sufficient denitrification in the AGS reactor it is important to maintain a high enough biodegradable-COD (bCOD) which is according to the model over 4 gbCOD/gN, even in those designs with CEPT integration where a large COD fraction is removed before AGS treatment.

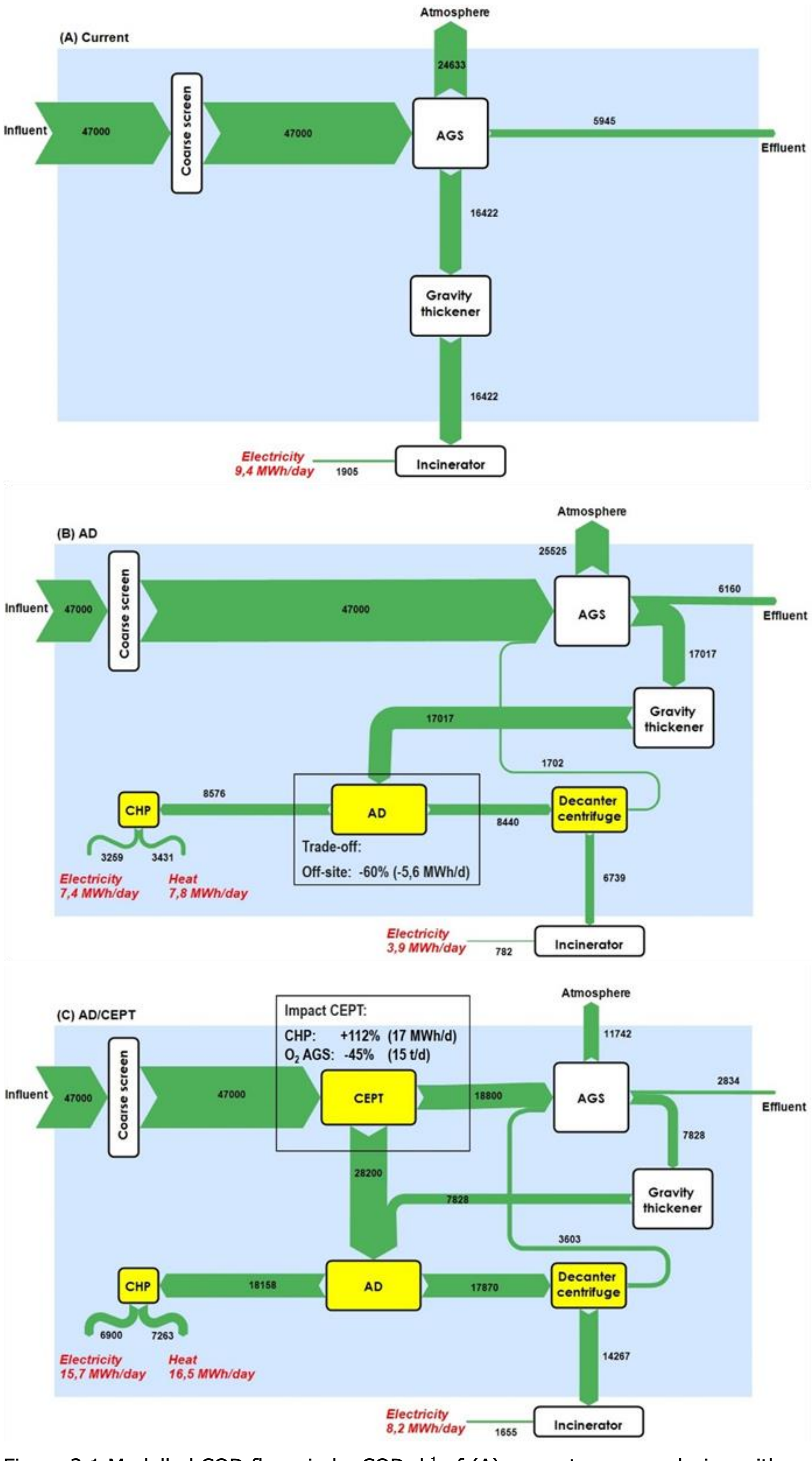

Figure 3.1 Modelled COD flows in kg COD d^{-1} of (A) current process design with no on-site resource recovery, (B) AD and CHP integration to recover energy on-site, (C) CEPT integration for maximum energy recovery on-site. Yellow unit operations represent integrated units compared to current process design. Recoverable product quantities in red.

CEPT integration increases not only the energy recovery potential on-site but also off-site compared to AD integration alone. This is due to the fact that much less COD is oxidised to CO_2 in the AGS reactor when CEPT is integrated and more COD enters the AD. Since AD converts the COD only up to 50% to biogas (Khiewwijit et al. 2016), relatively more influent-COD enters the incinerator as sludge if CEPT is integrated compared to AD integration alone. Thus, the total influent-COD recovery rate (on-site + off-site) can be doubled with CEPT integration as elucidated in (figure 3.2).

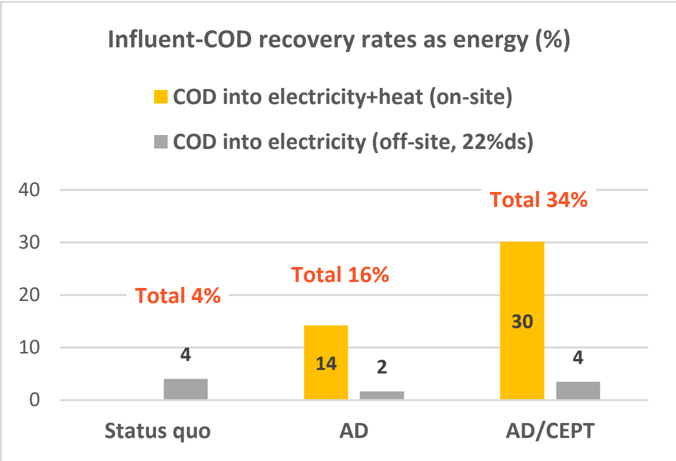

Figure 3.2 Influent COD recovery rates as energy on-site, off-site and in total.

The total-COD into biogas conversion rate of 50% is generally assumed disregarding the loads of waste activated sludge (WAS) or primary sludge (PS) into AD. In reality, the COD into biogas conversion rate might differ slightly depending on whether WAS or a mix of WAS and PS is loaded to AD because PS has a higher anaerobic COD biodegradability than WAS. The COD biodegradability of the latter largely depends on the solids retention time (SRT) in the aerobic treatment. The higher the SRT, the more recalcitrant the WAS is to further biological degradation and thus, the lower the conversion into biogas may be (Parker 2005). Being aware of this, the model may slightly overestimate the biogas yield in the "AD" design. The sensitivity analysis confirms that the total-COD into biogas conversion rate has a high impact on recoverable energy quantities on- site. This is also valid for the assumed methane content of biogas of 65% which has been reported in several studies as a reasonable assumption for sewage sludge digestion (Andreoli et al. 2007; Appels et al. 2008; Tyagi and Lo 2016). Nevertheless, it is important to keep in mind that this value assumes steady and well-managed anaerobic sludge digester systems and can be lower due to, for example, high temperature variations or overloads (Bachmann et al. 2015). Another assumption that is sensitive to the obtained energy on-site is the electricity/heat conversion efficiency of the CHP unit. The present study assumes a rather conservative efficiency of ca. 40% but depending on the size of the unit and its age there are increased efficiencies likely to be obtainable (Bachmann et al. 2015). Finally, the COD removal rate of the CEPT has a significant effect on the energy that is recoverable from the biogas pathway and therefore keeping the assumed 60% COD removal rate steady is necessary for a sustainable on-site energy recovery strategy.

In addition to a higher on-site energy recovery potential, another favourable effect of CEPT integration is a decreased oxygen requirement in the NEREDA reactor by -45% due to lower COD and TKN loads (figure 3.1 (C)). On the first glance CEPT integration may be promising in lowering GHG emissions of a process because it increases the methane recovery potential and decreases aeration. Nevertheless, both allegedly positive environmental effects can be severely offset by the necessary consumption of polymers that imply negative environmental impacts like abiotic depletion of elements and fossil fuel resources. This is also valid for cost-benefit calculations of CEPT units as polymers represent an additional cost factor (Solon et al. 2019b). Furthermore, also AD integration can lead to severe hidden direct CH_4 emissions which may even exceed emissions avoided through energy recovery from biogas combustion (Daelman et al. 2013). In addition, the integration of AD-based energy recovery technologies leads to higher operational costs as integrated units need regular maintenance like e.g. the CHP unit needs to be cleaned frequently from deposits that deteriorate its efficiency (Bachmann et al. 2015). Therefore, the presented results of AD and CEPT integration should be interpreted as an intermediate step towards a more complete process assessment including additional technical, economic, and environmental performance criteria.

Another challenge is that the off-site energy recovery potential from sludge incineration is sensitive to the assumed water content of the sludge. This study assumes the Dutch average value of 22% DS content for waste sludge which implies high energy recovery losses in the incinerator due to water evaporation requirements. This finally allows to recover less than one third of the energy loaded into the incinerator in form of organic matter (Frijns et al. 2013). Due to various possible sludge dewatering processes, each resulting in different water contents, DS content of waste sludge is reported with great variations in literature. Because the sensitivity of the assumed DS content to the recoverable electricity from sludge incineration is high, it needs to be adjusted for site specific circumstances before one aims to transfer the results of this study to another case. Another sensitive assumption in the off-site electricity recovery pathway is the electrical efficiency of the incinerator. It is assumed that 40% of the loaded dry solids that remain after evaporation energy subtraction are converted into electricity. But depending on the type and age of the incinerator this efficiency may also be lower (Faaij 2006; Oh et al. 2010).

It is also likely that surplus heat from the incineration process can be used to further increase the DS content of the sludge before incineration (Lederer and Rechberger 2010). Therefore, the estimated off-site energy recovery potential from the incinerator might be an underestimation and is strongly dependent on site specific conditions.

3.3.2. On-site COD recovery as EPS

Energy recoverable from COD has often been referred to as the second most valuable resource in wastewater after water (Lee et al. 2013). More recently it has been argued that, following the principles of circular economy and sustainability, COD should be preferably recovered as materials than as energy. This argumentation follows two reasons. First, COD in wastewater contains only a small fraction of the energy because thermal energy recoverable by water sourced heat pumps contains more energy and therefore has a higher potential to save carbon emissions. Secondly, COD has a relatively large exergy content which should be preserved thus converted into carbonaceous materials (Hao et al. 2019). The integration of EPS recovery from surplus granular sludge serves this rational. If EPS extraction would be integrated into the current process, more than 3t of EPS product could be recovered daily, assuming no losses in polymer downstream processing (figure 3.3 (A)). In combination with AD integration a trade-off between EPS and energy recovery occurs due to a fraction of COD leaving the system as EPS which is therefore not fed to AD. This is reflected in a decreased on-site energy recovery potential of -26% compared to no EPS recovery integration (figure 3.3 (B)). Integrating CEPT additionally to maximize on-site energy recovery implies less COD is fed to the granular sludge which essentially produces the EPS. Consequently, CEPT integration implies that the EPS recovery potential is halved while on-site energy recovery is more than doubled compared to only AD integration (figure 3.3(C)).

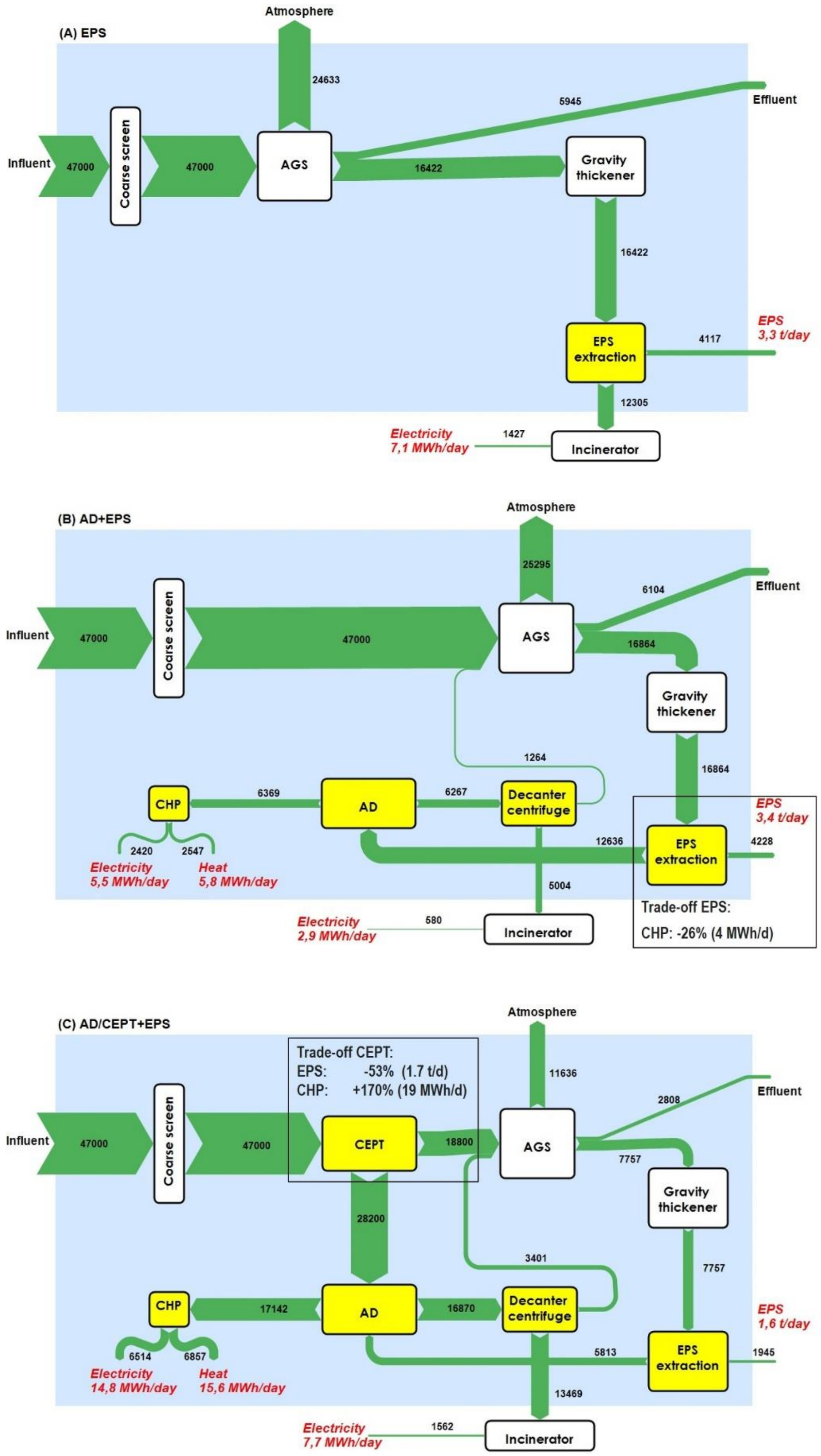

Figure 3.3 Modelled COD flows in kg COD d-1 of (A) EPS recovery integration, (B) AD and CHP integration to recover energy on-site and EPS recovery integration (C) CEPT integration for maximum energy recovery on-site and EPS recovery integration. Yellow unit operations represent integrated units compared to current process design. Recoverable product quantities in red.

When AD is integrated in combination with CEPT, EPS recovery does not necessarily imply a significantly decreased on-site energy recovery potential in comparison to AD integration alone. The difference in energy recovery of the designs "AD/CEPT" and "AD/CEPT+EPS" is just marginal. This is due to the high on-site energy yields obtained with CEPT integration. The elevated energy yields overrule by far the energy losses resulting from COD leaving the process as EPS before it can enter AD. In total, up to one third of influent-COD, which equals ca. 15 t COD per day could be recovered on-site in form of energy and/or EPS with the examined COD recovery technology integration (figure 3.4).

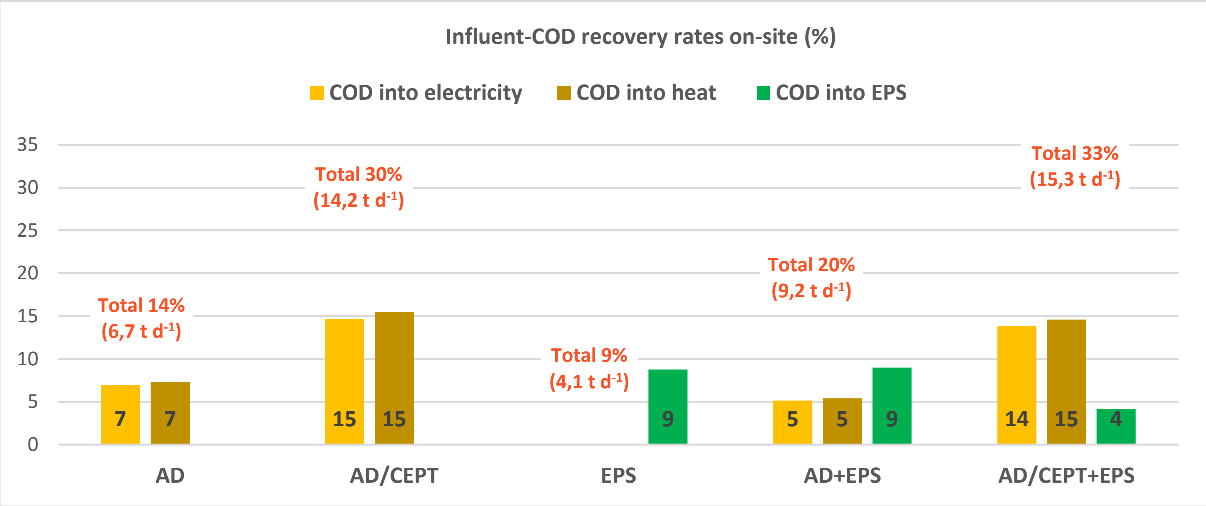

Figure 3.4 On-site influent-COD recovery rate of analysed process designs in % and in total values (tonnes per day).

It should be mentioned that the extraction of EPS from the sludge requires chemicals as inputs (Felz et al. 2016) which need to be accounted for in subsequent economic and environmental impact assessments of those designs which integrate EPS recovery. Furthermore, alkaline pre-treatment of sludge prior to AD has been reported to increase the anaerobic degradability of organics (Li et al. 2012) and possibly biogas yields (Fang et al. 2014). Since EPS is extracted from the granular sludge with an alkaline process followed by an acid precipitation step (Felz et al. 2019), its recovery leaves an alkaline waste stream and therefore may function simultaneously as an effective sludge pre-treatment step. Due to this pre-treatment-like function EPS extraction could lead to increased biogas yields which would reduce the trade-off between EPS and AD-based energy recovery. Maintaining the high pH after alkaline sludge disintegration and applying it also in an alkaline AD may turn the fact that EPS extraction produces a haloalkaline waste stream into another advantage. A few preliminary studies produced biogas with haloalkaline microbial consortia obtained from soda lake sediments (van Leerdam et al. 2008; Nolla-Ardèvol et al. 2015). Since the CO_2 produced during AD remains solubilized in the broth under alkaline conditions, one of the main advantages of alkaline AD applied to sludge waste from EPS extraction would be a methane rich biogas stream (>95%) that can be used directly as a fuel which would make expensive biogas upgrading redundant (Sels 2019). Nevertheless, whether a large scale alkaline AD is applicable and whether the methane yields of such a system are comparable to the ones of lab scale experiments, remains uncertain.

Finally, it should be mentioned that recoverable EPS quantities are sensitive to several assumptions made in the model. Obviously, as already discussed above EPS recovery can be significantly lowered the more COD is removed in the CEPT unit. Another sensitive parameter is the assumed EPS downstream yield of 100% which is likely to be lower in reality and has been chosen based on lab scale experiments because data from large scale applications are yet missing. Also the assumed parameters related to COD conversions in the AGS reactor determine the EPS recovery significantly. The sensitivity analysis reveals that the assumed total COD removal rate, the fraction of removed COD ending up in the sludge, and the EPS content of the sludge are all positively correlated to the recoverable EPS quantities. This should be kept in mind when transferring the results of this study to other cases where COD conversions taking place in an AGS reactor may differ due to site specific factors. For example, the easily biodegradable COD fraction of the influent influences recoverable EPS quantities and may differ greatly between cases. A key aspect to consider in EPS recovery estimation based on total COD instead of COD fractions is the pre-treatment applied before the AGS reactor as different pre-treatments may retain different rates of slowly biodegradable COD. Especially cellulose may play a vital role in how much of the total COD is converted into EPS because depending on the operational settings it may be hardly degraded during aerobic treatment (Ruiken et al. 2013) but still accounted for in a model like the one presented in this study. To sum it up, modelling the mass conversions of particular COD fractions instead of total COD may further improve the accuracy of the EPS recovery pathway.

3.3.3. On-site P recovery as struvite

Figure 3.5 (A) shows P flows in the current process in which almost all P ends up in the sludge which is then incinerated off-site. After incineration ca. 400 kg P per day can be recovered from the ash fraction. If AD or AD/CEPT is integrated, on-site P recovery of ca. 50 kg struvite-P per day from digestate supernatant becomes possible (figure 3.5 (B) and (C)). The polymer based coagulation applied in the CEPT leads to a much higher accumulation of influent-P in the obtained primary sludge compared to normal primary settling (Klute and Hahn 1994). Consequently the integration of CEPT into a process design means that a significantly higher influent-P fraction enters AD instead of the AGS reactor which may also lower P effluent concentration significantly.

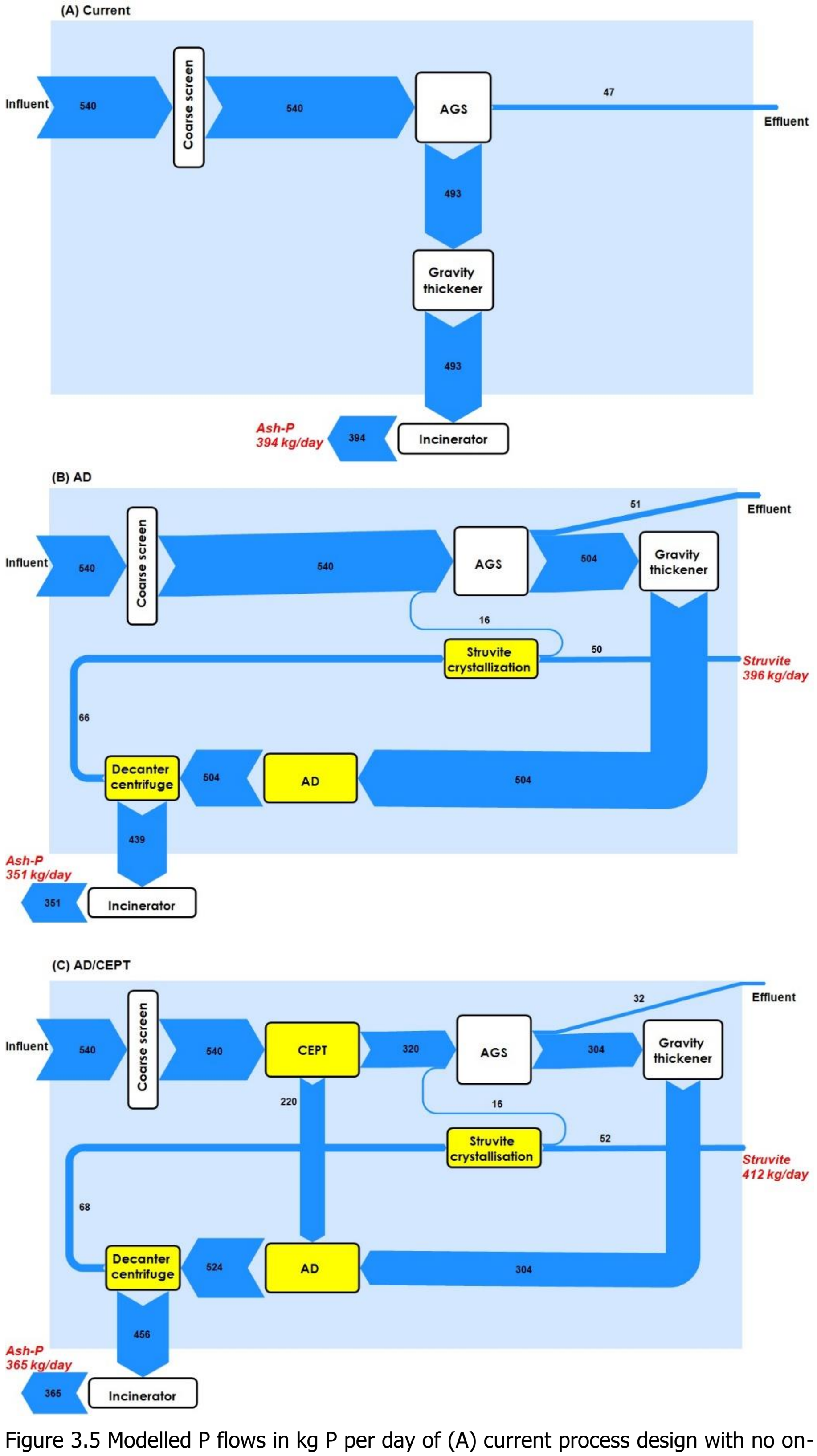

Figure 3.5 Modelled P flows in kg P per day of (A) current process design with no on-site resource recovery, (B) AD and CHP integration and subsequent struvite recovery on-site, (C) CEPT integration and subsequent struvite recovery on-site. Yellow unit operations represent integrated units compared to current process design. Recoverable product quantities in red.

Figure 3.6 shows that struvite recovery integration increases the total influent-P recovery rates only very minimally because recoverable quantities of struvite-P are very small compared to ash-P quantities. The reason is that only a small fraction of total-P loaded to AD can be expected to end up in the supernatant as soluble ortho-P because 80-90% of loaded P remains in the digestate (Khiewwijit et al. 2016). In addition, 80% of this soluble ortho-P can be recovered as struvite crystals (Martí et al. 2010). This leads to overall low influent-P recovery rates in the form of struvite-P. In addition, for successful struvite crystallization the molecular ratio of magnesium, ammonia and ortho-P (Mg^{2+}:NH_4^+:PO_4^{3-}) should be 1:1:1 (Verstraete et al. 2009). This leads to the required mass of at least 0,8 kg Magnesium per 1 kg P precipitated which represents another cost and environmental factor to be considered for the struvite recovery pathway. For these reasons, it is questionable to invest in a struvite crystallization unit if ash-P recovery from sludge incineration is possible because most influent-P ends up in the sludge and is not available for struvite crystallization.

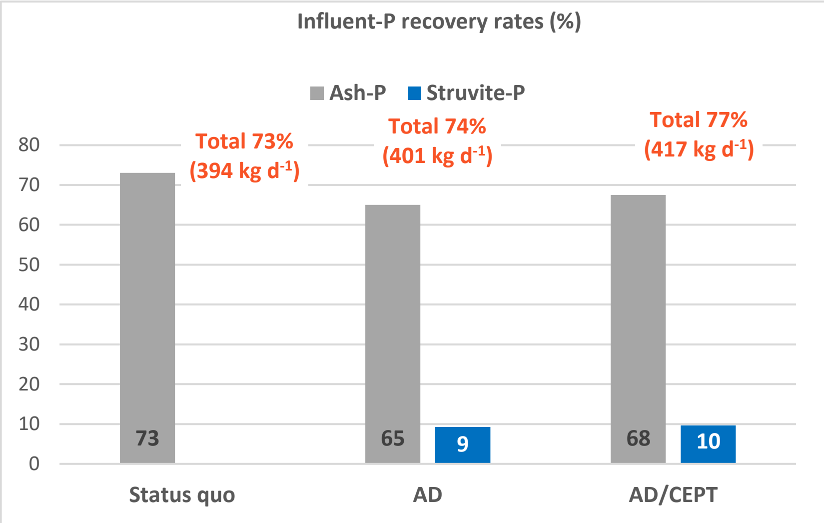

Figure 3.6 Recoverable influent-P rates in % and total P recoverable also in kg per day.

Struvite is a mineral and following its stoichiometry it contains only ca. 13% of P while the rest of its mass is ammonia, magnesium and crystal water ($(NH_4)Mg[PO_4]\cdot 6H_2O$). Considering its total mass, recoverable struvite masses are comparable to recoverable ash-P masses (figure 3.7). One can therefore argue that struvite is equally promising to be marketed as a fertilizer product because it can be supplied in comparable quantities as ash derived P. Since it can be recovered on-site, it may generate revenues for the water utility operating the plant while ash-P revenues are generated externally at the incinerator. In addition struvite recovery may prevent pipe clogging and therefore decrease operational costs (Zhang et al. 2013). The recovery of struvite from digestate supernatant is mostly sensitive to two process parameters: Firstly, the assumed fraction of P that ends up in the liquid supernatant after centrifuging the digestate (assumed to be 13%), and secondly, the crystallization rate of ortho-P during struvite formation (assumed to be 80%). Although assumed values for both mechanisms have been reported in literature (Martí et al. 2010; Khiewwijit et al. 2016), there is more research needed to reveal how both parameters can be enhanced in the future to increase struvite precipitation from side streams.

P recovery from sludge incineration ashes requires the realization of dedicated sewage sludge incinerators that are expensive to build (Wilfert et al. 2018). Still, from an overall societal P recovery perspective, the ash-P recovery route has clear advantages because it can bundle the excess P streams from several WWTPs. A standalone sludge incinerator can therefore function as a centralized P recovery unit in a region and may make use of economy of scale to recover high rates of regionally consumed P. There are yet various uncertainties on how to optimize the ash-P recovery route. There is little research published on P recovery rates from sludge incineration ashes and the question remains to what extend the assumed 80% (Lundin et al. 2004) can be further increased as it is a sensitive value to the results of this study.

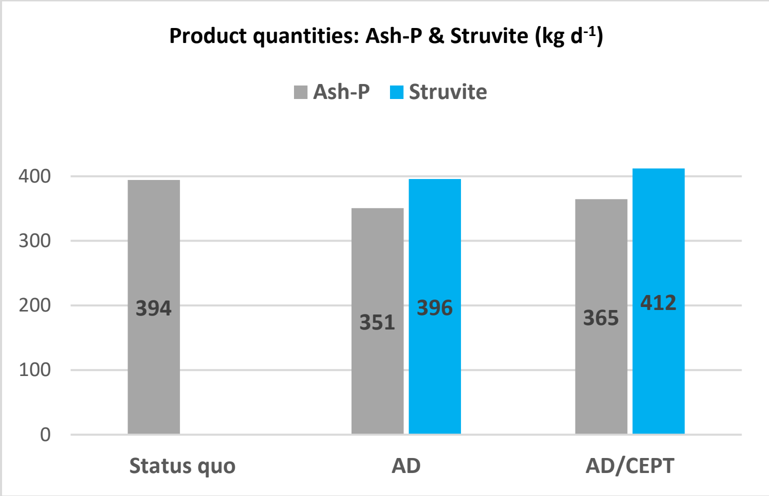

Figure 3.7 Recoverable P-fertilizer product quantities in kg per day.

1 t EPS contains ca. 30 kg P and therefore a trade-off between EPS and P recovery exists leading to ca. -20% decreased ash-P and struvite recovery potential in those designs with EPS recovery integration compared to those with none (figure 3.8 (A)). Obviously, if CEPT is applied the consequential lowered EPS recovery potential decreases this trade-off as less P is incorporated into EPS polymers and leaves the system as such (figure 3.8 (B)). While P-fertilizers are relatively cheap (Puyol et al. 2017), EPS are a potentially high value product (van der Roest et al. 2015). Therefore it is reasonable to argue that P recovery as EPS may be economically favourable over P recovery as fertilizer. But a reliable statement can only be made if EPS becomes an established product on biopolymer markets and a complete cost-benefit analysis of the investigated process designs is carried out considering site specific circumstances.

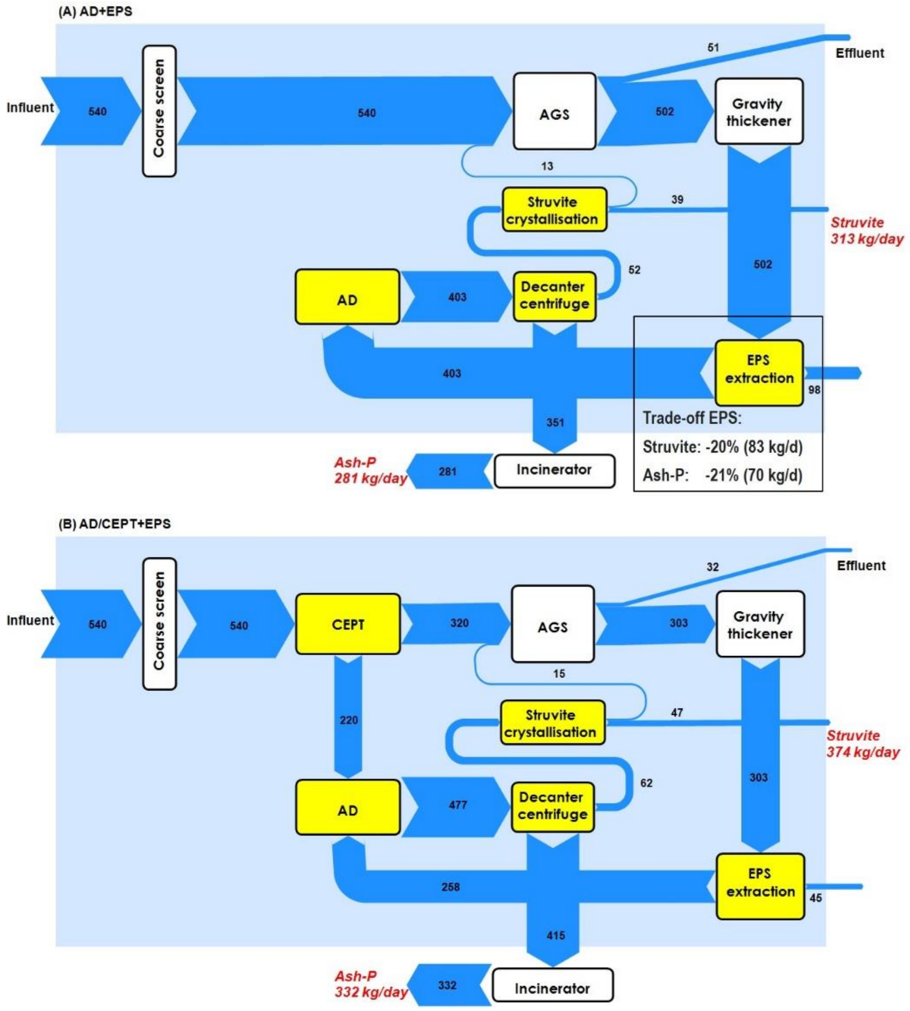

Figure 3.8 Modelled P flows in kg P per day of (A) AD+EPS design (B) AD/CEPT+EPS design. Yellow unit operations represent integrated units compared to current process design. Recoverable product quantities in red.

3.4. Conclusion

The results show how to integrate COD and P recovery technologies into an aerobic granular sludge (AGS) treatment process and therefore improve decision making in AGS based water resource factory (WRF) design in the future. Mass and energy balances allow to quantify recoverable products as well as to analyse trade-offs occurring from possible design choices and are therefore an important tool in WRF process design. Significant quantities of resources can be recovered on-site of existing aerobic granular sludge treatment plants. Regarding AD integration, the results of this study confirm previous studies who state that the fraction of influent COD recoverable by AD and CHP integration is rather low (Schwarzenbeck et al. 2008; McCarty et al. 2011). Nevertheless, the integration of AD for on-site recovery of electricity and heat may lower the net-energy consumption of a process and also may increase the total energy recovery from COD substantially compared to only sludge incineration. Furthermore, the conducted MEBs show that the integration of COD recovery via CEPT for maximum energy and/or via EPS recovery can increase the total COD recovery rate significantly. Up-concentrating primary COD by CEPT can significantly increase the energy recovery potential from COD compared to only secondary sludge digestion because it prevents COD oxidation to CO_2 but leads to a high fraction of influent-COD recovered as energy, either anaerobically or through incineration. Therefore, CEPT integration halves aeration requirements of the plant and since aeration is a major operational cost factor of aerobic wastewater treatment processes (Zubowicz et al. 2017) it may lower operational costs significantly. Despite these promising results, the overall performance of CEPT integration can only be revealed if required polymers are considered in complete process economic and environmental impact assessments.

The model shows that EPS recovery integration promises to recover several tonnes of biopolymers daily which could generate substantial economic revenues for the plant. The recovery of EPS means that a fraction of the influent COD and P leaves the process as polymers and is therefore not recoverable in other forms any longer. On the contrary, the integration of CEPT for maximum on-site energy recovery decreases the EPS recovery potential significantly. Thus, a clear trade-off between maximum EPS and maximum energy recovery exists. The integration of struvite recovery is questionable when ash-P recovery is a possibility because the recoverable influent-P rates are much lower for struvite than for ash-P. Therefore, the total influent- P recovery rate is not significantly improved by struvite crystallization from digestate supernatants. Nevertheless, ca. 400 kg of struvite could be daily recovered on-site the plant and when marketed successfully could generate additional revenues to the operating utility.

The revealed trade-offs between resource recovery technology integration, show that it is important during the early design stage of a WRF to decide which resources that are potentially recoverable from the wastewater stream should be preferred over others and why. This requires inclusion of case related arguments. The integration of

- AD (including decanter centrifuge and CHP unit),
- CEPT,
- EPS extraction, and
- struvite crystallization

into an AGS-based treatment plant will alter its technical, economic and environmental performance. To identify the most feasible process design each possible process alternative needs to be assessed further in these dimensions. This study shows that mass and energy balances are useful in early stage WRF design because they provide the basis for those subsequent assessment steps. In addition, the balances allow an early estimation about which processes are promising in terms of the quantitative resource recovery potential which provides important insights for the development of value chains for recovered resources. Since mass and energy balances require only data, they are relatively cheap to conduct and are therefore an excellent tool to assess a WRF process design at a very early design stage regarding its resource recovery potential and effluent concentrations of modelled constituents. Thus they provide a useful basis to pre-select promising designs for further in depth and more costly techno-economic and environmental impact assessments. To further improve the precision of predictions from comparable mass and energy balances it is useful to apply COD fractionation into easily and slowly biodegradable and non-biodegradable COD. Another parameter that has to be applied with greater accuracy in the future is the yield of full scale EPS down streaming processes as data is yet unavailable. The first commercial EPS recovery process in Zutphen (The Netherlands) will hopefully reveal more insights on this and other parameters related to successful EPS recovery up-scaling. In addition, this study suggests to investigate sludge incineration further as it is a key process regarding both COD and P recovery potentials. Uncertainties remain on how to optimally decrease the water content of waste sludge to maximize its heating value and what factors determine the recoverability of useful P from incineration ashes.

4

A techno-economic analysis of membrane-based advanced treatment processes for the reuse of municipal wastewater

This chapter has been submitted as:

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“We forget that the water cycle and the life cycle are one.”

Jacques Yves Cousteau

4.1. Introduction

Humans consume water across the globe for domestic consumption, for industrial manufacturing purposes and for agriculture. The share of each water usage type of the total water abstraction may differ greatly between countries. While the industrial sector and especially the power generation industry, is in many western countries the largest consumer of abstracted freshwater, agriculture is responsible for the highest water abstraction rates in other countries (Blackhurst et al. 2010; Ranade and Bhandari 2014). Water scarcity is the geographic and temporal mismatch between freshwater demand and availability and is expected to be increased due to climate change. In addition, increasing population, improving living standards, changing consumption patterns, and expansion of irrigated agriculture drive the growth in global water demand (Mekonnen and Hoekstra 2016). The reclamation of water from municipal wastewater treatment plant (WWTP) effluents has been widely recognized as a practical alleviation of regional water scarcity and is therefore promoted politically by the European Commission (European Commission 2018). It creates a yet untapped water source which guarantees a high level of supply reliability because its production is relatively constant (Garcia and Pargament 2015). Since WWTPs are inherently located close to cities, reclaimed effluents become available where water is demand is high(Rietveld et al. 2009).

Several technologies have been proposed to reclaim water from municipal wastewater, such as membrane filtration, advanced oxidation, activated carbon or constructed wetlands. Membrane-based technologies have attracted significant attention because membranes act as a physical barrier for a wide range of contaminants including contaminants of emerging concerns (CECs) (Fatta-Kassinos et al. 2016). Especially ultrafiltration (UF) and reverse osmosis (RO) have been successfully applied in full scale WWTP effluent reclamation processes (Shang et al. 2011; Helmecke et al. 2020). Another advantage of membrane processes is that they can flexibly be scaled up with different unit operations and membrane types to add treatment capacity if necessary (Quist-Jensen et al. 2015). Various full scale studies have demonstrated that membrane-based advanced treatment processes (MATPs) can be designed to reclaim WWTP effluents for all three water usage types: (i) (in)direct potable reuse for domestic consumption (Ortuño et al. 2012; Chalmers and Patel 2013; Van Houtte and Verbauwhede 2013), (ii) demineralised process water for industrial reuse (Majamaa et al. 2010; Shang et al. 2011) and (iii) irrigation water for agricultural reuse (Hamoda et al. 2015).

Despite the proven applicability and advantages, two main bottlenecks have been repeatedly identified in scientific literature that need to become optimized to make membrane driven wastewater reuse even more feasible. Those are the high energy consumption due to required operational pressure and high process costs (Verstraete et al. 2009; Yangali Quintanilla 2010; Batstone et al. 2015; Eslamian 2016; Helmecke et al. 2020). Little is known about the generic differences in energy consumption and costs of MATPs that reclaim WWTP effluents for different reuse types.

This is because a generic comparison between existing case studies that reclaim wastewater for the three reuse types is impossible due to different unit operations applied in each case study, differing feed water compositions, and different methods to calculate energy consumption and process costs. Therefore, it is difficult to state from an energy and cost perspective which reuse type should be preferably targeted by a wastewater reuse project and why. To provide decision guidance from a reclamation process performance perspective and enable a fair comparison, a common reference has to be defined. The WWTP effluent quality, the applied unit operations and the process assessment methodology should be consistent. Only then a valid comparison of MATP performances for different reuse types can be carried out (Raffin et al. 2013).

It is unquestionable that the energy consumption and process net costs of MATPs depend on the targeted reuse type because it defines the required water quality and consequently the process design. This implies also that each MATP has different water recovery rates and therefore, revealing the different energy consumption and costs of each reuse type requires to compare results based on m^3 reclaimed water. In addition, market prices for reclaimed water may differ significantly (e.g. potable water is usually more expensive than irrigation water) and a fair net cost comparison needs to take this into account.

The primary goal of this study is to estimate and compare the net costs and the energy consumption of MATPs that reclaim wastewater for industrial, potable, and agricultural use. This is achieved by theoretically designing and modelling four different MATPs that are all based on the same core process (figure 4.1).

The reclamation of WWTP effluents to alleviate water scarcity can conflict with other sustainability related goals of water utilities. Due to a high energy consumption and brine production of MATPs the environmental footprint of wastewater reuse has been criticized (Daigger 2008; Delacamera et al. 2016). We argue that due to renewable energy conversion technologies energy is no longer a limited resource while water remains more critical and should therefore be prioritised. The second goal of this study is therefore to present two different process optimisation possibilities that further improve the sustainability of MATPs. One possibility is the integration of renewable energy technologies (i.e. solar energy and biogas). The area of photovoltaic modules required to run the modelled MATPs solely on solar energy is calculated assuming Dutch climate conditions. Another possible renewable energy integration system investigated is the recovery of electricity from the chemical oxygen demand (COD) contained in municipal wastewater via anaerobic digestion (Rulkens 2008b).

The second process optimisation possibility studied is the integration of a softener/biostabilizer RO pre-treatment to increase the RO recovery rate. Membrane based processes are often presented as reliable to remove a wide range of organic pollutants which makes them a suitable option to cope with variations in WWTP effluent qualities (Hamoda et al. 2015). However, practical experiences from full scale UF-RO reclamation processes report severe biofouling potential in RO which leads to extensive membrane cleaning (Majamaa et al. 2010). Although the UF is successful in total suspended solids (TSS) removal, its capability to provide a high quality RO feed water is limited as it does not remove pollutants responsible for scaling, organic and bio fouling. Designing a RO pre-treatment that is more robust to variable feed water qualities would improve RO recovery rates, energy consumption and brine production (Slagt and Henkel 2019).

Since this study is based on the idea that a fair of MATPs for different reuse types requires a consistency in unit operations applied in each process model, it also shows what changes are needed to design a 'fit for multi-purpose' instead of a fit for single-purpose MATP. Therefore, in the outlook the possibility of designing a fit for multi-purpose MATP that can reclaim wastewater for different reuse types and cope with temporarily changing water demand patterns (e.g. from agriculture) is presented.

To the best of the authors' knowledge this is the first study to generically compare different reuse types and associated MATP process designs in the performance criteria of energy consumption and net costs. Beyond that, the introduction of new technical process optimization possibilities is urgent to reduce the trade-off between (i) the sustainability goal of decreasing water scarcity and (ii) carbon footprint reduction and cost effectiveness. The results allow to make better informed decisions because knowing in which range the differences in energy consumption and net costs lie helps to decide for a particular wastewater reuse type.

4.2. Methodology

In the following section the modelled membrane-based advanced treatment processes are described including the rationale for process design choices. Studied MATPs are shown in figure 4.1 while important process parameters and assumptions of each operational unit are presented in table 4.2. More detailed information of operational parameters can be found in the appendices.

4.2.1. Estimation of recovery rates and energy consumption

All studied MATPs are modelled with a fixed feed flow of 100 $m^3\ h^{-1}$ which represents a small WWTP and provides a generic scalable number. Each MATP is based on the same core process design including a buffer tank, a pre-filtration step and an ultrafiltration unit. Depending on the specific water quality requirements of a reuse type, the core process is extended by reverse osmosis, ion-exchange, remineralisation and/or ultraviolet light treatment (figure 4.1). All materials and process units applied in this study are commercially available technologies ready for full scale application. In total, three performance criteria have been investigated for each MATP: (i) recovery rates, (ii) energy consumption and (iii) net costs.

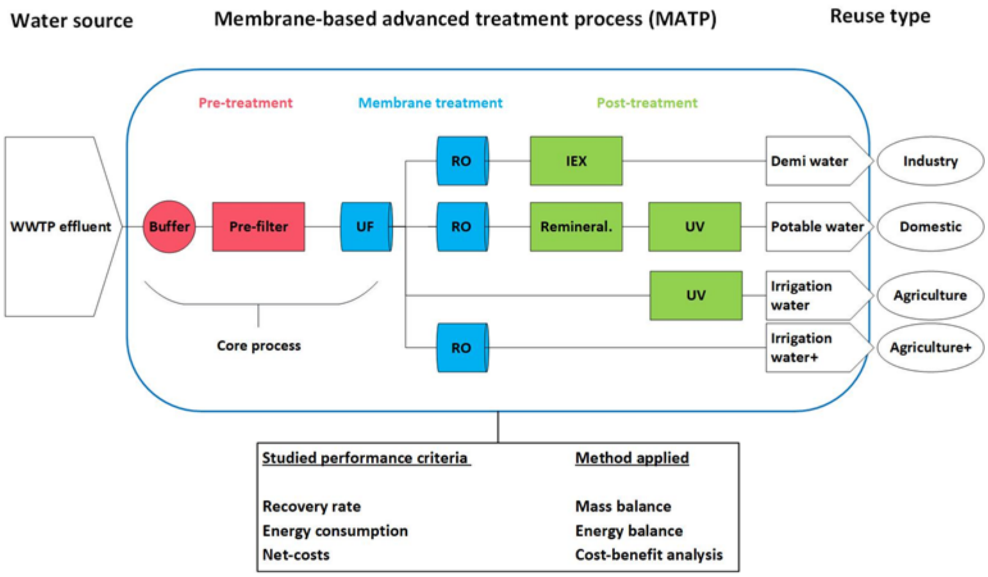

Figure 4.1 Scope and concept of the study with system boundaries in blue rectangle. Four individual MATP designs have been modelled. Irrigation water is reclaimed by two processes where 'irrigation water+' indicates the higher water quality obtained compared to 'irrigation water'.

To obtain generically comparable results, a "standard" WWTP effluent quality was defined that meets the Dutch legal effluent quality standards (table 4.1) (Shang et al. 2011). A sensitivity analysis has been conducted at the end of the study to reveal the impact of possible effluent quality fluctuations (± 20%) on the measured performance criteria.

Table 4.1 Modelled WWTP effluent quality (Shang et al. 2011)

Parameter	Value
Na^{+} (ppm)	311
Cl^{-} (ppm)	463
Mg^{2+} (ppm)	31
HCO_3^{-} (ppm)	325
SO_4^{2-} (ppm)	96,5
Ca^{2+} (ppm)	83
SDI 15	5
TSS (ppm)	6
Turbidity (NTU)	4
Conductivity (µS/cm)	2187
pH (15 °C)	7,5

Mass and energy balances (MEBs) are conducted to estimate the recovery rates and energy requirements of designed processes using different modelling tools. The "DuPont Water Solutions Water Application Value Engine" (WAVE) software is used for integrated modelling of the UF, RO, and ion exchange mixed bed (IEX). It uses harmonized data for all products and processes and provides complete mass and energy flows (DuPont).

The impact of fouling and the required chemical cleaning-in-place (CIP) in the RO are added using an in-house calculation method (Jafari et al. 2021). This means that both economical and operational impacts of fouling are calculated based on plant performance data (e.g., pressure drop, permeability and CIP events) using non-empirical cost models as explained in detail by (Jafari et al. 2021). This way, the energy consumption and costs caused by RO fouling have been taken into account through operational downtime, chemical use and CIP heat requirements. To model those unit operations not available in the WAVE software, i.e. pre-filtration, remineralisation (remin.), ultraviolet light (UV) performance data have been used from full-scale installations described in scientific articles and/or estimated in consultation with technology providers (Trojan Technologies inc., WeUVcare, Xylem Water Solutions Nederland B.V., Global Water Engineering B.V., Evides Industriewater B.V., DuPont Water Solutions). Table 4.2 summarizes the major process parameters applied in the models unit operations. More detailed model metrics can be found in the appendices.

Table 4.2 General process parameters shown for each unit operation applied in the model.

Pre-filter	
Pore size (µm)	100
Ultrafiltration	
Applied pressure (bar)	2,3
Operation mode	Constant flux
Total number of elements	28
Elements type	DOW™UF SFP-2880
Design flux (LMH)	50
Cleaning protocol:	
Forward flush	with UF feed water
Backward flush interval (h)	1
Backward flush with UF permeate (min)	3,8
CEB water with UF permeate (min)	16,1
CEB water interval (h)	12
CIP cleaning interval (h)	30
CIP water with UF permeate (min)	312,8
Reverse osmosis	
Configuration	Double stages (2:1)
Total number of elements	90
Element type	FilmTec™ ECO-PRO 400
RO average flux (LMH)	22,5
Antiscalant (mg/l)	3,5
RO feed flow rate (m3/h)	94,2
RO recovery (%)	80
Cleaning protocol:	
CIP frequency (events/yr)	40
CIP duration (h)	8
CIP step 1 acid cleaning (HCL)	4 hours at pH 2
CIP step 2 Alkaline cleaning (NaOH)	4 hours at pH 12
Rinsing after each step	demineralized water
Ion-exchange (mixed-bed)	
Vessel type	Amberpack™ Sandwich
SAC (internal regeneration)	AmberLite™ HPR1200 H
SBC (internal regeneration)	AmberLite™ HPR4200 Cl
Linear Velocity (m/h)	38
Design flow rate (m3/h)	75
Design run time (h)	48
Regeneration time (h)	4,5
SAC Volume (m3)	2,2
SBA Volume (m3)	4,5
Regeneration (SAC) (HCl g/l)	80
Regeneration (SBA) (NaOH g/l)	80
Regeneration Temperature ^{o} C	15
Remineralisation	
Total hardness achieved (mmol/L) (Ca + Mg)	>1
Type of remineralisation process	Lime saturator
Ultraviolet light disinfection	
Assumed UV dosage (mJ/cm2)	80

Assumed log removal	4
Lifetime of lamps (h)	12000
Energy consumed per lamp (W)	100
Softener/biostabilizer	
Vessel type	Amberpack™ Sandwich
SAC (internal regeneration)	AmberLite™ HPR1100 Na
SBC (internal regeneration)	AmberLite™ HPR4580 Cl
Internal bed area (m3)	4,4
Linear Velocity (m/h)	23
Design flow rate (m3/h)	100 (SAC); 94 (SBA)
Design run time (h)	10 (SAC); 10 (SBA)
Regeneration time (h)	1,71 (SAC); 3,46 (SBA)
SAC Volume (m3)	8,3
SBA Volume (m3)	5,3
Regeneration Temperature ^{o}C	15
Regeneration solution	RO brine

4.2.2. Modelled MATPs and process design choices

4.2.2.1. Core process

Prior to the core process, a buffer tank functions to balance out hydraulic load variations during day/night or seasons (figure 4.1) (Majamaa et al. 2010). The following pre-filter (100µm) protects the downstream UF from larger suspended solids. This combination of unit operations has been applied in various full scale WWTP effluent reclamation processes (Van Houtte and Verbauwhede 2013; Hamoda et al. 2015). Instead of UF, microfiltration (MF) membranes have been applied for wastewater reclamation in the Netherlands (Shang et al. 2011) but UF has distinct advantages. Firstly, it gives process designers more flexibility to choose a suitable membrane for a given feed quality as UF pore sizes have a wider range between 0,1 – 0,001 µm (Rao 2013). Secondly, in contrast to microfiltration, UF membranes also remove soluble organic particles including coliform bacteria more effectively and therefore may produce a permeate quality that lies closer to the legal standards for irrigation water (Oron et al. 2006). Sand filtration has also been discussed as an alternative unit operation to UF but the removal efficiency of suspended solids may vary greatly and the effluent still contains colloidal matter which can cause problems in the RO (Verstraete et al. 2009). However, in this study the UF unit is modelled with full redundancy of equipment to ensure a steady operation despite the operational downtime during membrane cleaning.

4.2.2.2. Process extension for industrial reuse

Figure 4.1 illustrates the process extension for demi water reclamation. Due to industrial water applications at high temperatures together with internal water evaporation processes, industrial process water should have a low hardness and a low salt concentration (Rietveld et al. 2011). In the past two decades, RO in combination with ion-exchange mixed bed (IEX) has become the standard process to treat water for industrial applications to a quality that prevents scale formation and/or corrosion in equipment, like e.g. high pressure steam systems. The IEX resins are regenerated using hydrochloric acid and sodium hydroxide. To ensure a steady operation during resin regeneration the IEX unit is applied with full redundancy of equipment. Moreover, an open tank degasification unit is applied prior to the mixed-bed IEX to remove 70% of CO_2 from the RO permeate.

4.2.2.3. Process extension for potable reuse

Removal of pathogens and toxic pollutants is paramount in reclamation of potable water to avoid potential health risks. Related to this, safe guarding the membrane integrity is very important (Trussel 2012). Generally speaking and dependent on the membrane type, RO is expected to reach a log removal value (LRV) of 6 while UF reaches a LRV of 4 (Warsinger et al. 2018). Recent studies even suggest that RO membranes can reach a LRVs of >7 for different natural viruses (Hornstra et al. 2019). But due to its modular design, a full-scale membrane installation contains a large number of O-ring seals, interconnectors, glue lines and other potential locations which could be vulnerable for integrity breaching

(Pype et al. 2016). Consequently, the modelled MATP for potable reuse includes a final UV disinfection step to reach an additional LRV of 4 (Pype et al. 2016).

Moreover, WWTP effluents still contain a wide range of unregulated inorganic and organic CECs (Helmecke et al. 2020). It has been stated that assessing only pathogen indicators is therefore not safe in case of potable reuse (Wang et al. 2015b) and that post-treatment with advanced oxidation or adsorption process for CEC removal have to be applied (Stefanakis 2016). For example, the incomplete removal of certain chemicals has been reported, e.g. for boron or di-butyl phthalate (DBP) (Trussel 2012). The removal of CECs might become a requirement in the future due to changing legislations and additional quality indicators (Hendry and Benidickson 2017). Nevertheless, the modelled process for potable reuse does not include a final advanced oxidation process because it is not clear which CECs could be primarily subject to new legislation nor which advanced oxidation process is most suitable to be applied then.

To ensure that no sand particles or other debris enter the RO system in case of potential UF system leakages or due to other unforeseen problems it is common practice to install cartridge filters before a RO (Farhat et al. 2020). Since the UF permeate is still biologically active, cartridge filter costs cannot be neglected. Therefore, the model includes cartridge filters as integral part of the RO system without labelling them as a separate filtration step.

The permeate of the RO unit is not directly potable due its low alkalinity and must be remineralized with hardening components (Ca^{2+} and Mg^{2+}). The model follows the remineralisation process described by (El Azhar et al. 2012), to reach a total hardness higher than 1 mmol/l required to meet Dutch potable water quality legislation (Beyer et al. 2014). Considering the low solubility of lime in water this process applies a lime saturation tank and a mixer that feeds an adequate amount of lime saturated water into the RO permeate (El Azhar et al. 2012). Remineralisation of water after reverse osmosis can improve drinking water quality significantly and even allows to adjust total dissolved solid contents to empirically confirmed concentrations that provide users with the most favourable taste intensity (Vingerhoeds et al. 2016).

4.2.2.4. Process extension for agricultural reuse

The discussion whether to apply UF or UF-RO for irrigation water reclamation from municipal wastewater is controversial due to the trade-offs between process costs and microbial and chemical safety (Helmecke et al. 2020). The EU guidelines on water quality for irrigation water from municipal wastewater (Table 4.3) differentiate between four water qualities depending on the targeted crop, its intended use and the irrigation method applied (European Commission 2018):

- Quality A allows direct contact of the reclaimed water with the edible part of the crop
- Quality B is not allowed to have direct contact with the edible parts of the crop but is suitable for food crops that are processed before consumption and for crops used as feed
- Quality C is only allowed when drip irrigation is applied to crops mentioned in quality B.
- Quality D only allows to irrigate crops for industrial use, energy and seeded crops.

Table 4.3 Recommendation of the European Commission to implement water quality standards for irrigation water reclaimed from municipal WWTPs (European Commission 2018)

	A (≤)	B (≤)	C (≤)	D (≤)
E.coli (cfu/100ml)	10	100	1000	10000
BOD5 (mg/l)	10	25		
TSS (mg/l)	10	35		
Turbidity (ntu)	5	-		
Legionella spp. (cfu/l)	1000 (greenhouse use)			
Intestinal nematodes (eggs/l)	1 (feed and pasture)			

Reaching the highest quality for safe direct contact of the reclaimed water with the edible part of the crop requires the following LRVs to be reached by any wastewater reclamation process: *E. coli* ≥ 5; total coliphages ≥ 6; *clostridium perfringens* ≥ 5 (European Commission 2018). Since UF alone can be expected to not consistently reach these LRVs (Warsinger et al. 2018), one might evaluate the use of UF permeate for irrigation as too risky. Although several studies suggest that UF alone can successfully remove bacteria and nematode eggs from effluents (Gómez et al. 2006; Sabater Prieto et al. 2012), in practice, UF membranes operated at reuse facilities did not always achieve complete bacterial rejection (Warsinger et al. 2018). Therefore it is recommendable to integrate a subsequent disinfection step which is usually achieved with UV light treatment. This might be especially valid for greenhouse irrigation where the risk of aerosolisation of pathogens is given (European Commission 2018).

Due to these uncertainties regarding UF, RO has been claimed to be the better option because it is a total barrier for pathogens and also salts and CECs are rejected at a high rate (Warsinger et al. 2018). Irrigating crops with lower quality water may require to add fresh water to prevent salt accumulation in the soil causing significant yield losses (Quist-Jensen et al. 2015). An additional argument in favour of RO integration is potentially higher crop yields achieved with RO permeate compared to UF permeate (Oron et al. 2006).

Considering these controversial results, two different processes are modelled for agricultural reuse. One consists of UF-UV treatment and reclaims irrigation water while the other uses UF-RO treatment. which is referred to as "irrigation water+" in this study. The plus sign indicates that it may easily meet quality standards for a wide range of other applications outside of the agricultural sector since RO can reach a LRV of >7 and removes most CECs (Hornstra et al. 2019).

4.2.3. Economic analysis

To estimate the economic performance of each MATP, cost-benefit analyses (CBA) have been conducted calculating their net present value (NPV). The procedure has been described in the field of wastewater resource recovery elsewhere in greater detail (Kehrein et al. 2020b) and applies formula 1.

$$NPV = \sum_{t=0}^{n} \frac{NB_n}{(1+i)^n} \qquad (1)$$

Where the net present value (NPV) at time t, calculated for a time horizon of n years, is the sum of discounted annual net benefits (NB) assuming a discount rate i (Kehrein et al. 2020b).

A discount rate of 5% has been applied in the CBA which accounts for the opportunity cost of time by discounting future costs and benefits because of the profit that could be earned in alternative investments (European Commission 2015b). All net benefits have been discounted along a 20 year time horizon which represents the life time of the reclamation process. Cost factors as well as water prices assumed in the CBA are representative for The Netherlands in the year 2020. Electricity costs are assumed to be 0,1 € per kWh which represents an average price for non-household electricity consumers below 2000 MWh yr^{-1} in the Netherlands (CBS Statline 2020).

As shown in formulas 2-4 and figure 4.2, the CBA includes revenues from water sales of either demi water ($€_{demi}$), potable water ($€_{potable}$), irrigation water ($€_{irrigation}$), irrigation water+ ($€_{irrigation+}$) as the only benefits. Costs include operational expenditures (opex) for energy ($€_{energy}$), input of chemicals ($€_{chem}$), waste disposal costs ($€_{waste}$), equipment replacements ($€_{replace}$) and labour for process operation ($€_{labour}$). Labour requirements and costs are estimated based on personal communication with sector specific companies and are typical for The Netherlands to operate a process of similar size. For the calculation of capital expenditures (capex) only costs related to the purchase of process equipment and its installation ($€_{equip\ \&\ install}$) are considered. The equipment purchase prices of each unit operation have been estimated in close consultation with existing technology providers to obtain realistic "plug and play" prices of each unit operation. Real prices are more accurate because prices of individual unit operations vary considerably in literature studies and at online market places. However, other possible capex factors, like e.g. land acquisition, planning, and construction of buildings are excluded from the capex calculations. The reason is that those cost factors are highly case dependent or underlie strongly effects of economy of scale.

$$€_{OPEX} = €_{energy} + €_{chem} + €_{waste} + €_{replace} + €_{labour} \quad (2)$$

$$€_{CAPEX} = €_{equip} \quad (3)$$

$$€_{Revenues} = €_{demi} + €_{potable} + €_{irrigation} + €_{irrigation+} \quad (4)$$

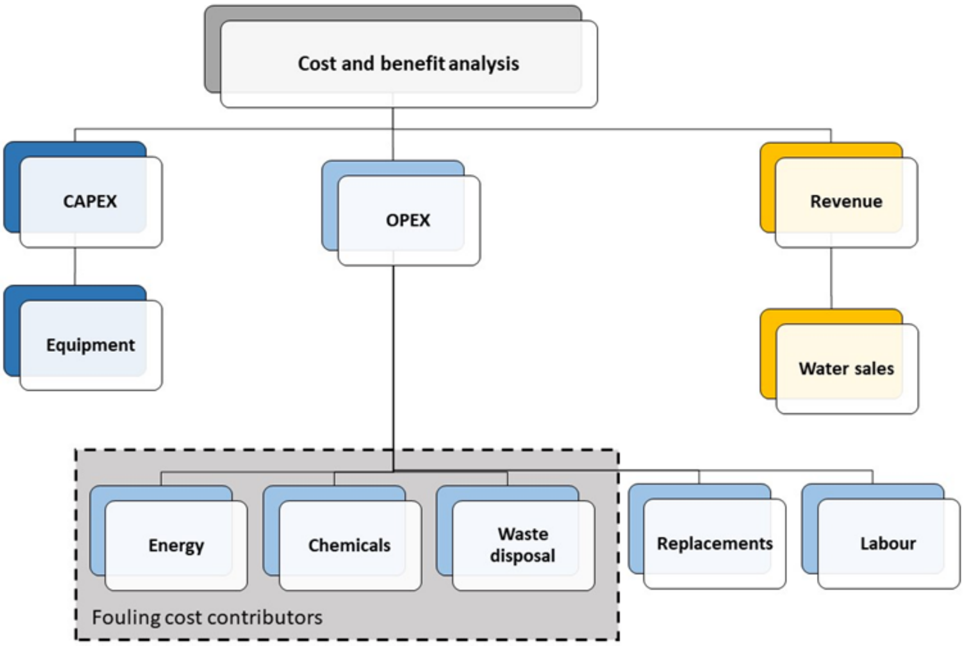

Figure 4.2 Cost factors (capex, opex) and benefits (revenues) included in the CBA calculations.

The water prices applied in the CBA represent Dutch gross market values and have been corrected for 9% value-added tax to obtain net prices (table 4.4).

Table 4.4 Assumed market prices (€/m^3) for reclaimed water for different reuse types.

Water prices	Gross price	Net price (excl. 9% VAT)
Demi water	1,10	1,00
Drinking water	0,93	0,85
Irrigation water	0,19	0,17
Irrigation water+	0,60	0,55

Gross potable water prices have been estimated to be 0,93 €/m^3 which is the price charged in 2020 to households by the water company ‵'Evides Waterbedrijf N.V." that serves over two million inhabitants in South-West Netherlands (Evides 2020). Gross demi water prices paid by industries may depend on different factors, like e.g. the delivered water quality (e.g. demineralised) and especially on the purchased volume. The modelled flow rate of 100 m^3 h^{-1} WWTP effluent leading to a typical recovery rate of 70-80% demi water represents a relatively small water quantity in an industrial context. Considering this low purchase volume, a gross price of 1,1 €/m^3 was assumed for demi water which may be lower (≈1 €/m^3) when purchased at large industrial scale (>1000 m^3 h^{-1}). Irrigation water+ has been accounted for with a gross price of 0,6 €/m^3 which is paid by fruit growers in the Dutch region Zuid-Beveland for potable water to irrigate fruit trees (STOWA 2019). The lower quality irrigation water is estimated to cost 0,19 €/m^3 which is the fee that farmers pay for the allowance to pump and use groundwater in the Dutch region of Brabant (STOWA 2019). These low prices for irrigation water have also been confirmed by studies from other European countries that have shown that farmers may perceive reclaimed municipal wastewater as of minor quality and therefore have a low willingness to pay if alternative water sources are available (Quist-Jensen et al. 2015).

Moreover it is assumed that all water prices remain constant over the 20 year time horizon applied in the CBA. The residual value of each reclamation process has been calculated by using the NPV of cash flows occurring for an additional five years after the computed time horizon is over. Costs of finance are not considered in the CBA.

4.2.4. Process optimisation and integration for more sustainability

As stated above, this study also aims to explore the potential to further improve the sustainability of MATPs by (i) modelling a process optimisation approach that increases RO recovery rates; and (ii) by integrating renewable energy sources into MATPs.

4.2.4.1. Increasing RO recovery rates

Although the UF is successful in TSS removal, its capability to provide a high quality RO feed water is limited as it does not remove pollutants responsible for scaling, organic and bio fouling. Applying a RO pre-treatment that is more robust to variable feed water qualities than only UF, would improve RO recovery rates, energy consumption and brine production. Consequently, an optimized process design for demi water reclamation is modelled and compared to the initially modelled standard process. It integrates a softener and a biostabilizer unit prior to the RO to maximize process recovery rates. The applied ion exchangers target the specific problem species that can influence RO performance.

First, a pre-softening unit is integrated that removes multivalent cations (Ca^{2+}, Mg^{2+}, Fe^{2+}) which lowers the RO's scaling potential and therefore may improve membrane performance (Salvador Cob et al. 2015; Hijnen et al. 2016). This can be achieved by exchanging those ions with monovalent ions (Na^{+}) under slightly acidic conditions. Secondly, to decrease the biofouling potential of the RO feed even further an anion exchange step is integrated that removes also fractions of TOC including organic contaminants from the RO feed, like e.g. humic acids. In addition, it removes multivalent anions (e.g. PO_4^{3-} and SO_4^{2-}) from the RO feed which provides a bio-stabilizing effect (Slagt and Henkel 2019). Such a combined cation/anion exchange unit prior to the RO, named here softener/biostabilizer, allows to increase RO recovery to significantly higher values (>90%) due to negligible risk of scaling and low biofouling potential due to phosphate limitation (known as biofouling control strategy) (Vrouwenvelder et al. 2010; Slagt and Henkel 2019). Therefore 50% less CIP events have been modelled for the optimized demi water MATP design.

4.2.4.2. Integration of renewable energy sources

Water reclamation is generally referred to as an energy intensive process leading to an increased carbon footprint of WWTPs (Eslamian 2016). To better understand how renewable energy sources can lower the carbon footprint the photovoltaic (PV) net energy that can be generated in the city of Delft (The Netherlands) has been calculated using the "Photovoltaic Geographical Information System" database and calculator provided by the European Commission's Joint Research Centre (European Commission 2020). Assumed parameters are shown in table 4.5. It is estimated how much PV area is needed to operate studied MATPs which is an important number to show if PV installations can be integrated on-site of a WWTP.

Table 4.5 Parameters applied to calculate required PV area to operate MATPs with solar energy.

Database used for calculation	PVGIS-SARAH
1 kWp PV capacity (m^2)	10
PV technology	Crystalline silicon
Yearly in-plane irradiation (kWh/m^2)	1263
Total loss (%)	-18,85
Yearly PV energy production (kWh)	1025

The second renewable energy integration system investigated is the recovery of electricity from the chemical oxygen demand (COD) contained in the WWTP influent via anaerobic digestion (Rulkens 2008b). The obtained methane is assumed to be converted into electricity in a combined heat and power unit to be then consumed by the MATP. Table 4.6 shows the realistic assumptions made in the calculations of electricity recoverable from the anaerobic sludge digestion route.

Table 4.6 Assumptions made to estimate the electricity recovery from municipal wastewater.

Parameter	Assumption	Reference
Influent COD concentration	750	(Henze and Comeau 2008)
Flow rate (m^2/h)	100	Own assumption
Energy content COD (kJ/g)	17,8	(Heidrich et al. 2011)
Primary COD capture (%)	60	(Wan et al. 2016)
COD into secondary sludge (%)	40	(Winkler et al. 2013)
COD converted into biogas (%)	50	(Khiewwijit et al. 2016)
Methane content biogas (%)	65	(Frijns et al. 2013)
Electrical efficiency (%)	40	(Verstraete and Vlaeminck 2011)

4.3. Results and discussion

In the following section the energy consumption and net costs are presented and discussed for all studied MATPs. To make a fair comparison between studied wastewater reuse types, the results need to be compared not only in absolute values but also based on 1 m^3 reclaimed water. Therefore, the recovery rate of each MATP was estimated first. Any detailed process design information can be found in the appendices.

4.3.1. Recovery rates

The results show the recovery rates for each reuse type taking an upstream perspective in the process designs that all consider 100 m^3 h^{-1} WWTP effluent as a feed (figure 4.3). The recovery rate is highly dependent on the RO unit which has 19% loss as brine leading to an overall process recovery rate of ca. 75% for demi, potable, and irrigation water+ reclamation. Applying no RO but only UF followed by UV light disinfection would lead to a significantly higher recovery rate above 90%. The reclamation of quality irrigation water provides therefore advantages due to less brine production and more water actually reused.

The model predicts that UF meets easily the turbidity and TSS requirements of all EU guidelines on water quality for irrigation water from municipal wastewater (table 4.3). But as explained in the methodology section, it is questionable if microbial standards can be reliably reached. However, the significantly lower overall process recovery rate associated with RO integration could be a valid argument to design processes for irrigation water reclamation only with UF to decrease process costs.

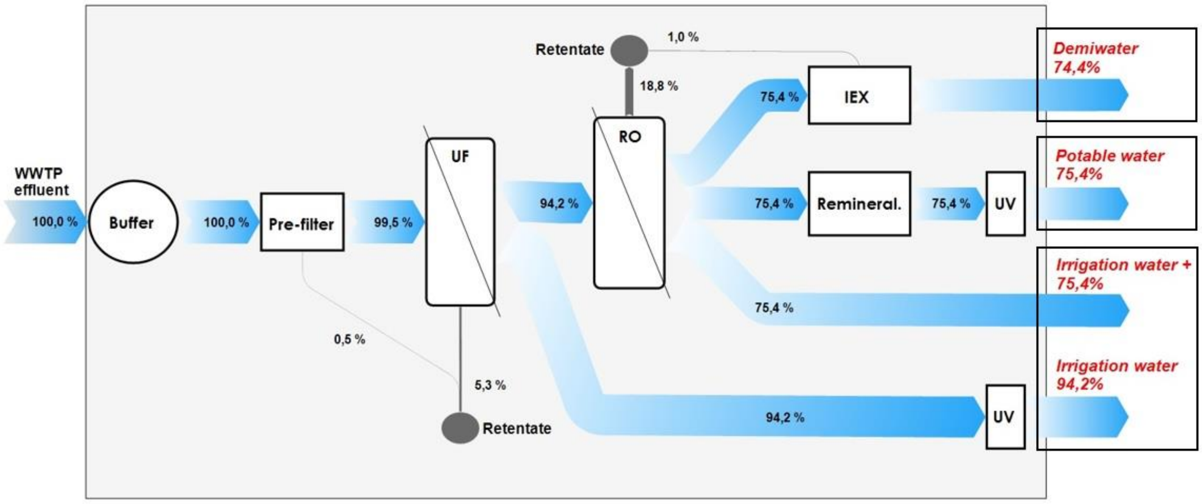

Figure 4.3 Modelled process designs showing water flows in %. Final recovery rates are highlighted in red.

4.3.2. Energy consumption

The operation of a WWTP in Europe has been estimated to require ca. 0,45 - 0,6 kWh/m^3 (Solon et al. 2019a) and can account for a significant share of the total energy consumption of a small municipality (Berger et al. 2013). The energy consumption of MATPs covers in literature also a broad range of 0,7 - 2,3 kWh/m^3 reclaimed water depending on the system boundaries of the respective study (Quist-Jensen et al. 2015). The study at hand confirms previous results that the energy consumption of an MATP is largely determined by the integration of RO. Figure 4.4 shows the absolute energy consumption of MATPs defined as the kWh required to treat 100 m^3 WWTP effluent. In absolute numbers, the treatment of 93,7 m^3 h^{-1} UF permeate with RO requires ca. 43 kWh which is significantly higher than all other operational units. The specific energy consumption of each MATP (table 4.7) reveals the energy required per 1 m^3 of reclaimed water and allows a fair comparison between modelled MATPs. Reclamation of irrigation water with UF and subsequent UV disinfection consumes much less energy compared to the other processes, because of the absence of RO and the higher recovery rate of the process. The energy consumption of UV disinfection shows that despite the fact that the larger UF permeate stream for irrigation water reclamation requires more UV lamps than the smaller and cleaner RO permeate stream for potable water reclamation, both UV units have similar specific energy consumption.

Table 4.7 Energy requirements of modelled unit operations shown as specific energy in kwh per m^3 reclaimed water.

	Demi water	Potable	Irrigation	Irrigation water+
Pre-filter	0,003	0,003	0,002	0,003
UF	0,15	0,14	0,11	0,14
RO	0,60	0,59	-	0,59
IEX	0,08	-	-	-
Remin.	-	0,002	-	-
UV	-	0,08	0,09	-
Total	**0,83**	**0,82**	**0,20**	**0,74**

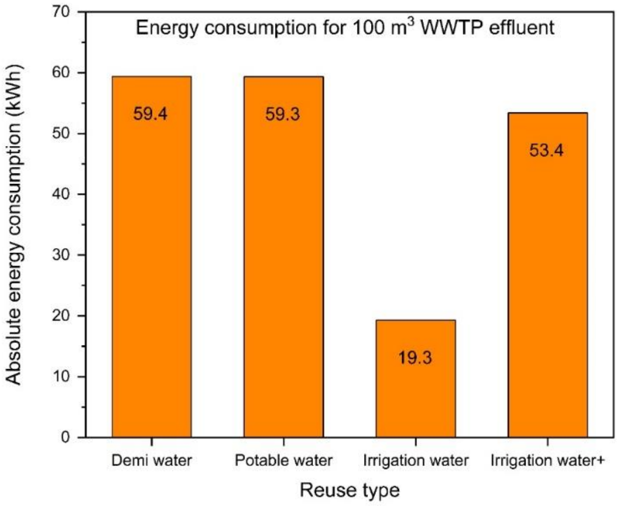

Figure 4.4 Absolute energy in kWh required to treat 100 m^3 WWTP effluent with modelled MATPs.

4.3.3. Cost and benefit analysis

The European water framework directive expects that urban water systems are managed in an economically self-sustained way. This requires that costs are covered by the system itself through pricing of reclaimed water and service fees for wastewater treatment (Castillo et al. 2017). Therefore, the economic performance of MATPs is critical for successful water reuse. To assess this a CBA has been conducted revealing the specific costs per m^3 reclaimed water and the NPVs of modelled MATPs. More detailed cost information can be found in the appendices. Figure 4.5 shows that the opex per m^3 reclaimed water are far higher than the capex needed for initial process equipment and installation. The difference is due to the fact that capex occur only once at the beginning of the assumed 20 year lifetime while opex are due constantly. Therefore, it is arguably more important to design MATPs with the goal to optimize its operation and to safe opex rather than saving capex. This is especially important to consider in tender procedures where decisions are usually capex driven. Nevertheless, it should be noted again that other potential capex factors, like e.g. land acquisition or buildings have been excluded in this study due to highly site specific variations. The inclusion of those cost factors would increase capex further but unlikely exceed opex. Table 4.8 reveals that the RO, UF determine the operational costs of MATPs to the largest extend compared to other unit operations and that labour costs are the second largest opex factor after RO. To show how to possibly design a more cost effective process an optimized process for demi water reclamation that decreases total opex by increasing RO recovery rates is presented below. When RO is applied the highest operational cost factor of MATPs is waste management which refers to the discharge or additional treatment of RO brines (table 4.9) followed by labour costs. If irrigation water is reclaimed without RO, labour costs represent the highest opex factor.

Table 4.8 Overview of capex and opex of each unit operation applied in each MATP (€ct per m^3 reclaimed water per year). All values are undiscounted. Capex consists only of initial process equipment and installation costs.

	Demi water		**Potable water**		**Irrigation water**		**Irrigation water+**	
	Capex	**Opex**	**Capex**	**Opex**	**Capex**	**Opex**	**Capex**	**Opex**
Buffer	0,7	-	0,7	-	0,5	-	0,7	-
Pre-filter	0,1	0,1	0,1	0,1	0,1	0,1	0,1	0,1
UF	3,2	10,6	3,2	10,6	2,5	8,1	3,2	10,5
RO	3,4	32,9	3,4	32,9	-	-	3,4	32,5
IEX	2,5	8,9	-	-	-	-	-	-
Remin.	-	-	0,1	4,1	-	-	-	-
UV	-	-	0,2	3,6	0,2	3,5	-	-
Labour	-	15,9	-	15,7	-	12,1	-	15,7

Table 4.9 Opex factor distribution (%) in total opex for modelled MATPs based on opex per m^3 reclaimed water.

	Demi	Potable	Irrigation	Irrigation+
Energy	12%	14%	10%	13%
Chemicals	11%	7%	2%	4%
Waste	42%	44%	21%	47%
Replacement	11%	10%	10%	10%
Labour	23%	25%	57%	26%

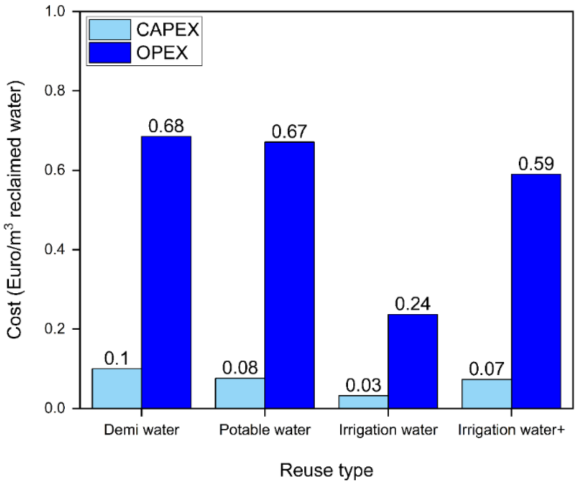

Figure 4.5 Total capex and total opex in € per m3 reclaimed water. All values are undiscounted.

The discounting of future cash flows reveals that both process costs and water prices determine the economic feasibility of MATPs significantly. Demi water reclamation is the most economically attractive reuse type showing a positive NPV of 1,3 Mil € (figure 4.6). This is due to the relatively high price of demi water (table 4.4). A lower price becomes realistic for very large scale industrial clients that purchase much higher quantities (>1000 m^3 h^{-1}) than the 100 m^3 h^{-1} assumed in this study which would lower the NPV. In contrast to demi water, it might be very difficult to develop an economically viable business case for irrigation water reclamation. Given the low prices for irrigation water it is not even economically attractive to forego the RO unit and only apply UF in combination with UV disinfection which implies relatively low opex and capex but still shows a highly negative NPV of ca. -1,2 Mil €. To generate a positive NPV a net price for low quality irrigation water of minimum 0,3 € per m^3 needs to be applied. At the first glance it seems surprising that the irrigation water+ (UF-RO) shows a similar negative NPV as the irrigation water (UF-UV) as process costs are higher and recovery rates lower. But the significantly large price difference of 0,38 €/m^3 between irrigation water and irrigation water+ leads to an equal NPV. This shows that from an economic perspective the application of RO for irrigation water reclamation is as feasible as the application of UF-UV if the difference in water quality is also reflected in the water prices.

When it comes to potable water reclamation, process costs may be covered by the revenues from water sales as a positive NPV is achieved. However, it is important to state that a positive NPV does not suggest an automatic profit can be earned with the reclamation of wastewater but only that the main process costs can be covered by the projected revenues. The positive net benefits can be used for covering cost factors which are excluded in this study, like e.g. construction and non-process related cost factors that occur in water reuse projects, like e.g. water distribution costs (Pearce 2008). Whether an overall solid business case can be developed for a reuse project depends therefore strongly on site specific cost factors, like e.g. distance to customers, land purchase costs and/or right of way costs.

Nevertheless, this study shows that the highest probability to operate a municipal wastewater reuse scheme in an economically feasible way is the reclamation of demi water for industrial purposes.

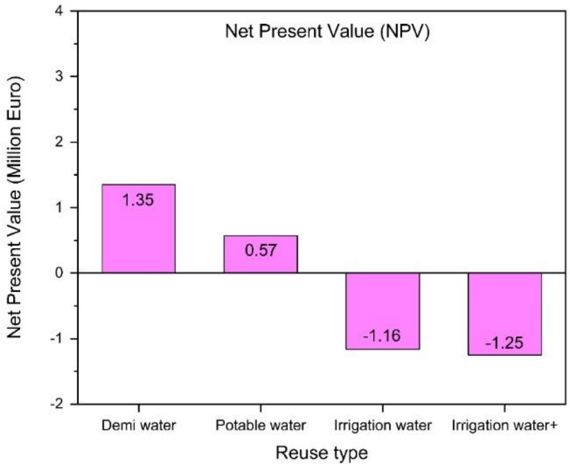

Figure 4.6 Net present value (€) of modelled MATPs.

4.3.4. Process optimisation and integration for more sustainability

In the following two process optimisation and integration concepts for more sustainable MATP operation are presented. First, a concept to increase the recovery rate of RO membranes has been modelled and evaluated that also implies energy and cost savings. Secondly, two renewable energy integration possibilities are presented. The photovoltaic area is calculated to operate a MATP with solar energy under Dutch climate conditions is presented. In addition, the electricity recoverable from anaerobic sludge digestion is calculated to reveal how energy recovery from WWTPs can satisfy the energy demand of MATPs.

4.3.4.1. Increasing RO recovery rates

The presence of multivalent ions in the feed water of RO systems contributes significantly to scaling and biofouling on the membrane surface which results into limited RO system recovery (Hilal and Wright 2018). To increase the RO recovery rate a softener/biostabilizer can be integrated to pre-treat the RO feed (Slagt and Henkel 2019). We applied this concept to the initially modelled MATP for demi water reclamation to compare both process performances in recovery rates, energy consumption and net costs. Since the obtained RO brine is due to the installation of softener/biostabilizer, free of any risk of biofouling and scaling it can be used for UF cleaning. Moreover, due to the absence of both multivalent cations and anions the brine is chemically suitable for the regeneration of the ion exchange resins of the softener/biostabilizer unit (Vanoppen et al. 2016). This has the advantage that no fresh water has to be subtracted from the process for these purposes and also the process's chemical consumption is lower (chemicals are often used in the IEX regeneration) (Slagt and Henkel 2019). For the presented optimized process an addition of only 2,2 g NaCl l^{-1} brine is enough to obtain a useful resin regeneration solution which saves costs. However, since the number of operational units is increasing the process complexation does too. Nevertheless, all applied technologies are mature and often used solutions that are in this concept only operated in a different manner than usually.

The comparison of recovery rates of the initially modelled demi water process (figure 4.3) and the optimized process (figure 4.7) shows that the overall recovery rate increases from 74,4 to 87,3%. As discussed above, this is mainly due to negligible scaling and biofouling potential of the RO feed which allows for elevated RO recovery rate from 80 to 95%. Moreover, lower number of CIP events are required for RO in the optimized process (due to lower fouling) leading to a lowered operational down-time of the plant which contributes to a higher process recovery rate too.

The overall performance improvement of the optimized process compared to the initially modelled demi water process is shown in figure 4.8. The energy consumption of the RO is lowered due to less scaling and biofouling and since RO is the most energy intensive unit the energy consumption per reclaimed m^3 of demi water of the optimized process is decreased by 11%. The comparison of the CBAs of both process designs reveals that absolute capex for the optimized process are 31% higher than for the standard process. This is not only due to the softener/biostabilizer unit but also due to an overall higher requirement of RO modules.

On the contrary, the optimized process requires lower opex per m^3 reclaimed demi water due to its higher recovery rate. Especially the significantly less brine production leads to brine management cost savings. In addition, the higher salt concentration of the brine would facilitate the extraction of solids to comply with zero liquid discharge policies. After discounting future cash flows the final NPV per m^3 reclaimed water is 30% higher for the optimized process. This shows that the initially higher capex are easily offset by the decreased opex. Thus a potential economic advantage has been revealed in this study by integration of a softener/biostabilizer as a RO pre-treatment. The decreased opex are a positive argument for the optimized process from an operators point of view (opex oriented) while a technology supplier (capex oriented) may be deterred by the high capex at first but should consider the full economic performance over time. This should especially be considered for tender procedures in public water reuse projects.

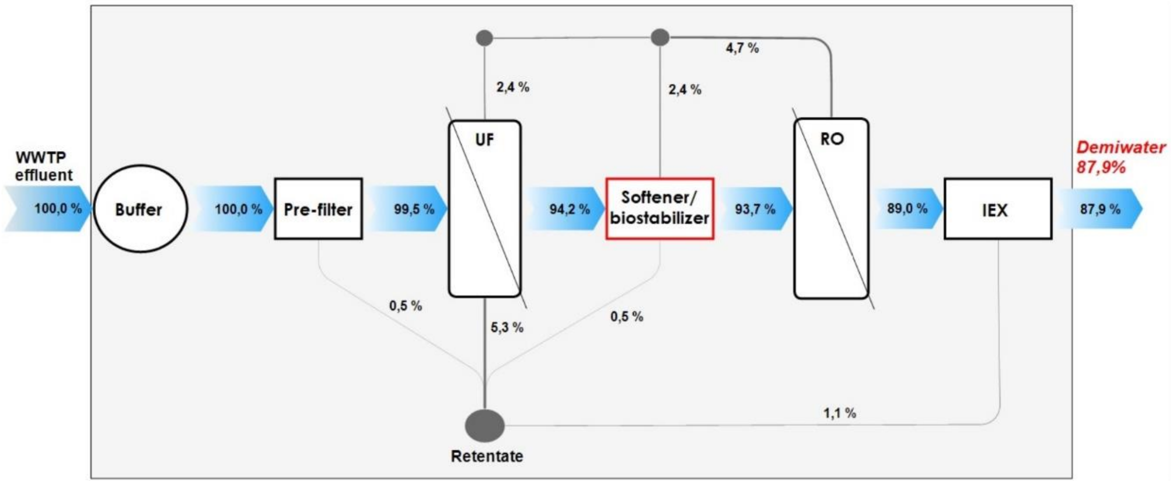

Figure 4.7 Optimized process design for demi water reclamation showing water flows in %. Final recovery rate in red.

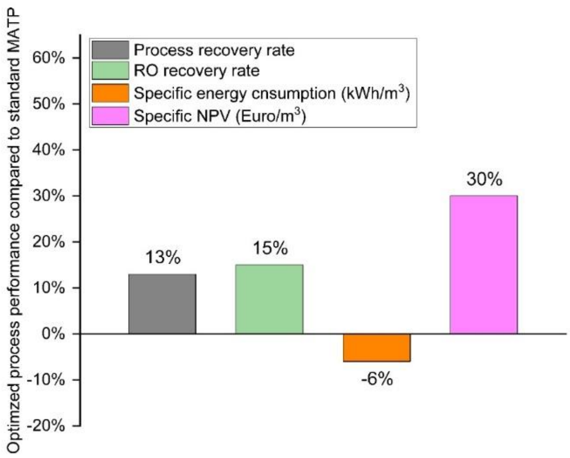

Figure 4.8 Performance of the optimized process design for demi water reclamation compared to the performance of the standard process for demi water reclamation shown in %. 'Specific' refers to results are based on m^3 reclaimed water.

4.3.4.2. Integration of renewable energy sources

The results presented in table 4.10 reveal how much photovoltaic (PV) area is required under Dutch climate conditions to operate each MATP with solar energy. For demi water reclamation a PV area of ca. 5000 m^2 would be required which represents 70% of the size of a football field. This shows that it is realistic to operate MATPs on solar energy as probably only relatively little extra space than the plant itself is required. However, the calculations are based on the assumption that the PV system is connected to the grid. If an off-grid system is applied, energy storage facilities are necessary which implies that additional energy conversion losses have to be considered and the required PV area would increase.

Table 4.10 Photovoltaic module area required to operate modelled MATPs (flow rate: 100 m3 h-1) on solar energy in the City of Delft (The Netherlands).

MATP	PV area required (m^2)	PV area required (m^2/m^3 reclaimed)
Demi water	5077	0,0081
Potable water	5072	0,0080
Irrigation water	1647	0,0020
Irrigation water+	4564	0,0072
Optimized demi	5728	0,0076

In addition to solar energy, the energy that can be recovered in a WWTP from COD via the anaerobic digestion and combined heat and power biogas combustion was estimated (Frijns et al. 2013). The question arises how much energy could come from anaerobic sludge digestion to offset the energy consumption of the MATP and achieve a good overall energy balance of both the WWTP and MATP. The realistic assumptions presented in table 4.6 lead to only $2{,}6\times10^{-5}$ kWh electricity that could be recovered from the 100 m^3 raw wastewater entering the WWTP. This is a negligible amount of electricity compared to the absolute electricity required to operate an MATP as shown in figure 4.4. This estimation excludes a similar amount of heat energy that is additionally recoverable from the biogas combustion process. It should also be mentioned that if a WWTP is designed to recover not only chemical energy but additionally also the heat energy from the effluent via heat exchangers, the total WWTP energy recovery could be significantly further increased (Hao et al. 2019).

4.3.5. Sensitivity analysis

The results of this study may be sensitive to the WWTP effluent quality assumed in the modelled (table 4.1). Therefore, a sensitivity analysis has been conducted for the demi water reclamation MATP (figure 4.9). The impact of the effluent quality was tested by changing the concentrations of selected ions (Na^+, Cl^-, Mg^{2+}, Ca^{2+}), TSS and turbidity by +/-20% of the initially modelled values. Total organic carbon (TOC) is not considered although it is possible that effluent TOC concentrations impact membrane fouling and thus would impact the process energy consumption. Since the distinct correlation of TOC composition and membrane permeability is not clear (Kennedy et al. 2008) and there is a lack of operational data about the impacts of other changing effluent quality parameters and membrane fouling, fouling costs are assumed to be linearly correlated to the effluent quality.

However, the results in figure 4.9 show that changing ion concentrations would change the conductivity of the effluent by +/-18% leading to opex changes of ca. +/-10%. While labour costs have been assumed to not be affected by WWTP effluent quality changes the other opex factors (i.e. energy, chemical reagents, brine management, and equipment replacements) have been included in the sensitivity analysis. The specific process energy consumption increases by 8% if effluent qualities decrease by 20%. In contrast to this, no significant changes in recovery rates occur due to changing WWTP effluent qualities.

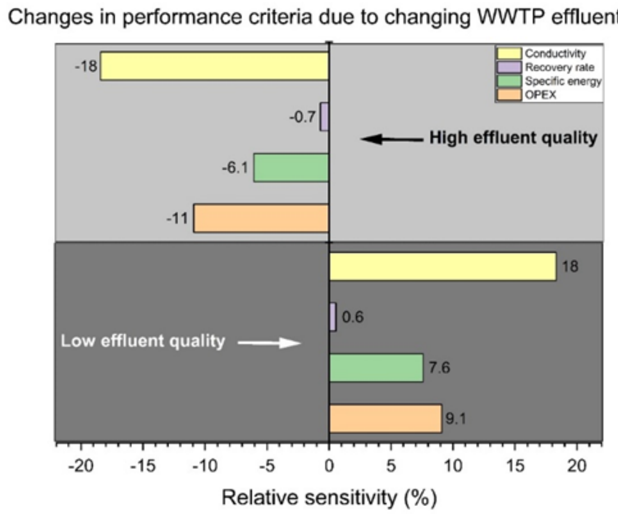

Figure 4.9 Sensitivity of WWTP effluent quality changes measured in Na^+, Cl^-, Mg^{2+}, Ca^{2+}, TSS, turbidity. High WWTP effluent quality represents 20% lower concentrations compared to the initially modelled WWTP effluent quality while low WWTP effluent quality represents 20% higher concentrations. Results are shown in % change.

4.4. Outlook and future research

Most studies in the field of WWTP effluent reclamation investigate potable or agricultural reuse possibilities but the results of this study suggest that industrial reuse is economically most attractive and deserves therefore more attention in future research. A process design approach that aims to improve RO performance is needed. RO pre-treatment steps, such as e.g. lime softening, and coagulation and flocculation could be more effective than only UF. They can better eliminate substances leading to scaling, organic and bio fouling. It is therefore important to understand how those formerly applied technologies worked and to challenge the current standardized approach of applying for example UF-RO-IEX subsequently for demi water reclamation. It is necessary to find new ways of how to integrate these three key technologies in the most efficient and effective way to improve robustness and recovery rates of RO systems (Slagt and Henkel 2019). This study indicates that the integration of a softener/biostabilizer could be very promising to improve the performance of RO driven water reuse and therefore this concept should be investigated further.

4.4.1. Fit for multi-purpose process design

The choice to design a MATP for a certain reuse type depends on the specific demands for reclaimed water (Garcia and Pargament 2015). The demand for a certain water quality can underlie high temporal variations (Wang et al. 2015b). For example, a major difficulty for agricultural reuse is that a varying demand of irrigation water throughout a year or vegetation period meets a relatively steady supply potential as WWTP effluent quantities are relatively steady. It could be increasingly necessary in the future to supply high loads of irrigation water in short drought periods that threaten harvest losses in summer due to increasing heat wave events (Buras et al. 2020). Therefore, it is necessary to study the usefulness of designing a 'fit for multi-purpose' MATP that can be adjusted flexibly to changing water demand patterns and reclaim different water qualities. Since this study is based on the idea that a fair comparison of MATPs for different reuse types requires a consistency in chosen unit operations in each process model, it shows also how many changes are needed to design a fit for multi-purpose instead of a fit for single-purpose MATP.

From figure 4.3 it becomes obvious that, if a demi or potable water reclamation process is installed at a WWTP already, it may be relatively simple to react on such a temporary urgent demand. Either RO or UF permeate (or both) needs to be abstracted from the process to supply irrigation water. If UF permeate is used, only a stand-by UV disinfection unit is required to treat it further to meet irrigation water quality regulations (table 4.3). The RO permeate instead has the advantage of being suitable to satisfy several non-potable applications with varying temporal demand patterns at once. Examples are firefighting, dust control or fish farm basin refilling (Garcia and Pargament 2015). Also the supply of additional water to river banks or other natural habitats that may fall dry in drought periods and lose their ecosystem services (Cazurra 2008) can be achieved with the irrigation water+. Other possible applications include landscaping or urban irrigation (Wang et al. 2015b), vehicle washing, recreational activities and street cleaning (Meneses et al. 2010). A fit for multi-purpose MATP design that could reclaim water of different qualities to supply it to various usage types and switch flexibly between them if necessary, would require that all unit operations studied in this paper are installed which implies higher initial capex. Another bottleneck to overcome would be the cost effective distribution of reclaimed water to its users possibly requiring costly infrastructure. Since WWTPs are usually located at the lowest point of a catchment area to use gravity flow, uphill pumping or transportation of reclaimed water is often necessary to reach the demand location (Lee et al. 2013). Nevertheless, a fit for multi-purpose concept could be a solution to ensure that wastewater is not only fully reused in a water stressed city or region but also that it is available at times and places where needed most and with the required quality.

4.5. Conclusion

This study contributes to better informed decision making in water reuse projects revealing differences in recovery rates, energy consumption and costs of MATPs designed for different reuse types (industrial, potable, agricultural) under Dutch conditions. The main findings are:

- Demi water for industrial purposes seems the most economically attractive reuse type in The Netherlands while the price for irrigation water is too low to reclaim it cost effectively. But this estimation needs further study and can probably only been addressed accounting for additional site specific cost factors, like e.g. water transport costs to users;
- High quality irrigation water (UF-RO) can reach a similar net present value as low quality irrigation water (UF-UV) if the difference in water quality is also reflected in the water price;
- In literature, high energy costs is often stated as the major bottleneck for water reclamation processes but when RO is applied waste management may be a considerably higher opex cost factor as brine disposal costs can be high. Therefore processes decreasing brine generation (as presented in the optimized demi water reclamation process) need further research attention. Also the integration of salt and water recovery technologies from brines seem economically interesting due to their disposal cost reduction potential;
- MATP process costs are mainly determined by the opex instead of capex and therefore processes with higher capex may be even more cost effective over a 20 year process lifetime (important for tender procedures);
- The integration of a softener/biostabilizer prior to RO may significantly improve a process's recovery rate, energy consumption and net present value. In addition it offers brine reduction potential as an environmentally important impact;
- Electricity recovery via anaerobic sludge digestion is not a solution for renewable energy integration to run MATPs because recoverable energy quantities are very small compared to the required energy. Solar energy integration seems feasible for flow rates of 100 $m^3\ h^{-1}$ in The Netherlands considering the required solar panel area.

5

The SPPD-WRF Framework: A novel and holistic methodology for strategical planning and process design of water resource factories

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“I have been impressed with the urgency of doing. Knowing is not enough, we must apply. Being willing is not enough, we must do.”

Leonardo da Vinci

5.1. Introduction

Implementing resource recovery from waste streams is a complex task and requires multidimensional planning and a whole-system perspective (Lacovidou et al. 2017). Domestic wastewater cannot any longer be considered as "waste" because it is a resource full of clean water, energy, and valuable materials including nutrients (Wan et al. 2016) and a sustainable municipal wastewater treatment plant (WWTP) recovers various resources from the wastewater stream and feeds into the circular economy (Verstraete and Vlaeminck 2011; van Loosdrecht and Brdjanovic 2014; Kehrein et al. 2020a). To emphasise the need for a paradigm shift towards resource recovery as a standard procedure in the wastewater sector, the term WWTP has been changed into WRF (water resource factory) (Aymerich et al. 2015). When existing WWTPs approach the end of their expected service life span, a unique window of opportunity exists to replace the aging infrastructure with innovative WRFs that integrate resource recovery technologies. Beyond the treatment of wastewater for safe discharge into the environment a WRF may reduce stress on water resources through water reuse, provide renewable energy, recover various products including fertilizers and contribute to economic activity (NSF et al. 2015). Reaching the transition from WWTPs to WRFs can mean a complete reimaging of the treatment process or modifying an existing process design by integrating innovative recovery technologies (Puyol et al. 2017).

Generally speaking, decision making in the early design stage of wastewater treatment processes is often based on previous experiences of the involved process engineers (Tchobanoglous et al. 2014). Usually, the overall objective of WWTP process design is to find the process among numerous alternatives that optimally treats a given influent under the prerequisite of process costs on the one hand and robust treatment performance driven by legal effluent quality requirements on the other (Hamouda et al. 2009; Bozkurt et al. 2017). Due to increasing available treatment technologies, WWTP design became more and more complex in past years. The resulting multi-criteria process design problem for an optimal treatment process has successfully been tackled by mathematical problem optimisation in superstructure methods using stoichiometric coefficients and kinetic constants (Alasino et al. 2007; Bozkurt 2015; Lu et al. 2017), or by applying computational environmental decision support systems in the process design procedure (Castillo et al. 2017). Software to choose between alternative treatment process designs on the basis of their techno-economic performance has been developed including also environmental impact assessment criteria (Garrido-Baserba et al. 2014). Despite these outstanding WWTP design methodologies, little attention has yet been given to design a WWTP from a resource recovery perspective instead (Wang et al. 2015a; Bozkurt et al. 2017). Although considerable interest in the technical development and integration of resource recovery technologies into WWTPs exists, resource recovery is not yet a major objective in WWTP design methodologies. The rationale and necessity to perceive wastewater as a resource has been emphasized intensively in the water sector but there are only few examples of actual WWTPs where an integrated design approach has been taken and multiple resources are recovered by a newly implemented process (Holmgren et al. 2016). The rather narrow decision making perspective limited to cost effectiveness and robust treatment performance excludes resource recovery as a crucial factor for a more sustainable urban water cycle because it blocks the transition from WWTPs towards WRFs (Guest et al. 2009).

The rapidly growing number of recovery technologies increases the complexity of process design choices further (Batstone et al. 2015). Consequently, designing WRFs in the future requires to make resource recovery a measurable process design objective. But the technical feasibility of certain resource recovery technologies alone does not guarantee their successful integration into innovative process designs because several existing non-technical bottlenecks need to be tackled as well to successfully implement available recovery technologies. Those bottlenecks relate to economics and value creation for recovered resources, but also to potential emissions and health risks and to policies and people's perceptions. If these bottlenecks remain unsolved, they impose severe uncertainties that may hinder decision makers in water utilities to implement WRFs in the future (Kehrein et al. 2020a). The design of WRFs requires therefore strategical planning from a multi-dimensional perspective and can only be successful if the design space and decision making process is opened up towards a multidisciplinary team effort. Expertise from multiple technical and non-technical domains need to complement the traditional process engineering tasks focused on treatment performance and process cost.

The goal of the framework proposed in this paper is therefore to establish resource recovery as a major objective in the design of new wastewater treatment processes that can justifiably be labeled as WRFs instead of as merely WWTPs. It aims to move forward from the concept that wastewater is a resource and answer the question: how to design or retrofit treatment processes from a circular economy perspective that cope with multidimensional site specific circumstances and are therefore ready for implementation? This requires a novel and holistic framework that combines strategical planning (SP) and early stage process design (PD) of innovative WRF processes that cope with:

- legal effluent standards,
- marketability and value creation for recovered resources,
- technical process feasibility
- economic feasibility
- environmental impacts
- and the inclusion of stakeholders into decision making procedures.

To fulfil these necessities the SPPD-WRF framework follows the project management principle "begin with the end in mind" (Leach 2005) and transfers it to WRF process design. It structures not only the process related design space of WRFs, but also the market related design space by introducing a production system perspective into the early design stage. WRFs ideally recover marketable commodities (van der Hoek et al. 2016; Stanchev et al. 2017; Kehrein et al. 2020a) and hence marketability related criteria like e.g. demands or monetary values of recovered resources need to be assessed early. The underlying rationale to assess the marketability of recovered resources is the following: the concept of a circular economy emerged from the observation of natural matter cycles that do not know wastes but only resources and aims to transfer those natural principles into societal production-consumption patterns. But the current market economy aims for perpetual short term economic value increment of individuals, organisations and states. Consequently, combining circular resource solutions with the current economic model requires that economic value is created from the cycle at a reasonable timeframe.

Resource recovery technology integration alters the overall technical, economic and environmental performance of municipal WWTPs. The proposed framework responds to these alterations by extending existing WWTP design and assessment methods and suggests a novel method suitable for WRFs. New technical challenges, like e.g. the integration of downstream processes and refinement steps of recoverable biomaterials are considered by including specific engineering expertise during process configuration and by introducing criteria to assess the resource recovery potential of a process. Compared to WWTPs, the economic performance of a WRF is no longer only determined by investment and operational costs but also by revenues from recovered resource sales while environmental impacts are determined not only by emissions but also by abated emissions that stem from avoided conventional resource extraction and consumption. The SPPD-WRF framework considers these new requirements and allows a complete assessment of technically feasible WRF processes in those non-technical dimensions. Since early stage process assessment relies greatly on secondary data, meaning any data that are examined to answer a research question other than the question(s) for which the data were initially collected, it is important to analyse the certainty of applied assessment criteria (Vartanian 2011).

The SPPD-WRF framework contributes also to the end-of-waste concept that has been manifested in the Waste Framework Directive 2000/98 to facilitate recycling by defining that waste is no longer perceived as such when it has undergone a recovery process in accordance with the following four conditions: (i) the substance or object is commonly used for a specific purpose; (ii) a market or demand exists for such a substance or object; (iii) the substance or object fulfils the technical requirements for its specific purpose and meets the existing legislation and standards applicable to products; and (iiii) the use of the substance or object will not lead to overall adverse environmental or human health impacts. Waste streams that are candidates for this change of definition must have undergone a recovery operation that complies with specific criteria which are yet to be defined for municipal wastewater (Saveyn et al. 2014). By introducing a holistic set of WRF assessment criteria, the SPPD-

WRF framework aims to facilitate the introduction of the end-of-waste concept into the municipal wastewater context.

5.2. Materials and Methods

Materials The SPPD-WRF framework for strategical planning and early stage conceptual process design of WRFs has been developed by reflecting on the multidimensional and multidisciplinary requirements for implementing WRFs successfully. It uniquely combines existing methodologies from different research fields: wastewater treatment and wastewater process design, industrial process engineering, project management, circular economy, market analysis, techno-economic assessment, environmental impact assessment. Firstly, an extensive literature analysis of these research fields was carried out. Secondly, procedures that are useful for designing WRFs from a holistic viewpoint have been identified in the literature. Thirdly, a step wise structure has been defined that allows strategical planning and process design of WRFs from a holistic perspective. Fourthly, quantitative and semi-quantitative assessment criteria to make innovative WRF process designs comparable to each other in different performance dimensions were adapted from its sources in literature.

The project management concept of a development funnel and stage gating (Quaglia 2013), which reduces the number of initially configured process designs gradually during successively executed assessments (figure 5.1), was used as a theoretic model for the assessment of WRF process designs. The original model was adjusted for those assessment dimensions useful to estimate the performance of WRFs. Each assessment represents a gate decision point and only when a process is assessed with promising results it may pass to the next assessment. When the results are unsatisfactory in a particular assessment, a re-design can be carried out to improve the process performance and feed it to the next assessment. If a process assessment result indicates that a process is unfeasible, it can be discarded. A process is first assessed for its marketability of recoverable resources because techno-economic and environmental impact assessments are in general more time consuming, thus more costly. Consequently, only those process designs that are promising in terms of value creation are suggested to be further assessed. After the marketability, technical feasibility, economic performance, and environmental impacts of innovative WRF processes has been assessed , an uncertainty analysis of applied assessment criteria is proposed. In addition, the framework includes stakeholder inputs in the formulation of process design objectives and during the final process selection step whereas the other steps, including technical process configuration and process assessments are supposed to be carried out by specialists of different expertise.

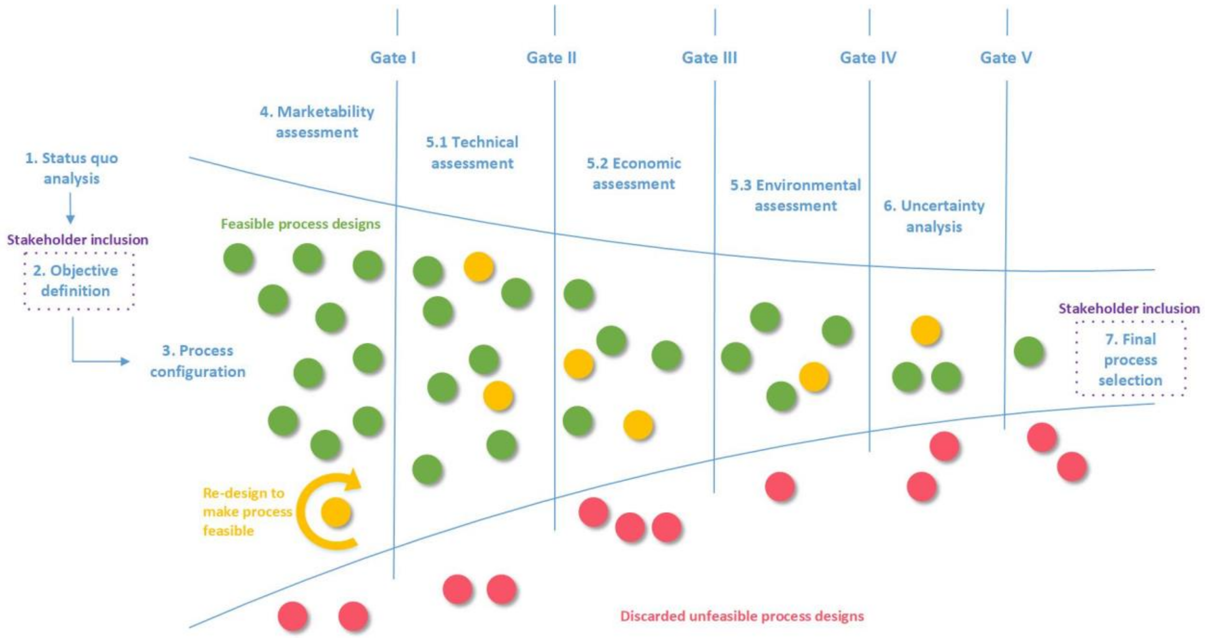

Figure 5.1 Funnel development and stage gating model adapted from Quaglia 2013 to make it specially applicable for WRF process design purposes.

5.3. Results and Discussion

The SPPD-WRF framework is a prescriptive methodology that incorporates seven consecutive steps including four WRF process assessment dimensions. In addition to the process related technical, economic and environmental dimension, also value creation and marketability of recovered resources were considered:

Step 1: status quo analysis

Step 2: design objectives definition with stakeholders

Step 3: process configuration and analysis by mass balances

Step 4: marketability assessment

Step 5: technical, economic and environmental assessment

Step 6: uncertainty analysis

Step 7: final process selection with stakeholders

A detailed overview of the framework is shown in figure 5.2. It structures the decision making process during the early stage planning and process design phase in a way that the finally selected process meets objectives carefully pre-defined by stakeholders, recovers marketable resources, is technically feasible, cost effective, and environmentally friendly. The framework provides a holistic but flexible instruction manual in the sense that proposed assessment criteria can be selected according to their usefulness for a specific WRF design project at stake while others can be added if necessary.

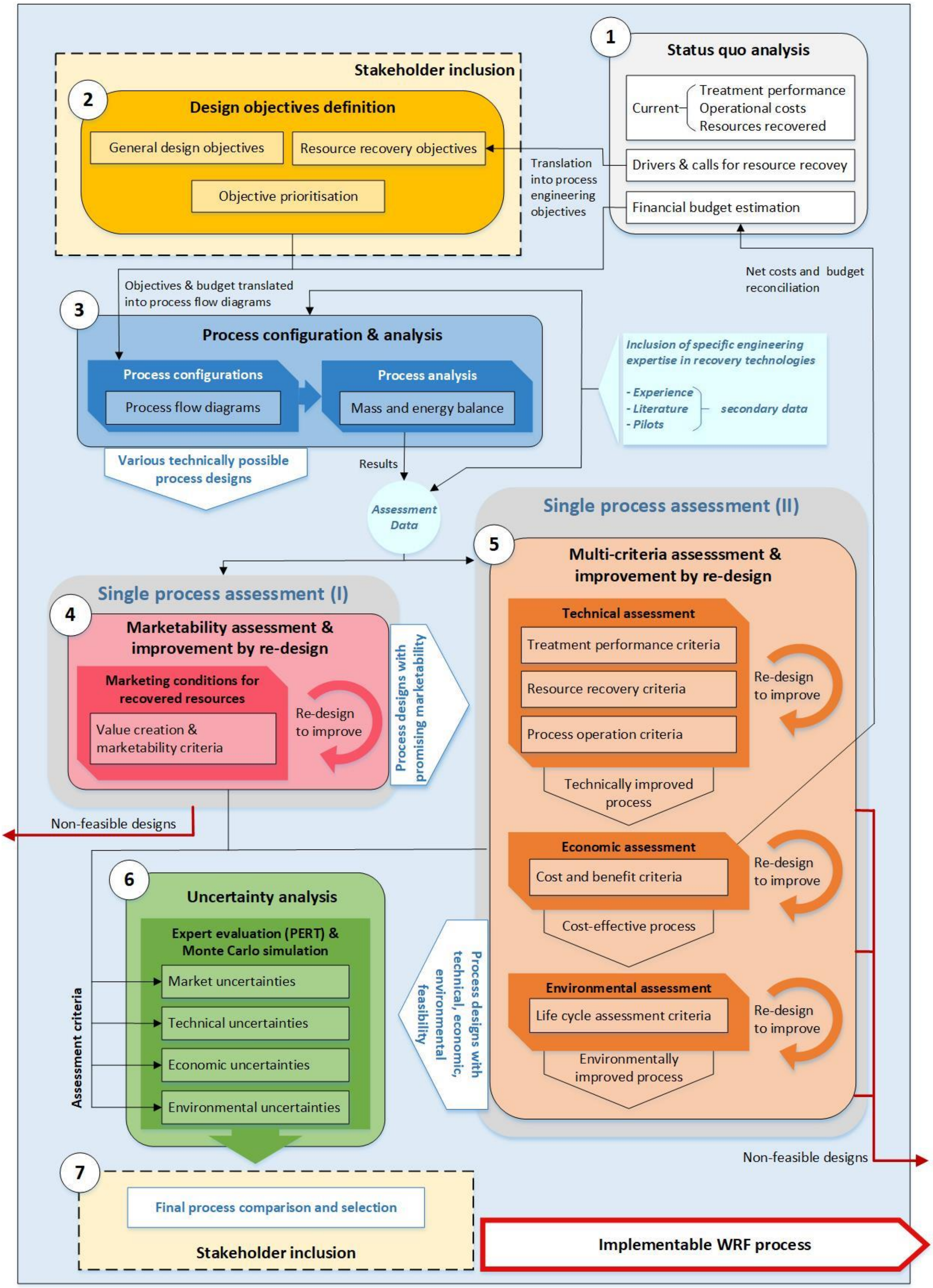

Figure 5.2 The SPPD-WRF framework for strategical planning and early stage conceptual process design of water resource factories.

5.3.1. Step 1: Status quo analysis

5.3.1.1. The existing treatment process

First, the current treatment performance can be measured by influent and effluent characteristics under consideration of legal effluent standards. Treatment performance measurement methods have already been developed for WWTPs and can be used at this step (Garrido-Baserba et al. 2014; Guerrini et al. 2016; Le et al. 2018). (Guerrini et al. 2016) recommends to measure 17 key performance indicators (KPIs) including that measure influent characteristics, treatment efficiency and effluent quality. Most key variables do not need to be measured in every treatment unit but can be calculated from influent and effluent concentrations by using existing stoichiometric models or simpler mass balance methods (IWA 2000; Henze 2008; Tchobanoglous et al. 2014). In addition to existing legal effluent discharge requirements, potential new legislation can be considered in this step. Emerging pollutant removal is a growing concern of legislators and stricter regulations which include new contaminant categories are expected in the future (Hendry and Benidickson 2017). Also stricter effluent nutrient concentration are debated which would require process designers to change treatment strategies and add further treatment steps to a process or re-design to meet the legal limits (Fatone et al. 2017).

The operational costs of the existing process can either be estimated based on pollutants removed, e.g. € per kg P (phosphorus) or on volume of wastewater treated (€ per m^3). Both measures may vary over time and therefore have advantages and disadvantages (Guerrini et al. 2016). In literature, cost estimations of WWTP operations usually consider five main operational cost factors: energy consumption, maintenance, waste disposal, labor, and reagents (Hernández-Sancho and Sala-Garrido 2009; Molinos-Senante et al. 2010). Furthermore, if the existing plant already recovers resources this can be assessed by determining the percentage of recovered energy and materials compared to influent concentrations (Guerrini et al. 2016). Table 5.1 provides a set of criteria that we suggest to apply to analyze the status quo of an existing WWTP.

Table 5.1 Criteria to assess the current treatment performance, operational costs, and resources recovered by the current WWTP based on the influent flow rate.

Treatment performance	Explaining remarks	Reference
Flow rate	Average influent load per day	(Vidal et al. 2002)
Effluent quality	COD removal efficiency	(Vidal et al. 2002)
	BOD_5 removal efficiency	(Vidal et al. 2002)
	TKN removal efficiency	(Vidal et al. 2002)
	TP removal efficiency	(Vidal et al. 2002)
Operational cost	**Explaining remarks**	**Reference**
Labor	Work-hours and wages (e.g. operation, training)	(Chong et al. 2016)
Energy	(Net-)energy cost for consumed kWh	(Garrido-Baserba et al. 2014)
Maintenance	Maintenance time and maintenance cost (e.g. repairs, inspection, replacements)	(Chong et al. 2016)
Waste management	Quantities, fees charged for disposal, transport costs (e.g. sludge)	(Garrido-Baserba et al. 2014)
Reagents	Costs of required chemicals (e.g. methanol, iron chloride, polyelectrolyte)	(Garrido-Baserba et al. 2014)
Resource recovery	**Explaining remarks**	**Reference**
Resource quantities	Quantification of already recovered water, energy, fertilizer, other products	(van der Hoek et al. 2016)

5.3.1.2. Budget determination

Since financing poses a perpetual challenge for the research, development, demonstration, and deployment of WRFs (NSF et al. 2015), it is necessary to determine the budget for a new process and its depreciation rate at an early stage as both are important boundary conditions to be considered throughout the conceptual process design phase. All funds that are likely to become authorized to plan and construct the plant, as well as the budget for operating the plant should be estimated here. This requires the analysis of funding opportunities like for example, European Union (EU) grants or public-private-partnerships (PPPs) using different forms of lease contracts (Molinos-Senante et al. 2017). Especially the investment costs allowed for the new WRF will be an important guideline in step three where process engineers configure possible process designs. Later in step five, process costs of WRF process designs will become estimated by cost benefit analysis (CBA) and the results need to be reconciled with the sum of authorized funds to identify any variance to the funding limit.

5.3.1.3. Drivers and calls for resource recovery

The concept of the Circular Economy is increasingly seen as a complete or partial solution to sustainable development and to the fulfilment of the United Nations sustainable development goals (SDGs) (United Nations 2017; Geissdoerfer et al. 2017). Since WRFs relate to several SDGs (table 5.2) which are also incorporated into European and national political objectives, it is useful to know how a WRF contributes to their fulfilment to trigger political support for the plans. In addition to sustainability as a driver, there might be actual calls for action for more circularity in the urban water cycle within the municipality or region where the WRF is planned. For example, calls for action for water reclamation and reuse could be sea water intrusion into groundwater (Van Houtte and Verbauwhede 2008), ecosystem service loss in dried out river banks (Cazurra 2008), or temporary water shortages in local agricultural systems (McCarty et al. 2011). If stricter legal effluent qualities are expectable in future wastewater regulations, this can be a call for implementing advanced treatment units that would also facilitate water reclamation if a process is accordingly designed (Høibye et al. 2008). The need for advanced treatment would imply higher energy requirements of the WWTP and therefore, integrating energy recovery technology into process designs to keep energy costs low, could be a follow-up call for action (Verstraete and Vlaeminck 2011). Another call for energy recovery could be that a governmental body puts pressure on the water utility at site to decrease greenhouse gas emissions of the urban water cycle and increase renewable energy consumption to contribute to political carbon emission goals, which has been the case in The

Netherland (Frijns et al. 2013). This would raise the call for e.g. recovery of chemical but probably even more the recovery of thermal energy stored in the wastewater stream (Kretschmer et al. 2016; Hao et al. 2019). Furthermore, policies that make it mandatory to recover P from sewage sludge, as required by German legislation valid from the year 2023, resembles a similar pressure that leads to a call for action of P recovery technology integration into the WRF process (Adam 2018). Given these reasons this step aims to analyze all existing or expected drivers and calls for resource recovery.

Table 5.2 UN sustainable development goals that a WRF may directly or indirectly contribute to.

SDG number	SDG
6	Clean water and sanitation
7	Affordable and clean energy
9	Industry, innovation and infrastructure
11	Sustainable cities and communities
12	Responsible consumption and production
13	Climate action

5.3.2. Step 2: Objective definition

5.3.2.1. Considerations on stakeholder involvement

After the status quo has been analysed, it is necessary to identify stakeholders of the new WRF and include them into the formulation of design objectives. In general, a broad range of stakeholders should be included into the process planning, especially experts like for example water utility managers, operators, WWTP process engineers, regulators, elected officials. But also non-experts, such as local neighbouring communities and public interest groups, as well as other parties affected by the new WRF need to be included in a process of stakeholder engagement. The importance of appropriately timed stakeholder participation in the decision making process has been acknowledged as a key component of socio-technological planning and design methodologies in water management projects because stakeholder participation is a vital component of embedding sustainable solutions and trust building (Guest et al. 2009). The key benefit of involving stakeholders already in the definition of objectives is that it allows to increase support and minimize resistance of critical stakeholders towards the process implementation (Project Management Institute 2017). Stakeholder participation facilitates positive social learning, minimizes and resolves conflicts, elicit and use local knowledge, and achieve greater acceptance of water management decisions (Guest et al. 2009). Nevertheless, as stakeholders can have different preferences regarding the new process, decision making can become more complex with an increasing number of stakeholders (Agudelo et al. 2007).

Therefore, it is debatable at which planning stage all stakeholders are included into the design process and may depend on project specific circumstances. It is very likely that the early stage process design team consists of a variety of experts that are aware of potential differing interests of non-expert stakeholders hence are able to define general process objectives from a well-informed viewpoint. Therefore, it might be feasible to follow the approach from (Larsen et al. 2010) who consider a full stakeholder inclusion after the general design objectives have been defined by a team of experts. Afterwards, other stakeholders need to be informed and included to have the opportunity to discuss defined objectives and possibly add others. In any case, the inclusion of non-expert stakeholders like the public, improves social sustainability factors like equity (Sjöstrand et al. 2018) or community involvement (Chong et al. 2016) and reveals which interest groups are made worse off or benefit more than others from the planned WRF. As different stakeholder analysis and participation methods exist, all showing unique weaknesses and strengths, we refer to the review of (Reed et al. 2009) to develop an approach most suitable for a specific planning case. However, we suggest that the following design objective definition includes all relevant expert and non-expert stakeholders and its results resemble commonly agreed goals.

5.3.2.2. Design objectives definition

As explained in the introduction, objectives in conceptual WWTP design are primarily based on investment and operational costs and treatment performance according to legal effluent discharge limits (Bozkurt et al. 2016). In addition, environmental impacts beyond the discharge of effluents into surface water bodies have been considered as important criteria in some WWTP design literature (Daelman et al. 2014; Naushad 2018). As the fulfilment of these three objectives remains also crucial in WRF design, resource recovery needs to be understood as an equally important, but additional design objective that can only be fulfilled if innovative processes meet the three initial objectives. Therefore, the SPPD-WRF framework proposes that the design space of WRFs is structured by these interlinked four key objectives:

- Legal effluent quality
- Economic feasibility,
- Environmental friendliness
- Resource recovery potential

The drivers and calls for resource recovery analysed in step 1 represent process requirements that need to be translated here into recovery objectives that will guide the process configuration carried out in step 3. For example, if a political will has been identified to decrease the municipal greenhouse gas emissions (GHG), decreasing the fossil energy consumption of the current WWTP by a defined percentage has to be formulated here as an objective for the new WRF. If a certain water reuse type has been identified as useful to solve a water supply issue within the regional water cycle, the requirements in quality and quantity of reclaimed water should be formulated as a clear objective here. Furthermore, the status quo analysis possibly revealed some weaknesses of the current WWTP which should be improved in the future by the new process and can be formulated as design objectives here. An example may be high operational cost factors due to high usage of reagents or large waste sludge volumes leading to high disposal costs. How the new WRF should perform in those identified criteria can be quantified here. Another objective defined here could be to design a process that fulfils a better removal of emerging pollutants if this was identified by stakeholders as a growing concern.

After the objective definition, a preliminary prioritization of objectives can be found together with all stakeholders. Several methods may be applicable to find a hierarchy for formulated objectives. To determine which resources should be prioritised over others if recovery measures compete with each other can be solved by a value pyramid that allows to rank recovered resources according to their economic value and estimated volume (van der Hoek et al. 2016). Operational cost reduction is likely to be a major objective in most process design projects which promotes the prioritisation of recoverable products of higher value over those with lower value. Therefore, the integration of recovery technologies that recover chemical oxygen demand (COD) as valuable biochemicals with high yields instead of low value methane might be formulated as a general objective (Kleerebezem et al. 2015). Another tool to find prioritisation of defined design objectives is the analytical hierarchy process (AHP) that reduces rather simple decision problems into pairwise comparisons assessed by weighted evaluation criteria (Taelman et al. 2018). The decision-making team, together with key stakeholders, assigns each objective a weight factor which reflects a particular objective's relative importance against the other objectives in percentage so that the sum of all weights adds up to 100% (Sjöstrand et al. 2018). However, ranking objectives may also be achieved by discussion in stakeholder meetings and many objectives may be priority over others simply because they are a prerequisite for the new process implementation, like for example being cost effective and fulfil legal effluent standards. Regarding the desirability to recover a certain resource, it should be argued that water recovery is preferable over energy recovery because it is more critical as energy which can be obtained from a great variety of alternative sources including renewables (Ma et al. 2015). Thermal energy recovery from effluents has recently been formulated as priority over chemical energy recovery as it has higher potentials to design carbon neutral processes. Instead of energy organic carbon should be recovered as biomaterials which preserves its high exergy content to a higher extend (Hao et al. 2019). For nutrient recovery, phosphorous recovery has been referred to as preferable over nitrogen recovery due to a projected scarcity (Khiewwijit et al. 2016). A further prioritisation of resources to be recovered will become clearer during individual process assessments in steps 4 & 5 where some resource recovery pathways will be discarded due to marketability constraints, techno-economic or environmental unfeasibility.

5.3.3. Step 3: Process configuration & analysis

5.3.3.1. Process flow diagrams

Wastewater treatment process configuration is the selection of technologies from numerous alternatives to interconnect them to possible process flow diagrams (PFDs) (Bozkurt 2015). The primary goal of this step is to configure a variety of process designs that use innovative technologies to recover all the resources that have been defined as recovery objectives in the previous step. Other already defined objectives related to effluent quality, technical-economic feasibility, and environmental friendliness are of secondary concern in this step. They will be thoroughly assessed in step 5 where each PFD can still be re-configured to match them. For now, process designers can creatively design innovative processes from a resource recovery perspective and simultaneously keep the boundary conditions like effluent quality and investment cost budget in mind but not focussing on them. Regarding the latter, the process design team can integrate unit operations already existing in the status quo process (analysed in step 1) in their new process configuration if convenient. Keeping existing and potentially useful unit operations may reduce investment costs which will be assessed later in step 5.

Like in other process engineering fields, the selection of technologies in wastewater treatment process configuration is often based on heuristics and experiences of responsible process engineers (Harmsen 2004; Fernández-Dacosta et al. 2015). But the configuration of PFDs from a resource recovery perspective must be a creative and innovative procedure that overcomes the habitual attempt to integrate dominantly technologies that are already well known by involved process engineers. There is a clear necessity to bring a broad range of experts from different technical disciplines together as not only treatment but also recovery technologies are integrated into the PFDs. The latter requires knowledge beyond the wastewater process engineering domain, like for example the production and downstream processing of biopolymers (Fernández-Dacosta et al. 2015) or the recovery process of protein from ammonia dissolved in side streams (Matassa et al. 2016). If water reclamation has been defined as an objective, know-how that is probably more commonly applied in the technical domain of potable water supply is required and should be represented in the process configuration team (Eslamian 2016).

To reach a creative design procedure the state of the art in applicable technologies should be reviewed using recent literature (Egle et al. 2016; Puyol et al. 2017; Kehrein et al. 2020a). This way missing expertise for certain processes intended to be integrated can be invited to join the configuration step. In addition, an outlook about the future potential of technologies that are not fully mature yet is recommended. Assessing innovative technologies according to when they can be operational in full scale is an important criteria for designing innovative WRFs (van der Hoek et al. 2016). Data and information to estimate the potential for up-scaling innovative technologies that are still on pilot scale can be collected from literature, communication to experts and researchers, and from pilot plants (Egle et al. 2016). The maturity of innovative technologies may be then systematically assessed using the technology readiness level method (TRL) which ranges from 1 (basic principles observed) to 9 (technology proven in operational environment) (DIN 2013). Some of the objectives defined in step 2 might be interrelated because they can be achieved by integrating the same technology. Those potential key technologies that are able to tackle several objectives should be identified and integrated into one or several PFDs. For example, if the objectives of waste sludge reduction and energy neutrality have both been formulated, primary sludge up-concentration with chemically enhanced primary treatment (CEPT) and subsequent anaerobic sludge digestion should be integrated into a PFD as this combination of technologies may tackle both objectives (Bdour et al. 2009).

5.3.3.2. Mass and energy balances

After a variety of PFDs have been configured to likely meet the defined recovery objectives, a better understanding of the processes and implications of specific technology integration is needed. This can be achieved by the method of mass and energy balances (MEBs) which are an excellent method to analyse how selected wastewater constituents convert in a process (Solon et al. 2019b). Selected wastewater constituents can be tracked throughout the PFD to quantify concentrations in targeted flows, like for example in waste sludge and effluent. Finally, the results will reveal products, wastes and emissions associated with the process in space and time. Modelling the mass and energy conversions in each process unit provides insight on how integrated treatment and recovery technologies may influence each other and the treatment performance. Thus, the understanding of how and where innovative resource recovery units preferably become integrated is enhanced (Egle et al. 2016). (IWA 2000; Tchobanoglous et al. 2014; Le et al. 2018) provide data and detailed procedures for the selection and identification of key variables of WWTPs to perform accurate balances of COD fractions, total suspended solids (TSS), total Kjeldahl nitrogen (TKN), or total phosphorous (TP). The functional unit of the MEBs should be the influent flow rate to make the MEBs of the different PFDs comparable to each other (Hernández-Sancho and Sala-Garrido 2009).

An integrated mass and energy balancing approach needs to include all unit operations that are planned for on-site resource extraction, processing, and refinement to products (Solon et al. 2019b). More specific data, like e.g. operating parameters of innovative technologies can be collected from scientific literature, from existing pilot- or full-scale applications, or from experience (Fernández-Dacosta et al. 2015). The balancing can be conducted by applying specialised process modelling software, like e.g. Aspen Plus but also more generic calculation tools can be used. Another reason why the analysis of configured processes with MEBs is a crucial step in WRF design is, that results will reveal for each PFD the recoverable resource quantities and therefore possible trade-offs between integrated recovery technologies in different PFDs are revealed. Influent constituents, like COD or P can be recovered as different products depending on the recovery technology applied in a process (van der Hoek et al. 2016). The integration of a particular recovery technology restricts others and therefore it should be analysed which PFDs convert constituents into which and how much products. Examples for these possible trade-offs between process configurations are cellulose recovery by influent sieving could reduce the methane recovery rate in anaerobic digestion (van der Hoek et al. 2016) compared to a process that recovers no cellulose from the influent. Whereas methane recovery in turn could reduce the recovery potential of biomaterials, like e.g. polyhydroxyalkanoate (PHA) (Kleerebezem et al. 2015). In addition to trade-offs, interrelations and synergies between particular resource recovery technologies will become more obvious by conducting MEBs. One possible interrelation relates to the water-energy-nexus. Water reclamation by advanced treatment processes, like e.g. membrane-based treatment of secondary effluents, may require 0.7 - 1.5 kWh m^{-3} additional energy input (Pearce 2008). Therefore, if water reclamation has been formulated as a priority it may also raise the need to simultaneously integrate energy recovery technology into the PFD to keep energy costs of the whole process as low as possible. How the prioritisation between energy and water recovery is made is case dependent. However, MEBs provide the basis for follow-up marketability, technical, economic, and environmental assessments that will reduce the number of feasible PFDs stepwise.

5.3.4. Step 4: Marketability assessment (Gate I)

The circular economy concept aims to transform value chains from linear to closed-loop to promote a more environmentally friendly use of resources. The implementation of such a green economy however needs solid business models that create value from wastes that are reintroduced into societal consumption (Ghisellini et al. 2016). Despite the technological possibility of a WRF that recovers various resources from a given wastewater stream, its implementation may still be hindered by market related bottlenecks which can introduce high uncertainties (van der Hoek et al. 2016). Consequently, after exploring possible PFDs and conducting MEBs to better understand their potential to recover resources and treat the wastewater, critical criteria for value creation and marketability of recoverable resources need to be assessed for each PFD. The reason to assess the marketability conditions of each PFD at a fairly early process design step is that technical, economic and environmental impact assessments require time and are costly and are therefore conducted in the next step. This way, only those processes showing promising marketability chances for the resources they recover will be assessed further while processes not promising regarding marketability can be discarded already. We suggest to first assess seven criteria that may hinder a successful implementation of a technically feasible process (table 5.3). The listed criteria should not be understood as final barriers but as challenges that have to be analysed and addressed to implement innovative WRFs. At this early stage in the conceptual process design phase, the possibility of re-designing and adjusting a PFD to meet the assessment criteria is still possible if necessary. The proposed marketability assessment criteria are predominantly qualitative. To make the marketability of different PFDs comparable to each other, each criteria can be scored according to its expected performance. The balanced scorecard model invented by (Kaplan and Norton 1992) is a useful multi-criteria analysis method as it has already been adapted to be applied in technical process design (Kralisch et al. 2018), and even been used in assessing WWTP performance (Guerrini et al. 2016). In the following each criteria is further explained.

Table 5.3 Criteria to assess the marketability and value chain development potential for recoverable resources. Each criteria can be scored according to its expected performance.

Value chain & marketability criteria	Explaining remarks	Reference
Demand	Quantifying and localizing demands for recoverable resources	(Pan et al. 2015)
Logistics	Analyzing distance, topography, and transport of recoverable resources to reach demand	(Yi et al. 2011)
Acceptance	Analyzing the customer acceptance for resources recovered from municipal wastewater	(NSF et al. 2015)
Legal situation	Analyzing regulations and policies that support or hinder the recovery of a resource	(McConville et al. 2014)
Supply potential	Estimating quantities of recoverable resources and relate them to the demand	(Kleerebezem et al. 2015)
Applications	Exploring and defining applications and utilization routes for recovered resources	(Kleerebezem et al. 2015)
Monetary value	Estimating the monetary value/market price of recoverable products	(van der Hoek et al. 2016)

5.3.4.1. Demand & logistics

As explained in the introduction, the end-of-waste concept proposes to recover resources from waste streams only if a clear market and a demand exists (Saveyn et al. 2014). Therefore, to proactively develop supply chains for recovered resources, the demand of potential customers needs to be quantified and located beforehand to decide on required capacities and scales of recovery units. Spatial and temporal demand patterns need to be analysed to decide for storage capacities and distribution lines for resources recovered from WRFs (Pan et al. 2015). The latter aspect is especially valid for water reclamation for non-potable reuse that may be demanded with hourly, daily, and seasonal variations. Also the distance to water users and the topographical location of the WRF which is usually build downhill to use gravitational flows, might require uphill pumping, long distance pipeline construction, or other complex and costly transportation solutions (Wang et al. 2015b). To compare the logistic requirements implied by each PFD, the location of potential customers of resources can be mapped in detail by using geographical information systems (Shandas et al. 2003). Demand side management is a method that can actively influence demand patterns of consumers to fit the process output. By engaging potential customers of recovered resources and analyse collaboratively logistic bottlenecks can help to find fit for purpose solutions (Pan et al. 2015).

Ideally the motivation of commercial customers to purchase recovered resources stems from a clear strategic advantage over conventional resources, like e.g. from an increasing uncertainty in conventional resource purchase (Geissdoerfer et al. 2018). If conventional resource markets are volatile, long term contractual agreements can provide security for commercial customers and therefore make them long term partners. Long term supply and demand contracts have been successfully established in other circular economy related fields, like e.g. in the waste-to-energy domain (Pan et al. 2015). Therefore, to approach potential commercial customers actively with the argument of long term price stability can be a valid argument for negotiating long term agreements which would in turn provide financial security for the WRF.

5.3.4.2. Acceptance of customers

Stakeholder engagement is an essential part in step 2 where objectives of the new WRF have been already reflected and agreed on. This must now be extended by including the viewpoints of potential customers of recoverable resources who are key stakeholders for a successful value chain development. Customers are either end-users or commercial companies that buy recovered resources to market them directly or to refine them further as a raw input material in their production process.

The acceptance of commercial customers and end-users is crucial as negative perceptions can be a major barrier for the successful marketing of resources recovered from wastewater (Yi et al. 2011). To shift customer perceptions from faecal matter containing wastewater towards positive associations with resource recovery, they should be actively engaged during the design of a WRF to raise acceptance. The general goal should be that customers associate the plans with human morality, human interaction, and a feeling of participating in their community (Balkema et al. 2002). It is likely that some potential customers have already been represented in step 2, like for example the local community that would need to accept and trust intended water reclamation strategies. The acceptance of local customers can be improved by engagement campaigns that inform about societal benefits of sustainable infrastructure in general but particularly of WRFs. If local end-users or commercial customers perceive the usage of locally recovered resources as contribution to their community, acceptance is likely to increase (Geissdoerfer et al. 2018). For example, to underline that a planned WRF could reduce GHG emissions or provide opportunities for local economic growth is important to make end-users feel supporting a more sustainable development of their direct environment (NSF et al. 2015).

If customers are local, nationwide or even global, several examples show that without the conviction of end-users and commercial customers about the harmlessness of wastewater derived resources it would be complicated for any water utility to locate, finance, develop and operate a WRF (Garcia and Pargament 2015). Only if customer needs are reflected and addressed early in the planning process, vital trust and support can be build (Dean and Fielding 2017). Furthermore, the end-of-waste concept could provide the opportunity to bypass the acceptance barrier because it allows to label recovered resources as full-fledged products instead of waste. As explained in the introduction, the end-of-waste status can only be applied if the recovery process of and the resource itself fulfil specified end-of-waste criteria. Therefore it should be assessed here what is required for each resource intended to be recovered to market it as a product without the usually required information of its origin.

5.3.4.3. Legal & policy context

The concept of waste, and its definition, is far from being obvious in different policies. The definition of waste as a non-useful material is not supporting the paradigm of the circular economy to perceive waste as a valuable resource. One aspect that is hindering trading and this the marketability of waste derived products is uncertain legal ownership of resources that are legally labelled as waste (Pongrácz and Pohjola 2004). The end-of-waste concept proposes to recover resources from waste streams only if they comply with existing legislation and standards (Saveyn et al. 2014). Therefore it is worth to include policymakers in the design process of WRFs and discuss to what extent recovered resources can be legally re-defined, what has to be done to do so, or which policies are missing in that context. Numerous examples show that legislation and policies including environmental regulations but also subsidies or other economic incentives can be discrete uncertainties for WRFs. For example, the application of P recovery technologies is especially relevant in countries where agricultural reuse of sewage sludge is restricted by legislation (Egle et al. 2016). Policy driven P recovery can be expected to become more important in the EU as various member states including Germany introduced legislation to enforce P recovery from WWTPs (Günther et al. 2018). The concern of legislators that recovered resources may be contaminated can prevent necessary legislation allowing their marketing. One example is the prohibition in the European Union of using microbial protein produced from ammonia that has been recovered from municipal wastewater (Alloul et al. 2018). Although the ammonia is stripped from side streams as gas and therefore the microbial protein production system can be completely decoupled from the actual wastewater flows (Matassa et al. 2015) it remains a legislative safety concern and hence forbidden. Concerns over product harmlessness have also been exemplified in the long process of changing the Dutch fertilizer act to allow struvite marketing in The Netherlands (van der Hoek et al. 2016).

These examples show that identifying and addressing legislative barriers and develop strategies to overcome them is crucial to successfully implement WRFs (Stanchev et al. 2017). Addressing potential policy and legislative shortcomings for recoverable resources early is important because clear policies provide economic security for investors of recovered resource value chains. Therefore, it is important to include policy makers into the stakeholder meetings and work collaboratively on necessary regulative changes. For example, it has been reported that subsidies can help to overcome financial unattractiveness of water reclamation which can be severe although a clear need for regional water reuse exists (Yi et al. 2011). Making decisions in WRF design despite complex and uncertain policy situations could be achieved by the method of dynamic adaptive policy making which has been proposed for WRF implementation in the Dutch capital of Amsterdam. It follows a step wise problem analysis, measure identification and solution strategy to bring tailor made policies on the way that enable the recovery and marketing of wastewater derived resources (van der Hoek et al. 2016).

5.3.4.4. Supply potential & applications

Compared to conventional production systems, only little quantities of a resource may be recoverable by a WRF. This can be due to low yields of a recovery process, low resource concentrations in the side streams emerging from a process, or low overall resource quantities contained in the influent. Compared to industrial productions, the scalability of production is not given in waste based systems and therefore recovered product quantities might be comparably small which can be a severe economic disadvantage (Kleerebezem et al. 2015). The MEBs conducted in step 3 provide quantities of all resources that are recoverable by a process and can now be compared to the identified demands that are satisfied by conventional suppliers. This way it becomes clearer what supply potential the particular process can reach for a recovered resource and if it is competitive as a supplier in terms of quantities. If commercial demanders cannot become supplied sufficiently, the exploration of niche applications, as described by (Tamis and van Loosdrecht 2015) for the case of biodegradable PHA as a niche within the biodegradable plastic market, may be crucial for certain resources to be competitive.

Combining wastewater fractions with other similar waste streams could increase the recoverable quantities of a product and therefore benefit from economy of scale (Kleerebezem et al. 2015). Therefore, the possibility and usefulness of receiving other waste streams than municipal wastewater and act also as a waste disposal facility should be investigated here. Especially organic waste streams arising at different parts in the food supply chain could be processed in combination with the organic fraction of the wastewater. In Europe ca. 20% of the total food produced is wasted (Stenmarck et al. 2016) and the scientific community elaborating circular solutions for food waste identified the same products as the wastewater resource recovery community like e.g. volatile fatty acids or biopolymers (Pfaltzgraff et al. 2013). It has also been shown that co-digestion of sludge and organic municipal wastes to recover methane provides both environmental and economic benefits (Nghiem et al. 2017). As explained above, the acceptance of people for products recovered from wastewater can be a severe bottleneck. The range of possible applications for a recovered resource can be limited by the perception of stakeholders and therefore applications that are not affected by this issue need to be explored creatively. For example, the marketing of consumer products that contain wastewater derived resources is a challenge and therefore other applications should be preferred (Verstraete and Vlaeminck 2011).

5.3.4.5. Monetary value

Estimating the monetary value respectively the market price of each resource recoverable by a process design is a necessary step for the cost-benefit analysis following in step 5. Together with the quantities estimated by MEBs in step 3, the monetary value defines expectable revenues. Estimating market prices may give a first indication about which resource is preferably recovered compared to another. For example, given that the value of electricity and heat recovered by anaerobic COD digestion and subsequent methane combustion might be very low, alternative COD recovery technologies that lead to higher value products, like e.g. biochemicals might be recognized as preferable (Kleerebezem et al. 2015; Puyol et al. 2017). (van der Hoek et al. 2016) provides an orientation for decision making in accordance with monetary values of recoverable resources by introducing a value pyramid showing which recovery pathways are preferable over others in conflicting situations. The method suggests to estimate the value of recovered resources by considering monetary values, recoverable volumes and markets targeted, like e.g. the health and lifestyle market or the energy market.

5.3.5. Step 5: Multi-criteria assessment

After each PFD has been analysed with MEBs and the marketability potential of the resources it aims to recover has been assessed, process designs with promising marketability chances will be assessed further in this step to estimate their performance in the technical, economic and environmental dimension. It has been proclaimed that the fundamental shift from treatment towards resource recovery promises reduced operating costs and lower emissions (Hering et al. 2013). But a WRF also likely leads to new cost factors and emissions that are not existing in conventional WWTPs. Therefore, each PFD needs to be individually assessed considering the whole process by using existing methodologies and applying a range of performance criteria. The first assessment aims to provide insights about the technical performance of a process and only technically feasible process designs will then be assessed further by CBA to discard economically unfeasible designs. Finally, the environmental impacts of those process designs showing promising technically and economical performances will be assessed. During each of the three assessments, there is a chance to re-design a process if possible so that it copes with those criteria that turn out to be yet unsatisfying.

5.3.5.1. Technical process assessment (Gate II)

As described above, the shift from WWTPs towards WRFs requires that resource recovery technology integration becomes a central objective in process design, hence technical decisions need to be guided to meet both treatment- and recovery requirements. Therefore, the technical assessment will assess the performance of a process in the dimensions (i) treatment, (ii) operation and (iii) resource recovery. The MEBs conducted in step 3 provide the basis for estimating various technical performance criteria that are presented in tables 5.4 & 5.5. The treatment performance of new processes is usually assessed by removal efficiencies of pollutants that define the legal effluent quality, like e.g. COD, TKN, or TP (Vidal et al. 2002). Since the environmentally safe release of treated water into surface water bodies remains the primary goal of designed processes each process design has to be assessed accordingly. It is possible that effluent quality requirements will become stricter and ask for more indicator substances in the future. If that is expected the treatment performance assessment can be extended by estimating the capability of a process to fulfil potential future legal requirements for micro-pollutants (Høibye et al. 2008) and/or nutrient removal (Fatone et al. 2017).

The integration of resource recovery technologies into treatment plants can imply operational uncertainties (Quaglia 2013) and therefore the question about which technologies are most useful and how to combine them in process design has to be tackled (Li et al. 2015). Therefore, in addition to the treatment performance, the operation of each process design can be assessed. There have been criteria proposed and established for treatment process operation assessment which can be applied here (Agudelo et al. 2007; Bozkurt et al. 2016; Guerrini et al. 2016). Operational data and information for single process units, like e.g. sedimentation tanks or bioreactors, can be extracted from the vast literature available on particular treatment technology operations (IWA 2000; Henze 2008; Tchobanoglous et al. 2014). Operational data for more innovative resource recovery units can be extracted from specific articles describing pilot or case studies. Searching the term "wastewater resource recovery technology" in an online search engine for scientific publications showed over 243 thousand results in 2020.

After assessing the treatment performance and operation, each process can be assessed regarding its resource recovery performance which includes criteria to assess the expectable quality of recovered resources on the one hand, and the recovery efficiency of the process on the other hand. The end-of-waste concept proposes to facilitate recycling by defining that waste is no longer perceived as such if it has undergone a recovery process that ensures the use of a recovered product will not lead to overall adverse human health impacts (Saveyn et al. 2014). Several studies have shown that resources recovered from municipal wastewater may be of uncertain quality or even contaminated which may impose health risks. For example, struvite has been reported to possibly contain heavy metals (Xie et al. 2016) and reclaimed water may contain harmful by-products of chemical biocides used in tertiary water treatment (Zanetti et al. 2010). Studies examining health risks in the field of circular economy seem mostly to deal with occupational health risks which relate to the work place and less with health risks arising from recovered product use. In the context of WRFs, the only resource that has been subject to extensive risk management considerations is reclaimed water. Risks and therefore quality requirements for reclaimed effluents depend on the intended reuse type. Biological and human health safety control measures need to be proactively developed as legal standards are often missing for each reuse type (Wang et al. 2015b). Major concerns are pathogens that can cause acute infections at very low dose upon exposure but also chemical micro-pollutants need to be removed from the water for safe reuse (Jiménez Cisneros and Asano 2008). A systemic risk management approach that covers all aspects of the reclaimed water production, distribution and utilization has been proposed but needs further elaboration and a proactive management approach (Yi et al. 2011; Paranychianakis et al. 2015). To resolve legal uncertainties the European Commission recently proposed minimum risk control standards for water reuse that can provide guidance for designing advanced treatment processes for safe to use reclaimed water (European Commission 2018). The quality of a recovered resource should not only be safe for human health but also competitive with conventional alternatives on the market. After potential applications and customers associated with a process have been analysed in step 4, it is now important to assess whether a process can cope with the quality requirements of both. This can be a challenge as it is one thing to recover a product from wastewater but another to obtain a marketable quality from the recovery process. For example, it is technically feasible to recover biochemicals like e.g. volatile fatty acids (VFAs) or PHA from COD but, to obtain a certain purity requested from the industry is a technical challenge in mixed culture systems (Puyol et al. 2017) thus has to be considered from the early design stage on.

Next to resource quality the resource recovery performance of a process can be assessed in terms of recovery efficiency. The mass efficiency of process designs can be assessed with the results of the mass and energy balances conducted in step 3. For example, if two different processes recover P but process (A) integrates struvite crystallisation (Li et al. 2019) while process (B) integrates magnetic extraction of vivianite (Prot et al. 2019), it is useful to assess the recovery rate of influent-P of each process. Furthermore, if resource recovery technologies include not only the extraction but also on-site refinement of a recovered product, KPIs originally developed for the chemical or pharmaceutical process assessment could be applied to assess the performance of a particular recovery technology in comparison to an alternative one. For example, the chemical extraction of extracellular polymeric substances (EPS) from aerobic granular sludge is possible with different solvents (Felz et al. 2019) and could be assessed by the solvent score method (Kralisch et al. 2018).

Table 5.4 Criteria to assess the expectable technical performance of a process in the dimensions treatment and operation

Treatment performance criteria	**Explaining remarks**	**Reference**
Effluent quality	see table 5.1	(Vidal et al. 2002)
Future effluent quality	Capability to fulfil potential future legal requirements for micro-pollutants	(Hendry and Benidickson 2017)
	Capability to fulfil potential future legal requirements for nutrient removal	(Fatone et al. 2017)
Process operation criteria	**Explaining remarks**	**Reference**
Sludge treatment efficiency	Capability of a process to reduce suspended solids in sludge and its water content	(Guerrini et al. 2016)
Sludge production	The expected weight of excess sludge	(Guerrini et al. 2016)
Reagent efficiency	Estimate scores for chemicals used (e.g. metal-salts, polymers, methanol etc.)	(Guerrini et al. 2016)
Energy efficiency	Expected net kWh to be consumed by the process (consumed - recovered)	(Guerrini et al. 2016)
Exergetic efficiency	Energy useful for work in the process	(Kralisch et al. 2018)
Process safety	Potential hazard estimation (e.g. explosion risk)	(Chong et al. 2016)
Process robustness	Based on failure records, problem frequency, reliability of existing processes	(Agudelo et al. 2007)
Process flexibility	Susceptibility to shock loads of certain constituents	(Vidal et al. 2002)
Equipment wear	Equipment wear (based on e.g. operating hours, speed, load, start-up's)	(Lindberg et al. 2015)
SRT	Sludge retention time in days	(Bozkurt et al. 2017)
HRT	Hydraulic retention time in hours	(Bozkurt et al. 2017)
Reactor volume	Required sizes of units based on SRT and HRT	(Bozkurt et al. 2017)
Contextual independence	Influence of external factors on the process performance (e.g. seasonal temperature)	(Agudelo et al. 2007)

Table 5.5 Criteria to assess the expectable technical performance of a process in the dimension resource recovery.

Resource recovery assessment dimension	Recovery criteria	Reference
Recovery process operation	Reliability of recovery process	(Castillo et al. 2017)
	Process control requirements	(Castillo et al. 2017)
	Operation simplicity	(Castillo et al. 2017)
	Compatibility between different units (e.g. with treatment units)	(Castillo et al. 2017)
	Process flexibility to certain parameters (e.g. oxygen availability, reaction time)	(Castillo et al. 2017)
	Need for skilled staff	(Castillo et al. 2017)
Quality of recovered resources	Reclaimed water (e.g. bacterial and viral indicators)	(Zanetti et al. 2010)
	Energy carrier (e.g. methane generation rate in biogas)	(Appels et al. 2008)
	Fertilizer (e.g. macro-nutrient content and plant availability, struvite contamination)	(Egle et al. 2016)
	Products (e.g. controlling the product spectrum in open culture volatile fatty acid fermentation)	(Kleerebezem et al. 2015)
Recovery efficiency	Mass efficiency (e.g. fraction of effluent that is reused)	(Kralisch et al. 2018)
	Mass intensity (mass of external raw materials in per mass of desired product out)	(Jimenez-Gonzalez et al. 2011)
	Share of recovered energy used on-site	(Schmidt et al. 2016)
	Solvent score of a recovery unit	(Kralisch et al. 2018)
	Number of redox changes in a recovery unit	(Kralisch et al. 2018)

5.3.5.2. Cost-benefit analysis (Gate III)

After the technical assessment has been completed and processes with a technically unsatisfying performance have been either re-designed for improved performance or discarded as unfeasible alternative, the economic performance of technically promising designs needs to be assessed. The European water framework directive demands that urban water systems should be economically self-sustained, meaning that costs should be covered by the system itself through water pricing and service fees for wastewater treatment (Castillo et al. 2017). Taking the latter into account, a WRF has a different 'business model' compared to conventional WWTPs. Although its priority remains to treat wastewater and charge a fee for this environmental and human health service provision, a WRF ideally also generates revenues from recovered resource sales (Li et al. 2015). Resource recovery introduces new financial uncertainties and leads to a whole different cost-benefit structure within the wastewater treatment equation (Lee et al. 2013; van der Hoek et al. 2016; Molinos-Senante et al. 2017). Not only additional investment costs are likely to occur from required recovery units and installations but also substantial changes in operating costs can be expected if processes are designed from a resource recovery perspective. For example, integrating chemically enhanced primary treatment into a process that uses anaerobic sludge stabilization may alter the process economics in various ways. On the one hand it promises economic benefits from increased methane yields and decreased aeration requirements for the biological treatment. On the other hand it requires the usage of special polymers that represent an additional cost factor (Solon et al. 2019b). Another example would be the reclamation of reusable water accomplished by advanced membrane-based treatment. The additional treatment step leads on the one hand to additional operational costs for energy consumption (Frijns et al. 2013), maintenance costs to prevent membrane fouling (Rao 2013) and waste management costs for retentate handling (Pérez-González et al. 2012), but on the other hand it may generate steady economic benefits in form of revenues from water customers (Molinos-Senante et al. 2011). Furthermore, P recovery requires additional investment costs but may also lead to lower waste management costs due to improved sludge dewaterability and benefits from recovered product marketing (Prot et al. 2019).

These examples show clearly that a reliable statement about the economic feasibility of a WRF can only be made if its combined costs and benefits are assessed. Therefore, this step aims to conduct a cost and benefit analysis which is a systematic approach for estimating and comparing positive and negative economic consequences of an investment to determine its net profitability. It follows the simple formula (1):

$$NP = \sum Bi - \sum Ci \tag{1}$$

Where NP is net profit, B is benefits of item i, and C is costs of item i (Molinos-Senante et al. 2010).

To assess the economic performance of processes, the time horizon needs to be defined that the new process is expected to be operated in. A 20 year time horizon to calculate costs and benefits associated with wastewater resource recovery processes has been suggested (Lee et al. 2013; Fernández-Dacosta et al. 2015) but it may be defined more accurately in accordance with site and project specific circumstances. The economic performance of different process designs can only be compared if the same time horizon is applied in the assessment so that they have the same time to accumulate costs and benefits (Boardman 2014). The costs and benefits that will arise during the expected time horizon are calculated for each process by using the net present value (NPV). The NPV expresses the monetary value of future cash flows and is an indicator to determine the economic value of a process design and thus allows to rank alternatives. The higher the NPV, the more economically favourable a process design is. The NPV method requires to determine a discount rate that accounts for the opportunity costs of time by discounting future costs and benefits because of the profit that could be earned in alternative investments. It is a wide spread practice in CBA to use the current market interest rate as discount rate (Boardman 2014). The discount rate calculation is shown in formula (2). To calculate the NPV, the time horizon is usually divided into yearly periods and net profits are discounted and calculated on a yearly basis which leads to formula (3).

$$r = 1/(1+i)^n \tag{2}$$

Where the discount rate r equals the present value of 1€ received in n years when the interest rate i is compounded annually (Boardman 2014).

$$NPV = \sum_{t=0}^{n} \frac{NP_n}{(1+i)^n} \tag{3}$$

Where the net present value (NPV) at time t, calculated for a time horizon of n years, is the sum of discounted annual net profits (NP) assuming a discount rate i. Adapted from (Boardman 2014).

Table 5.6 shows cost and benefit factors to be included in CBA for a WRF process. The method required that all cost and benefit factors related to a process design are estimated for each year of the time horizon. The annual net benefits then need to be discounted and summed up to obtain the NPV of the process. To avoid the need of predicting future price level changes by e.g. inflation, costs and benefits should be expressed in current real prices and not with nominal prices (European Commission 2015b). Many occurring cost and benefit factors can be deduced from market prices or estimated by using literature studies or expert judgement (Egle et al. 2016).

Like in most infrastructure projects a high initial total investment cost occurs at the beginning of the time horizon when the new process is planned, purchased and constructed. A residual value of the fixed investments must be included within the investment costs occurring in the last year if the plant is believed not to be liquidated after the time horizon has ended. It reflects the capacity of the remaining service potential of fixed assets which are not yet completely exhausted (European Commission 2015b). Operational costs of wastewater treatment are usually measured on the basis of contaminant removal (Hernández-Sancho and Sala-Garrido 2009) which requires to take the influent characteristics into account that have been defined in step 1. Some cost factors may require general assumptions to keep a certain degree of simplicity during this early stage economic assessment. For example, the electricity needed to supply oxygen into biological treatment units depends on the saturation concentration of oxygen at an assumed temperature, pressure and salinity (Bozkurt et al. 2017). Expectable benefits from resource sales can be calculated based on the quantities of recoverable resources that have been estimated for each process design in step 3 by MEBs and their market prices analysed in step 4. In addition to resource sales, benefits could also be gained from charges for handling additional waste streams as explained in step 4, like e.g. food wastes.

Costs and benefits are often defined in CBA as decreases and increases in human wellbeing which can include various external effects of human behaviour that have no market prize (Molinos-Senante et al. 2010; Sjöstrand et al. 2018). Although the monetization of these effects can be achieved by different methods (Boardman 2014), the cost and benefit factors suggested in table 5.6 do not account for external effects, like e.g. the cost of undesirable effluent constituents entering surface water bodies. The reason is that the monetisation of external effects is a complicated procedure usually conducted by experts like e.g. environmental economists. This framework is supposed to be useful for institutions and decision makers in the wastewater sector that do not necessarily have a strong expertise in those specialisations. The environmental impacts of a process will be carefully assessed in the next assessment stage. After the CBA has been completed for each process design, the results can be reconciled with the budget estimation conducted in step 1 to identify any variances between the available budget and the estimated costs of a process.

Table 5.6 Template for a complete cost and benefit analysis of water resource factory processes.

Investment costs (IC)		**Year 1**	**Year n**	**Reference**
IC1.	Planning			(Chong et al. 2016)
IC2.	Land acquisition			(Castillo et al. 2017)
IC3.	Right of way			(Lee et al. 2013)
IC4.	Installations e.g. buildings, reactors, pumps			(Chong et al. 2016)
IC5.	Construction and engineering			(Chong et al. 2016)
IC6.	Contingency			(Lee et al. 2013)
IC7.	Resource value chain creation (non-process assets)			
a)	Vehicles to transport recovered resources			(Chong et al. 2016)
b)	Pipelines to customers e.g. reclaimed water			(Yi et al. 2011)
IC8.	Residual value (last year)			(European Commission 2015b)
	Total investment costs			
Operational costs (OC)				
OC1.	Labor			(Chong et al. 2016)
a)	Operators			
b)	Marketing and sales experts			
OC2.	Energy required from grid e.g. electricity for aeration			(Castillo et al. 2017)
OC3.	Maintenance			(Lee et al. 2013)
a)	Inspections, repairs			
b)	Replacements e.g. membranes			
OC4.	Waste management			(Castillo et al. 2017)
a)	Sludge disposal			
b)	Resource recovery related wastes e.g. brines			
OC5.	Reagents & Raw materials			(Garrido-Baserba et al. 2014)
a)	For treatment e.g. iron-salt, coagulants, carbon			
b)	Resource recovery unit inputs e.g. acids			
c)	Packaging of recovered resources			
	Total operational costs			
Periodic costs (PC)				
PC1.	Cost of financing e.g. interests			(European Commission 2015b)
PC2.	Fees e.g. to regulatory authorities, insurances			(Chong et al. 2016)
	Total periodic costs			
Benefits				
B1.	Service fees paid by the public for treatment			(Molinos-Senante et al. 2017)
B2.	Subsidies for treatment or resource recovery			(Chong et al. 2016)
B3.	Resource value creation			(Lee et al. 2013)
a)	Sales revenues resource x			
b)	Sales revenues resource y			
c)	Sales revenues resource z			
B4.	Charges for disposing other waste streams			(Nghiem et al. 2017)
	Total benefits			
	Annual net benefits			
	Discount rate			
	Annual discounted benefits			
	Net present value			

5.3.5.3. Environmental impact assessment (Gate IV)

Finally, after the technical and economic assessment has been completed and those process designs that did not perform well in one or both dimensions have been either re-designed for improved performance or discarded as unfeasible alternatives, the environmental impact of promising process designs can be assessed. It has been shown, that overall, the environmental impact of WWTPs can be decreased through resource recovery implementation (Wang et al. 2015a). The growing possibilities in recovery technology integration into treatment processes implies that identifying the most environmental friendly process alternative requires careful impact assessment. Life cycle assessment (LCA) is a comprehensive and well established method to analyze the environmental impact of products, services and processes. The assessment embraces the entire system involved in the production, use and disposal of a product or service under investigation. All environmentally relevant substances emitted, as well as extracted natural resources can be identified and quantified in a "cradle to grave" approach (Baumann and Tillman 2004). LCA allows therefore to make environmentally beneficial decisions at an early design stage and compare process designs regarding their impacts. The execution of an LCA should follow a standardized methodology provided by The International Organization for Standardization's ISO 14000 &14040 (Corominas et al. 2013). A recent review of LCA studies conducted for domestic wastewater treatment plants since year 1990 concludes that the development of guidelines and standards is necessary to further shape a consistent LCA methodology for the field. For example, different functional units which serve as reference units in LCA are used in different studies which aggravates a comparison of already assessed treatment processes in the environmental dimension (Sabeen et al. 2018).

Since a WRF is in addition to a treatment process also a production system, the system boundaries of a WRF process LCA will differ from a WWTP LCA because the recovered and successfully marketed resources avoid conventional production of similar goods (Fang et al. 2016). The inclusion of these presumable positive impacts is achieved by the so called approach of consequential LCA. To include the avoided impacts of substituted conventional goods into the assessment, assumptions have to be made on how they are produced. LCA databases provide readily defined impacts for a wide range of different conventional products and materials (Foley et al. 2010). Other needed life cycling inventory data can be collected from published studies in the field of WWTP LCA (Corominas et al. 2013; Sabeen et al. 2018). Already available LCAs that include impacts associated with wastewater resource recovery mostly assess both energy recovery (Mills et al. 2014; Mu et al. 2018) and/or fertilizer recovery (Moreira et al. 2018; Sena and Hicks 2018) as a consequence of different sludge handling technologies. In addition, the environmental impacts associated with nutrient recovery by aquatic species, like e.g. algae (Mo and Zhang 2013) and impacts associated with water reclamation and reuse have been a focus in wastewater treatment related LCAs (Pasqualino et al. 2011).

A conceptual scheme including relevant LCA impact categories of an LCA that assesses the combined impacts of operating a WRF process is drawn in figure 5.3. It has been shown that impacts of WWTPs are mainly occurring in the impact categories (i) eutrophication and (ii) ecotoxicity in effluent receiving water bodies, and (iii) global warming potential (GWP) due to sludge handling and electricity use (Corominas et al. 2013). In the following we shortly describe those three impact categories but also which other aspects need to be taken into account when assessing the environmental impact of a WRF process. Since there are a variety of possible negative but also positive environmental impacts, only a complete assessment provides an overview of how environmental friendly a process design performs in a certain impact category and in total. For more detailed information about proposed impact categories we refer to (European Commission et al. 2011) who provide evaluation criteria and other requirements related to their application in LCA.

The impact category "eutrophication potential" is determined by the effluent concentrations of COD, P, and N that are released into the receiving water body (Agudelo et al. 2007). They have ideally been estimated by the MEBs during step 3. Heavy metals and micropollutants responsible for ecotoxicity are probably more difficult to estimate due to the early design stage but especially in process designs that apply advanced treatment steps for indirect water reuse, the ecotoxicity impacts can be expected to be significantly lower compared to processes applying only secondary treatment (Jegatheesan et al. 2013).

The most important emissions for the impact category "global warming potential" relate to direct GHG emissions of CO_2, CH_4, and N_2O which can occur at different treatment and resource recovery unit operations. Direct CO_2 emissions from aerobic biological wastewater treatment are considered by the Intergovernmental Panel on Climate Change (IPCC) of biogenic origin and therefore can be excluded in GWP estimations of WWTPs (Lebrero et al. 2017). But this might not be completely true as (Law et al. 2013) showed that 4-14% of total organic carbon in municipal wastewater is from fossil origin, namely from synthetic products used in industrial and residential products. It is therefore debatable if this minor fraction of the total direct CO_2 emissions should be included in an LCA. WRFs may not only emit CO_2 but also sequester carbon as products recovered from COD, like e.g. cellulose fibers may store carbon long term when used as composite construction materials or in other long lasting applications (Visser et al. 2016). Even waste sludge that is finally disposed in landfills has already been accounted as a carbon sequestration method (Rosso and Stenstrom 2008).

Furthermore, in contrast to direct CO_2 emissions, one has to account for direct CH_4 emissions as it may represent a significant share of the overall GHG emissions of WWTPs hence of WRFs. On a time scale of 100 years CH_4 has a 28 times higher GWP relative to CO_2 (IPCC 2014) and therefore WRF process designers should be aware of potential direct emission sources and take preventing measures. To assess direct CH_4 emissions several sources have to be considered. The most severe source (up to three quarters of the total CH_4 emissions of WWTPs) is anaerobic sludge digestion that leads especially at low temperature to a high fraction of CH_4 remaining solubilized in the digestate from where it can emit to the atmosphere. Those emissions may even exceed emissions avoided through energy recovery from biogas combustion. Therefore appropriate digestate handling is important, like e.g. the capture of ventilation air applied to sludge handling processes for subsequent use as combustion air in combined heat and power generation (Daelman et al. 2013). A critical side stream that contains high amounts of dissolved CH_4 is supernatant from digestate handling which is usually recirculated into aerobic treatment units where the CH_4 is biologically oxidized to CO_2 by methanotrophics. This can be improved by different design and/or operation measures: the aeration rate should be high enough to sustain methanotrophic growth in the reactor but low enough to prevent CH_4 stripping; more CH_4 present in the reactor is beneficial for its aerobic conversion compared to low concentrations suggesting to merge CH_4 rich streams into aerobic treatment units; preferably use stirred-tank reactors over plug flow types (Daelman et al. 2014). In addition, the influent COD consists approximately of 1% CH_4 that has been produced by microbes in anaerobic zones in the sewer network. This should be taken into account in WRF design because the unit that enters the influent into the WRF should prevent intense contact with the ambient air. For example, screw conveyors imply a more intense contact than centrifugal pumps (Lebrero et al. 2017).

The third GHG that is directly emitted from a WRF is N_2O which has been estimated to have a 265 times higher GWP than CO_2 on a 100 year time horizon (IPCC 2014). The IPCC uses fixed emission factors to estimate N_2O emissions from WWTPs (e.g. 0.035 N_2O-N per kg^{-1} influent-TKN) but N_2O emissions depend strongly on the process design. Direct N_2O emissions are in any case relevant hence should be minimized and may occur at different unit operations that have either anoxic or aerobic zones. In aerobic treatment units aeration may act as a stripping gas for N_2O (Lebrero et al. 2017). Although microbial N_2O formation is associated to happen during anaerobic denitrification more significant N_2O emissions occur during aerobic nitrification due to various species of ammonium oxidizing bacteria (AOBs). Partial nitrification that accumulates NO_2^- as an end product is probably the highest potential source that leads to N_2O formation (Lebrero et al. 2017) and therefore a process with integrated nitrification should be designed to prevent that. At low dissolved oxygen (DO) concentrations the so called nitrifier denitrification pathway may be used by particular autotrophic nitrifiers that not only oxidize NH_3 to NO_2^-, but also reduce NO_2^- to NO, N_2O and N_2 which is optimally only happening in the denitrification process. Nitrifier denitrification contrasts therefore with coupled nitrification-denitrification where many different organisms oxidize and reduce NH_3 to N_2 step wise (Wrage et al. 2001). In contrast, at high DO concentration the so called hydroxylamine pathway contributes to N_2O emissions in aerobic treatment units. During nitritation NH_3 is first converted by AOBs to hydroxylamine (NH_2OH) and then to NO_2^- and N_2O can be formed as a by-product during hydroxylamine oxidation. In addition, partial nitritation is another aerobic process often applied to treat side streams from e.g. digestate handling, and may lead to N_2O emissions under low DO concentrations (Lebrero et al. 2017).

All in all, these examples show that direct GHG emission prevention by process design is complex and requires good knowledge of microbial nitrogen and carbon conversion pathways under different operational conditions, like e.g. differing DO concentrations. It might be difficult to quantify CH_4 and N_2O emissions accurately at this early design stage but analysing potential critical point sources of a particular process design is necessary to consider emission prevention measures early. For example, choosing those reactor types that allow to establish a homogenous DO concentration in a critical process unit can already be considered at this early design stage to decrease the risk of N_2O emissions from aerobic treatment operations.

Another impact category to compare process designs is the amount of hazardous and non-hazardous waste leaving a process. However, we suggest to not include off-site waste handling into the assessment because data about applied waste management practices of WRF wastes might be difficult to obtain if this is carried out by external companies. When data is available about external waste management practices, it is possible to extend the system boundaries to assess different options like e.g. the destination of waste sludge (agricultural use, composting, landfill, incineration) which would all have different impacts (Garrido-Baserba et al. 2014).

In addition to the described direct environmental impacts of operation, emissions associated with the construction of a process should not be forgotten as the resource intensity of construction may differ substantially between alternative designs. Those emissions need to be evenly distributed over the time horizon of a planned WRF. In contrast to construction, impacts related to the end-of-life phase of a process are usually negligible compared to those from operation and construction (Foley et al. 2010). Also land degradation due to the plant construction might be accounted for since some processes are significantly more compact than others, like e.g. aerobic granular sludge treatment would be preferable over conventional activated sludge in this impact category (Pronk et al. 2015).

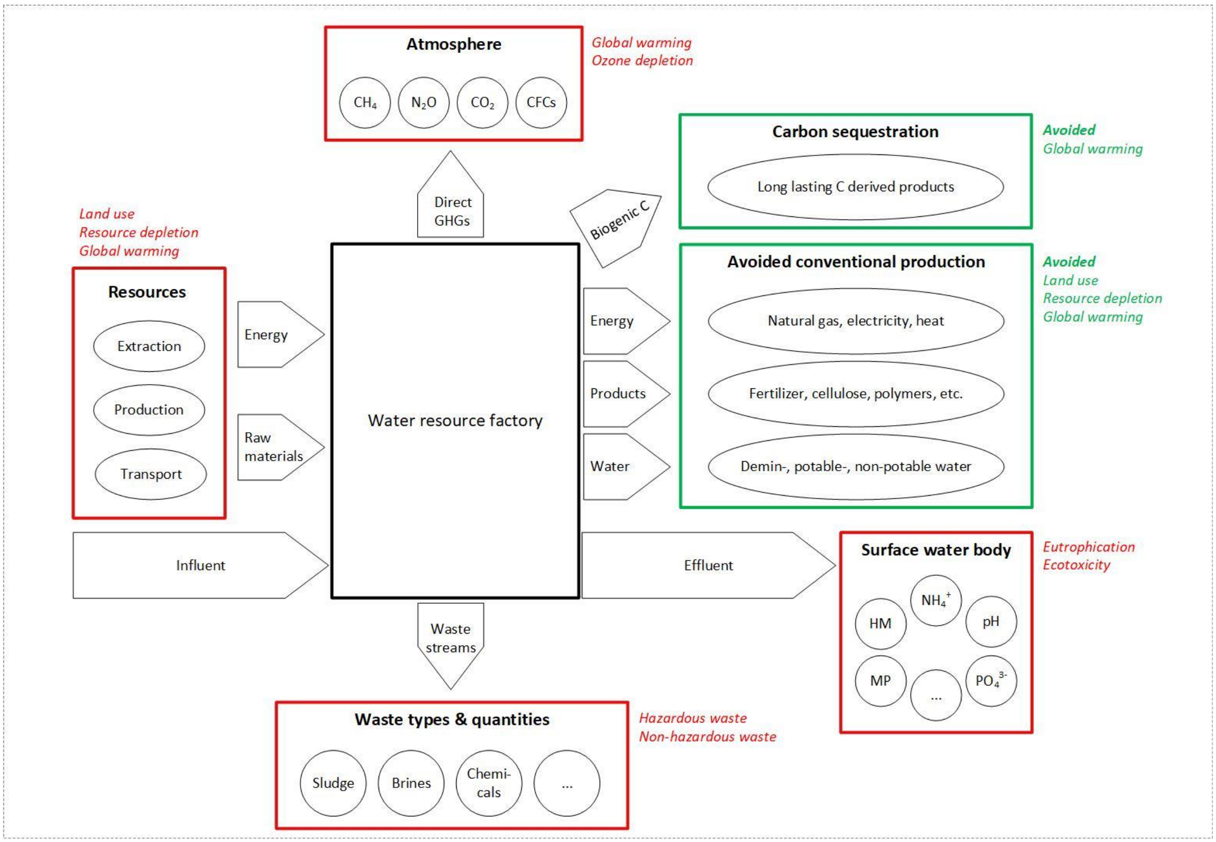

Figure 5.3 Environmental impacts that may occur due to the operation of a WRF. Negative impacts framed in red. Positive impacts are framed in green. LCA system boundaries are represented by the dashed line.

5.3.6. Step 6: Uncertainty analysis (Gate V)

Since the design of most complex processes is based on a number of multidisciplinary data and premises mostly describing future events, only hypothesises can be made about the value of these data at the time at which design decisions are taken. These data are therefore subject to a certain degree of uncertainty and may be classified as intrinsic uncertain data (e.g. prices, market sizes), or evidence based uncertain data (e.g. when different sources report different values for the same data), data obtained from models to which a certain degree of uncertainty is associated (e.g. the estimation of physical properties from thermodynamic models) (Quaglia 2013). Analysing the uncertainty of input parameters in early stage WWTP design is necessary for better informed decision making. For example, as any other predictions, estimating costs and benefits of future actions holds uncertainty and a reliable statement about the economic feasibility of a process can only be made after the uncertainty of cost and benefit estimations are analysed (Sjöstrand et al. 2018). During the LCA the highest potential to introduce uncertainties to the assessment outcome is during the life cycle inventory where inputs and outputs (e.g. energy requirements, wastes, GHG emissions) of a process are compiled and quantified (Hung and Ma 2009).

To conduct an uncertainty analysis for each assessment stage individually, the uncertain assessment criteria need to be selected and their uncertain domain needs to be defined before the domain can be sampled through Monte Carlo analysis (Bozkurt et al. 2017). Defining the uncertain domain (the probability of occurrence in reality) of possible criteria variables can be achieved by using the program evaluation and review technique (PERT) distribution method. This method assigns a probability to 3 possible values considered for each criteria. The probability of occurrence in reality is not the same for all 3 values as they are subjectively estimated by experts who define a minimum, a maximum and a moderate value for each criteria (formula 4 & 5). Experts are usually capable of giving a more confident guess about the probability of the moderate value than the extreme values hence it is given four times the probability weight of the minimum and maximum guess (Salling and Leleur 2011; Sjöstrand et al. 2018):

$$\mu = \frac{\alpha + 4 * m + \beta}{6} \tag{4}$$

$$\sigma = \frac{\beta - \alpha}{6} \tag{5}$$

Where the mean μ and the standard deviation σ of the PERT distribution are determined by the minimum value α, the most likely value m, and the maximum value β (Sjöstrand et al. 2018).

Once the uncertain criteria variables are selected and their uncertain domain is defined by PERT distribution, the domain can be sampled through Monte Carlo simulation. For example, 50 future scenarios with respect to realization of the chosen set of uncertain criteria variables applied in an assessment stage can be simulated to reveal the overall uncertainty of a process assessment dimension. In the simulation it is assumed that no correlation exists among the assessment criteria. This way Monte Carlo simulation helps decision making because it reveals which assessment criteria might need more data gathering to reduce uncertainties (Sjöstrand et al. 2018).

5.3.7. Step 7: Final process comparison and selection

Each process design that passed the assessments in the four dimensions of marketability, technical feasibility, economic feasibility, environmental impacts and shows a high probability that it meets the assessment specifications can now be subject to final comparison and selection. This final selection step should be carried out with the stakeholders on board that have been present during the objective definition in step 2 to ensure that all stakeholders agree with the outcome of the design procedure and commonly support the implementation of the finally selected process.

5.4. Conclusion

Water resource factories (WRFs) may recover water, energy, fertilizers and other products from municipal wastewater and feed into a circular economy. The SPPD-WRF framework allows to strategically plan and design a WRF step by step to finally obtain a process that is innovative, recovers marketable resources, is technically feasible, cost efficient, and shows low environmental impacts. By following the funnel and stage gating method innovative WRF processes are assessed in multiple dimensions at an early design stage. This provides decision makers and process design engineers the possibility to re-design a WRF process and improve its performance during the actual process design phase. To meet the multidimensional requirements that WRFs need to fulfil, process performance criteria traditionally applied in WWTP design need to be extended by new criteria from other research fields, like e.g. circular economy, industrial process engineering, project management, value chain development, and environmental impact assessment. After applying the SPPD-WRF framework, the process finally selected for implementation has been designed under careful consideration of the site specific necessities and circumstances in which a new WRF has to operate. The necessity to replace an existing outdated WWTP with an innovative WRF needs to be clarified early and drivers for more circular wastewater treatment practices, like e.g. projected water scarcity or the need for emission reduction of current treatment processes need to be identified. The following definition of process design objectives ideally includes all stakeholders to increase support while minimizing resistance of critical stakeholders. Process flow diagrams need to be configured in a creative and multidisciplinary engineering team effort as resource recovery oriented process configuration requires special technical expertise in fields beyond traditional wastewater treatment engineering. Each process configured then has to be analyzed by mass and energy balances which are the basis to assess its performance further. The consequential process assessment and re-designing steps do not only need to structure the process related design space but also the market related design space. Therefore, seven marketability criteria for recoverable resources are proposed in the SPPD-WRF framework to estimate the potential of an innovative WRF to recover marketable goods. This is novel in wastewater process design methodologies and aims to narrow down the number of technically possible WRF process designs to only those that are promising in terms of value chain development. Only those processes with promising marketability potential are assessed in greater detail regarding their technical performance. In addition to treatment performance criteria, this also includes criteria to assess the resource recovery potential of a process design. Quality criteria for recovered products to meet health and safety requirements but also specific customer expectations have to be considered here. WRF processes that show a technically promising performance are then further assessed by cost-benefit analysis which has to include not only investment-, variable- and periodic costs but also expectable revenues from recovered resource sales which may have an important effect on the overall process costs. In a final assessment step environmental impacts of remaining process configurations have to be quantified using life cycle assessment. Both direct process emissions and consequential emissions related to avoided conventional production of successfully recovered resources are considered in the SPPD-WRF framework. With this strategical planning, process design and assessment methodology the framework provides decision makers in the urban water cycle with a practical tool to develop a WRF from a holistic perspective without compromising the primary goal of WRFs which remains to produce good effluent qualities. Due to the multidimensionality of the framework the finally selected WRF is more sustainable than conventional treatment processes that are usually designed with a rather narrow focus on treatment performance and costs. In addition, the SPPD-WRF framework provides guidance for organizing the recovery of resources in accordance with the main conditions of the end-of-waste concept and therefore contributes to its application within the wastewater sector where it is yet underdeveloped (Hukari et al. 2016).

6

Conclusion and Outlook

“It’s difficult to make predictions, especially about the future”.

Piet Hein

6.1. Conclusion and outlook

This dissertation started in **chapter 1** with the ongoing paradigm shift on wastewater treatment from pollutant removal towards water, energy, fertilizer, and product recovery. This transition has been recognized as a necessity to contribute to a more circular economy and ultimately to sustainable urban development. The presented results confirm three general aspects to consider in the debate about the applicability and usefulness of a circular economy. Firstly, it is confirmed what has been postulated as the thermodynamic limitations of a circular economy. The second law of thermodynamics dictates that a recovery process can never reach 100% efficiency because the entropy within the system decreases (increase in order) which is only possible if an entropy increase of the environment counteracts this. Consequently, the recovery of matter and energy that are dispersed in a wastewater stream always implies losses (easac 2015). The second aspect confirmed by this dissertation is that resource recovery requires energy and resource inputs and both increase in a nonlinear manner as the percentage of recycled material rises (easac 2015). This demands the careful study of matter and energy flows on a process level to identify the most efficient processes among numerous alternatives. The third aspect confirmed is that despite of thermodynamic limitations it is very clear that in the current linear "take-make-waste" market economy, radical improvements can be achieved through the integration of innovative technologies that rearrange the physical flows towards a more cyclical model (Korhonen et al. 2018). Hence, in an ideal sense of the circular economy a water resource factory is designed to recover various resources simultaneously while consuming only little resources and energy itself and minimizing the losses occurring within the various recovery pathways it applies.

To contribute to developing such practices this dissertation aimed to answer the general question raised in **chapter 1**: how can innovative resource recovery technologies become effectively and efficiently implemented into municipal wastewater treatment processes to design water resource factories in the future? I hope that this dissertation enhances the understanding about how to combine innovative technologies most effectively and design water resource factories that fulfil several objectives simultaneously which lie outside the scope of traditional municipal wastewater treatment plant design. It is shown that successful integration of resource recovery technologies can be achieved by carefully studying the techno-economic performance of innovative processes at an early design stage through mass and energy balances and cost benefit analysis. Beyond that, it became clear that for successful process implementation a variety of non-technical aspects have to be considered as well at an early process design stage. Therefore, the main conclusion of this dissertation is that a multidisciplinary perspective and expertise is required to design water resource factories that fulfil both: (i) technical feasibility and cost effectiveness while recovering several resources without compromising on effluent quality and robust operation, and (ii) the potential to become successfully implemented within site specific market related, socio-economic and environmental conditions. This dissertation reveals various insights, methods and metrics that can help to assess and reach this two-fold goal.

The extensive literature review in **chapter 2** indicates that, although municipal wastewater may not fully satisfy the elemental or energy demands of industrialised societies, it still represents a substantial resource that should be fully utilised in the future. This is especially valid for those resources that can gain substantial market shares. Being able to supply a recovered resource at sufficient quantities is necessary to compete with conventional suppliers on the market. Therefore it is valid to analyse the demand side of recoverable resources and estimate which resource is preferably recovered over another simply because it can be supplied in relevant quantities. To come to a valid conclusion the quantities of a resource contained in a wastewater stream and the recovery yields obtainable with innovative recovery technologies must be estimated.

Among the wide variety of developed recovery technologies only a few have ever been applied on large scale due to technical immaturity and/or non-technical bottlenecks. Chapter 2 identifies nine major bottlenecks of which six relate to economic feasibility and value chain development for recovered resources. This shows that the successful implementation of water resource factories relies to a large extend on solid business case development. In addition, environmental and health issues related to recovered resources and processes need to be addressed and consumer acceptance and policy frameworks have to be established. The power of water management utilities to influence identified bottlenecks may be limited. Bottlenecks that relate to the actual process and its operation can easier be influenced than those that relate to the external circumstances in which a water resource factory has to function.

To actively take responsibility and design water resource factories, innovative process designs need to be explored. This means to study at an early design stage the performance of possible process designs regarding (i) recoverable resource quantities, (ii) trade-offs between potentially applicable recovery technologies, (iii) economic performance and (iv) energy consumption. Therefore, the techno-economic design space of water resource factories has been explored in chapters 3 and 4 by mass and energy balances and cost and benefit analysis as important methods to enhance the understanding of process integration at an early process design stage.

Chapter 3 tackles the fact that resources (i.e. COD or phosphorous) can be recovered in different forms and at different recovery rates depending on the applied technology. Therefore it is necessary to understand the trade-offs of integrating a particular recovery technology instead of another one and how to possibly combine various technologies to maximize the overall recovery rate of a process. The recovery of COD as either energy or as biopolymers represents such a clear trade-off within the aerobic granular sludge treatment process. It is shown that a process can be designed in a way that both products are recovered simultaneously with an even higher total COD recovery than it would be achieved if only one of the two products is recovered. Therefore, the decision about which recovery technologies to integrate into the aerobic granular sludge process needs a clear goal definition. If the goal is to design a water resource factory that generates revenues for the operating utility it is wise to prioritize biopolymer recovery because it is a higher value product than energy. But if the utility aims for a smaller carbon footprint it might be useful to prioritize a process for maximum energy recovery which considerably lowers the process's energy consumption but leaves also lower COD left for biopolymer recovery. Moreover, it has been revealed that phosphorous recovery as struvite is not an advisable option if sludge incineration is in place that allows to recover phosphorous from the ash fraction because the latter option reaches a significantly higher overall phosphorous recovery rate. Nevertheless, although struvite contains only a small fraction of the phosphorous contained in the influent it can still be recovered in quantities that may be competitive to bulk fertiliser suppliers because it contains also ammonia and magnesium. To sum it up, chapter 3 shows that the integration of resource recovery technologies into a aerobic granular sludge treatment plant will likely alter its technical, economic and environmental performance and therefore it is important to decide early which resources that are potentially recoverable from the wastewater stream should be preferred over others and why. Mass and energy balances provide a useful basis to pre-select promising designs for further in depth and more costly techno-economic and environmental impact assessments.

Chapter 4 analyses the techno-economic performance of membrane-based advanced treatment processes that can be applied to reclaim water from wastewater treatment plant effluents. Water is the most precious and abundant resource contained in municipal wastewater. Since water can be reclaimed in different qualities for different reuse types (i.e. industrial, domestic, agricultural reuse), it is necessary for a better informed decision making to understand the generic techno-economic differences of processes that target these different reuse types. The economic feasibility of advanced WWTP effluent reclamation processes depends heavily on the price to which reclaimed water can be sold. Prices for industrial process water, potable water and irrigation water can vary considerably within a region or country with industrial process water (demineralised) being the most expensive in The Netherlands followed by potable water. Therefore, to design a process that allows to supply reclaimed effluents to industries is economically most promising.

When the goal is to reclaim water for irrigation, the reclamation process may not require an energy and cost intensive reverse osmosis unit but only an ultrafiltration step followed by ultraviolet light disinfection. The decision depends on the targeted crop, its intended use and the irrigation method applied as these factors determine the required legal irrigation water quality. However, both process options can have the same net present value if the difference in water quality is also reflected in the water price. The higher water quality of reverse osmosis permeate enables the supply of reclaimed water to various other reuse types than irrigation and therefore increases the flexibility of a reclamation process to switch between different demands if necessary. Generally speaking irrigation water can be considered as too cheap to be reclaimed from municipal waste water in a cost effective manner under Dutch market conditions.

Membrane based wastewater reclamation processes have often been criticised as too energy intensive to compete with conventional drinking water treatment systems. Therefore it is helpful to study energy recovery possibilities from wastewater and renewable energy technology integration to reveal to what extend this could provide the energy for a reclamation process. Chapter 4 reveals that the recoverable electricity from influent COD via maximized anaerobic digestion and biogas combustions is negligible compared to the electricity required to operate membrane-based advanced treatment processes. In contrast to this, photovoltaic seems a feasible renewable energy source as required solar panel areas have been estimated to be manageable. Another process optimisation concept is the increase of the reverse osmosis recovery rate by integrating a softener/biostabilizer unit prior to it. This could improve the sustainability of wastewater reclamation processes significantly as energy consumption and costs significantly decrease per m^3 water reclaimed.

Reclaimed wastewater demand may underlie seasonal or even daily variations especially if irrigation water is reclaimed. The design of a fit for multi-purpose membrane based effluent reclamation process seems feasible as only a few additional unit operations may be required compared to a process that targets only one reuse type. Therefore, this concept should be investigated further with more in depths analyses of the techno-economic performance. It could provide a more flexible way to react on varying demands and even supply different water qualities simultaneously if needed.

Chapter 5 combines the insights gained in chapters 2-4 and tackles the fact that conventional wastewater treatment plants are usually designed with a rather narrow focus on robust treatment performance to meet legal effluent quality standards on the one hand and process costs on the other hand. The design of water resource factories requires a more multi-dimensional approach. Resource recovery technology integration alters the overall technical, economic and environmental performance of treatment plants. Therefore, the design space of water resource factories has to be opened up and newly structured to include additional useful process assessment methods. Traditionally applied process performance criteria need to be extended by new criteria from other research fields, like e.g. circular economy, industrial process engineering, project management, value chain development, and environmental impact assessment. In addition, the early assessment of the marketability and value chain development potential of recoverable resources is an especially important step yet often overlooked in scientific literature. The demand, logistics, consumer acceptance, legal situation, market supply potential and application possibilities of recoverable resources have to be assessed to ensure their suitability as commodities. The underlying rationale to assess the marketability of recovered resources is that the concept of a circular economy emerged from the observation of natural matter cycles that do not know wastes but only resources and aims to transfer those natural principles into societal production-consumption patterns. But the current market economy aims for perpetual short term economic value increment of individuals, organisations and states. Consequently, combining circular resource solutions with the current economic model requires that economic value is created from the cycle in a predictable timeframe.

Next to treatment and operational performance indicators, technical process assessments of innovative water resource factories need to include also criteria that assess the recovery potential of a process including product quality and mass efficiency. Economic process assessments need to account for the diverse revenues a water resource factory can have and not only consider process costs alone. Moreover, when the environmental impact of a water resource factory is assessed, the system boundaries of the life cycle assessment differs substantially from conventional treatment plant assessments. Since a water resource factory is also a production system the recovered and successfully marketed resources avoid conventional production of similar goods and their impacts and this has to be considered.

To sum it up, the vast variety of possible water resource factory process designs requires to ask about the necessities and site specific circumstances in which a process needs to function. When the goals of an innovative process are clearly defined, the technical design phase can start. Effective process integration of resource recovery technologies that fulfil pre-defined goals can only be achieved if the matter and energy conversions of innovative water resource factories are understood. Successful resource recovery means to design processes that supply existing demands with competitive products. The technologies to achieve that are becoming increasingly ready but careful planning and assessment of technical but also non-technical factors is needed to make the most feasible design choices and finally design a process that is innovative, recovers marketable resources, is technically feasible, cost efficient, and shows low environmental impacts.

Water resource factories can be a key infrastructure that create cyclic resource flows within a city. They may connect different sectors that have traditionally not worked together, like e.g. the drinking water suppliers with wastewater management utilities when wastewater is reclaimed. If wastewater resource recovery should become the new norm that replaces wastewater treatment in the future, the challenge for water management utilities will be to systematically innovate and create networks that cross disciplinary boundaries. To create value chains from wastewater derived resources requires to engage stakeholders with economic, industrial, policy and research interests as well as different public authorities and the civil society. Water management utilities need to perceive themselves as market participants and reliable suppliers of eventually scarce resources and commodities. They need to engage politically as organisations that significantly contribute to sustainable development and serve the public in more ways than merely the protection of surface water bodies.

Although the main bottlenecks that hinder the implementation of water resource factory relate often to economic unfeasibility, the decision of resource recovery technology implementation should not be decided solely based on economic considerations. This is especially valid if a recoverable resource is projected to become scarce in the future. The alleviation of future resource scarcity and the need for more sustainable urban spaces should be weighted higher than short term economic value creation. Therefore, water resource factories should be designed to primarily recover those resources that show a high potential to diminish conventional resource exploitation of scarce resources. This can be an important guidance to recover resources that are considered scarce but do not reflect the scarcity in their market price yet. In such a case it needs to be a political goal to fund the recovery of a resource that cannot be recovered cost-effectively yet although its availability in the future is threatened. This can be especially valid for water reclamation from wastewater treatment plant effluents in water scarce regions.

A limitation of this dissertation is that it assumes a centralised wastewater treatment system that consists of a sewer network and a wastewater treatment plant. Although such a system is in place in many countries, it may not be feasible for others. The assessments and methods used may not be useful to study decentralised wastewater management solutions because the scale in which resources are recoverable determines the type of value chain that can be developed.

The dissertation clearly shows the importance of a multidisciplinary approach of future research in the field of wastewater resource recovery. The ongoing technological development needs to be accompanied by expertise in marketization, policy formulation, stakeholder engagement, circular economy. In addition, disciplines that understand the possible applicability of recovered resources, like e.g. material science or agricultural science should be included. The successful implementation of water resource factories will depend on the creativity and cooperation of a broad range of fields and needs to include the public and private sector.

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Appendices

Appendix I

In this section the supplementary information of chapter 3 are presented.

Parameters applied in mass and energy balances

Influent			
Average flow rate	64400	m^3/d	Personal communication M. Pronk
COD	729,8	mg/l	" "
BOD	287,3	mg/l	" "
TSS	314,4	mg/l	" "
TKN	60,6	mg/l	" "
NH_4-N	70	%/TKN	(Hartley 2013)
Organic-N	30	%/TKN	(Hartley 2013)
P-total	8,4	mg/l	Personal communication
Ortho-P	67	%/P-total	(Henze and Comeau 2008)
Organic-P	33	%/P-total	(Henze and Comeau 2008)
Fe total	1,0	mg/l	(Wilfert et al. 2015)
Fe^{3+} share	100	%	(Wilfert et al. 2016)
Fe^{3+} forming vivianite	100	%	(Wilfert et al. 2016)
Energy COD	17,8	kJ/g COD	(Heidrich et al. 2011)
Coarse screen			
No impacts on any relevant flows are assumed			
CEPT			
Coagulant	Polymer		(Klute and Hahn 1994)
COD removal	60	%	(Wan et al. 2016)
TKN removal in conventional primary settler (with 40% COD removal, no coagulant)	16,4	%	based on (Hartley 2013)
P-total removal in conventional primary settler (with 40% COD removal, no coagulant)	11	%	(Cornel and Schaum 2009)
Effect of coagulation compared to conventional primary settler with 40% COD removal:			
TKN effluent concentration	-9,4	%	based on (Klute and Hahn 1994)
P-total effluent concentration	-33,3	%	based on (Klute and Hahn 1994)
AGS			
COD oxidized	60	%	(Winkler 2012)
COD into sludge	40	%	(Winkler 2012)
COD removal	85	%	based on (Pronk et al. 2015)
TKN removal	94	%	(de Kreuk et al. 2005)
TKN into sludge	20	%	(Matassa et al. 2015)
Organic-N converted to NH_4	90	%	(Makinia et al. 2011)
P-total removal	87	%	based on (Pronk et al. 2015)
Ortho-P removal	91	%	based on (Pronk et al. 2015)

Organic-P converted to PO4	90	%	(Krishnaswamy et al. 2011)
Fe-P removed with sludge	100	%	own assumption
EPS extraction			
EPS content in AGS	20	%	(van der Roest et al. 2015)
Yield of downstream process	100	%	own assumption
C content EPS	47,05	wt%	(Felz et al. 2019)
N content EPS	7,61	wt%	(Felz et al. 2019)
P content EPS	2,92	wt%	(Felz et al. 2019)
Anaerobic digester			
COD converted to biogas	50	%	(Khiewwijit et al. 2016)
CH_4 content biogas	65	%	(Frijns et al. 2013)
Producible amount of CH_4	0,35	Nm^3 CH_4/kg COD	(Frijns et al. 2013)
Calorific value CH_4	35,9	MJ/Nm3	(Frijns et al. 2013)
Organic-N converted to NH_4	100	%	(Mehta et al. 2015)
Organic-P converted to PO_4	100	%	(Mehta et al. 2015)
Combined heat and power			
Electricity conversion efficiency	38	%	(Verstraete and Vlaeminck 2011)
Heat recovery potential	40	%	(Verstraete and Vlaeminck 2011)
Decanter centrifuge			
COD ending in liquid fraction	10	%	(Andreoli et al. 2007)
NH_4-N ending in liquid fraction	10	%	(Khiewwijit et al. 2016)
PO_4-P ending in liquid fraction	10	%	(Khiewwijit et al. 2016)
Incinerator			
Energy content COD	0,0178	kJ/mgCOD	(Heidrich et al. 2011)
Electrical efficiency	40	%	(Faaij 2006)
Dry solid content sludge	22	%	(Frijns et al. 2013)
P recovery rate	80	%	(Lundin et al. 2004)
Struvite crystallization			
PO_4 crystallization rate	80	%	(Martí et al. 2010)
Required molecular ratio of (Mg^{2+}:$NH4^+$:PO_4^{3-})	1:1:1		(Verstraete et al. 2009)
O_2 consumption AGS reactor			
COD removal rate	87	%	(Pronk et al. 2015)
COD into CO_2	60	%	(Winkler 2012)
NH_4 per TKN in	70	%	(Hartley 2013)
Organic-N per TKN in	30	%	(Hartley 2013)
Organic-N hydrolyzed to NH_4	90	%	(Makinia et al. 2011)
NH_4 removal rate	94	%	(de Kreuk et al. 2005)
NH_4 uptake in AGS	20	%	(Matassa et al. 2015)

Biodegradable-COD:N ratio AGS reactor

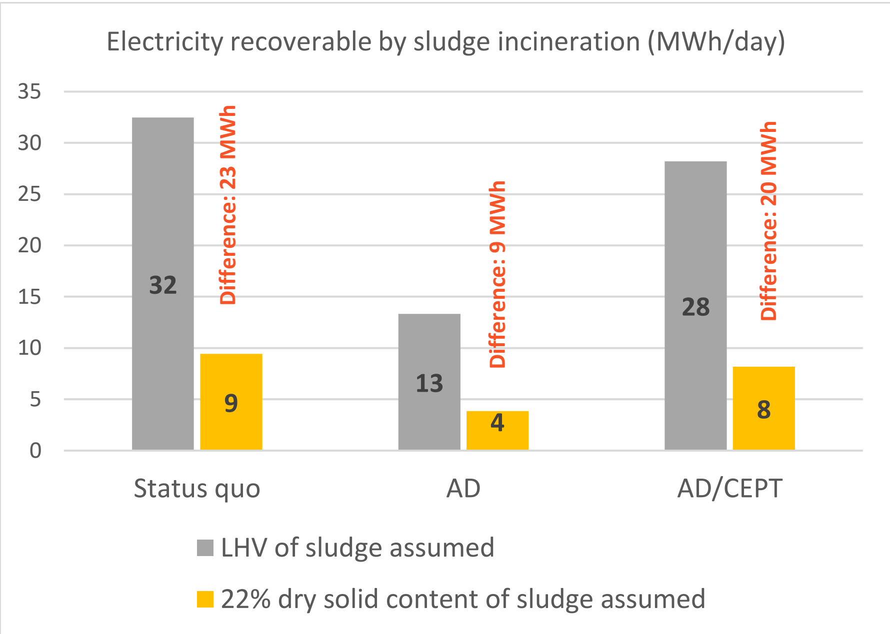
To ensure a sufficient denitrification in the AGS reactor it is important to maintain a high enough bCOD:N ratio. During CEPT ca. 60% bCOD (Wan et al. 2016) but only ca. 25% TKN (Hartley 2013) can be assumed to be removed before AGS treatment. Therefore, it needs to be checked if sufficient bCOD is still present in the AGS reactor. Assuming that the total influent-COD consists to 75% of biodegradable COD (bCOD) (Pasztor et al. 2009; Hartley 2013), a bCOD:N ratio of 4,9 can be expected in those designs with CEPT integration.

Effluent qualities

All designs would meet the Dutch legal effluent requirements.

Pollutant	Legal	Design					
mg/l	(Pronk et al. 2015)	Status quo	AD	AD/CEPT	EPS	AD+EPS	AD/CEPT+EPS
COD	125	92,3	95,7	44,0	92,3	94,8	43,6
P-Total	1	0,7	0,8	0,5	0,7	0,8	0,5
TKN	7	5,3	5,9	4,6	5,9	5,9	4,6

Sensitivity of waste sludge dry solid content



Explanation of figure "Electricity recoverable by sludge incineration": In the mass and energy balances it is assumed that waste sludge arrives at the incinerator with a dry solids (ds) content of 22% which represents the Dutch average. This implies that over 70% of the sludge COD energy content is needed to evaporate the water until a sludge ds content is reached that is energy positive (Frijns et al. 2013). Since the assumed water content of incinerated sludge is a sensitive value for its energy recovery potential, the possible maximum recovery potential has been calculated using the lower heating value (LHV) of the sludge. The assumed 22% ds implies that ca. 30% of the theoretical energy would be recovered.

Appendix II

In this section the supplementary information of chapter 4 are presented.

Process design parameters, energy consumptions and costs

Buffer tank

Design and cost parameters buffer tank [1]	
Buffer capacity (h)	10
Size (m3)	1000
Material	steel
Price (Euro)	85000

Reference:

1. Personal communication with Global Water Engineering B.V.

Pre-filter

Design and cost parameters pre-filter [1]	
Type	Disk filter
Energy	
Pressure (bar)	0,05
Pump efficiency	0,6
Flow rate (m3/h)	100
Energy consumption (kwh)	0,2
OPEX	
Energy cost (Euro/d)	0,55
Energy cost (Cent/m3)	0,02
Lifetime filter (year)	10
Cost filter (Euro)	5000
Replacement events	1
Replacement costs (Cent/m3)	0,028
Total OPEX (Cent/m3)	0,051
CAPEX	
Full unit cost (Euro)	10000

Reference:

1. Personal communication with Global Water Engineering B.V.

Ultrafiltration

Type	Disk filter
General Info [1]	
Operation mode	constant module flux
Number of train	1
Number of module per train	1
Total number of elements	28
Element type	UF SFP-2880
UF recovery (%)	94,2
Design flux (LMH)	50
General Operation info [1]	
Feed flow rate (m3/h)	99,5
Permeate flow rate (m3/h)	94,2
Applied pressure (bar)	2,3
Average net driving pressure	0,52
UF average flux (LMH)	50
Cleaning protocol	
Forward flush	with UF feed water
Backward flush interval (h)	1
Backward flush with UF permeate (min)	3,8
CEB water with UF permeate (min)	16,1
CEB water interval (h)	12
CIP cleaning interval (d)	30
CIP water with UF permeate (min)	312,8
Flow properties (UF Feed) [1]	
Turbidity (NYU)	4
TOC (mg/L)	9,8
SDI 15	5
TSS (mg/L)	6
Flow properties (UF permeate) [1]	
Turbidity (NYU)	<0,1
TOC (mg/L)	8,8
SDI 15	<2,5
OPEX [1]	
Utility and chemicals (Euro/m3)	0,06
Membrane replacement (Euro/m3)	0,02
CAPEX per train [2]	
Single element cost (Euro/module)	2000
Total module cost[3] (Euro)	14000
Feed pump[3] (Euro)	5000
Feed buffer tank [3] (Euro)	9000
Filtrate tank [3](Euro)	8100
Concentrate tank [3] (Euro)	900
Automation control, electrics (Euro)	100000
CEB chemical dosing pump (Euro)	500
Compressor (Euro)	10000
CIP (pumps, tanks etc.) (Euro)	20000
Miscellaneous [3] (Valves, piping) (Euro)	8000

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)

2. (Hitsov et al. 2018)

3. Full redundancy is considered.

Cartridge filter

General info [1]	
CF operation model	Inside-out
Type	String wound filtration
CF housing material	PVC
Flow rate (m3/h)	100
Micron rating	5 μm
Length (in)	40
Operational pressure (bar)	1
Pressure drop (bar)	0,2
Net applied pressure (bar)	1,2
Number of filters	22
Lifetime (month)	1
Operation time (year)	20
Unit cost (Euro/piece)	2
Energy consumption	
Unit cost of electricity (Euro/kWh)	0,1
Pump efficiency	0,6
Energy consumption (kWh)	5,56
OPEX	
Energy (Euro/year)	4393
Replacement (Euro/year)	10560
OPEX (Euro/year)	14.953
CAPEX	
Pump (Euro)	5000
Cartridge filters (Euro)	440
CF housing (Euro)	1000

Reference:

1. (Farhat et al. 2020)

2. Personal communication Evides Industriewater B.V.

Reverse osmosis

General Info [1]	
Configuration	Double stages
Stage 1 (No. PV)	10
Stage 2 (No. PV)	5
Number of elements per PV	6
Total number of elements	90
Elements type	ECO-PRO 400
RO recovery (%)	80
General Operation [1]	
Feed flow rate (m3/h)	94,2
Permeate flow rate (m3/h)	75,3
Feed pressure (bar)	9,6
Average net driving pressure	6,7
Specific Energy (kWh/m3)	0,42
RO average flux (LMH)	22,5
Antiscalant (mg/l)	3,5
Flow properties [1]	
RO feed flow rate (m3/h)	94,2
Pressure (bar)	9,6
TDS (mg/L)	1314
RO concentrate	18,9
Pressure (bar)	8
TDS (mg/L)	6500
RO permeate (m3/h)	75
Pressure (bar)	1
TDS (mg/L)	12,13
CIP cleaning [2]	
CIP frequency (events/yr)	40
CIP duration (h)	8
CIP chemicals	HCl/NaOH
CIP cleaning pH	2/12
OPEX [1]	
Energy, waste disposal and antiscalant (Euro/day)	423,6
Cost of fouling (Euro/m3)	0,03
Cartridge filter (Euro/m3)	0,02
CAPEX [3]	
Single element (Euro/module)	600
90 Elements (Euro)	54000
Pressure vessel (Euro/module)	600
15 Pressure vessels (Euro)	9000
Feed Pumps (Euro)	200000
Feed buffer tank (Euro)	9000
Filtrate tank (Euro)	7200
Concentrate tank (Euro)	1800
Automation control, electrics (Euro)	100000
CIP (pumps, tanks etc.) (Euro)	30000
Valves, piping etc. (Euro)	15000

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)

2. (Shang et al. 2011)

3. (Hitsov et al. 2018)

Fouling and cleaning in reverse osmosis

Feed Channel pressure drop increase [1]	
Initial Normalized feed pressure drop (NPD0) (kPa)	120
Average Normalized feed pressure drop (NPD) (kPa)	140
Unit cost of electricity (Euro/kWh)	0,1
Pump efficiency	0,6
Feed flow rate (m3/d)	2248,8
Annual energy consumption due to pressure drop (Euro)	7600
Annual cost of pressure drop (Euro)	760
Permeability reduction [1]	
Virgin membrane permeability (m/s/kPa)	3,6
Average fouled membrane permeability (m/s/kPa)	2,88
Permeate flow rate (m3/h)	1800
Design flux(constant value) (LMH)	20
Annual energy consumption due to permeability (Euro)	43452
Annual cost of permeability reduction (Euro)	4345
Membrane replacement [1]	
RO module (Euro/stuck)	600
Total number of membrane replaced	90
Life time (year)	5
CIP cleaning [1]	
Chemicals:	
C HCl (pH=2)	0,36
C NaOH (pH=12)	0,4
Cost of acid (Euro/kg)	100
Cost of base (Euro/kg)	100
Acid volume (m3)	2,88
Base volume (m3)	2,88
Number of CIP	40
Annual cost of chemicals (Euro)	8801
Waste disposal	
Waste cost factor (Euro/m3)	0,6
Annual cost of down-time	2400
Down-time cost (Due to CIP)	
Water production capacity (m3/d)	1800
Profit loss due to down-time (Euro/m3)	0,1
CIP solution heating	
pump efficiency	0,9
Ambient Temperature (C)	10
T base (C)	35
T acid (C)	35
Specific heat capacity kJ/kg/C	4,2
Density (kg/m3)	1000
Total energy consumption (kWh)	7467
Annual heating cost (Euro)	747

Reference:

1. (Jafari et al. 2021)

Degasifier

Designed for saturated stream	
CO_2 removal (%)	70
Type	Tank/Agitator
Design flow (m^3/h)	75
Residence time (min)	10
Agitation power intensity (W/m^3)	50
Mixing tank volume (m^3)	12,5
Specific Energy (kWh/m^3)	0,01
OPEX	
Energy cost ($Euro/m^3$)	0,01
CAPEX [1]	
Tank (Euro)	7000
Agitator (Euro)	2500
Instrumentations (Euro)	2375

Reference:

Personal communication (Global Water and Energy)

Ion exchange (mixed-bed) resin column

Ion Exchange mixed bed- Permeate polishing [1] [SAC|SBA]		
Resin form	SAC	SBA
Layout	[SAC|SBA]	
Vessel type	Amberpack sandwich	
Resin type	AMBERLITE™ HPR1200 H	AMBERLITE™ HPR4200 Cl
Feed flow rate (m3/h)	75	75
Regeneration bypass (m3/h)	0,6	
Design flow rate (m3/h)	75	75
Permeate flow rate (m3/h)	74,4	74,4
Design run time (h)	48	48
Regeneration time (h)	4,47	
Bed volume (m3)	2,23	4,47
Feed water pH	9,8	
Product pH	7,9	
Vessel pressure drop (bar)	1,5	
Safety factor	0,95	0,95
Resin info		
Volume (m3)	2,23	4,47
Ionic load (eq)	2282	2527
Regenerate	HCl	NaOH
Regenerate dose (g/L)	80	80
Regeneration ratio (g/L)	160	370
Vessel type		
Linear Velocity (m/h)	38	
Vessel diameter (inside) mm	1580	
Vessel diameter (outside) mm	1600	
Inert resin		
Resin	AMBERLITE™ 14i	
OPEX[1]		
Utility-service-Chemical cost (Euro/d)	125,17	
Specific Energy cost (Euro/m3)	0,01	
Resin replacement (Euro/h)	1,03	
Resin replacement (Euro/m3)	0,01	
Total OPEX (per m3)	0,02	
CAPEX[2]		
SAC (Euro/m3)	2000	
SBA (Euro/m3)	5000	
IEX vessel (plug and play) (Euro)	120000	
Pumps (Euro)	3000	
Buffer tank (Euro)	4000	
Total net CAPEX (Euro)	319495	

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)

2. Personal communication (DuPont Water Solutions)

Remineralisation

Remineralisation [1]	
General info (lime milk)	$2CO_2+Ca(OH)_2$<------->Ca^{2+} + $2HCO^{3-}$
$Ca(OH)_2$ concentration(gr/m3)	144,1
Inlet flow rate (m3/h)	75
Outlet flow rate (m3/h)	75
Energy consumption (kWh)	0,17
OPEX [1]	
Lime consumption (kg/day)	259
Lime consumption (tone/year)	95
Cost of Lime (Euro/year)	14199
RO Permeate (m3/year)	657000
Cost of lime (Euro/m3)	0,02
Electricity cost (Euro/m3)	0,008
Total OPEX	0,03
CAPEX [1]	
Lime milk prep tank (Euro)	1000
Pumps (Euro)	1000
Saturators (Euro)	10000
Total CAPEX (Euro)	12000

Reference:

1. (El Azhar et al. 2012)

UV disinfection

UV disinfection[1]		
	Potable (RO permeate)	Irrigation (UF permeate)
Assumed UV dosage (mJ/cm2)	80	80
Assumed log removal	4	4
TDS (mg/l)	41,2	1333,3
CAPEX calculation		
Price of unit	12000	15000
Lifetime of unit (yr)	10	10
Total CAPEX	24000	30000
Cost of lamp energy consumption		
Flow (m3/h)	75,4	94,2
Required lamps	16	24
Lifetime of lamps (h)	12000	12000
Energy consumed per lamp (kW)	0,1	0,1
Unit cost of electricity (Euro/kWh)	0,1	0,1
Cost of pumping energy consumption		
Average feed pressure (bar)	0,05	0,05
Pump efficiency (eta)	0,6	0,6
Flow rate (m3/h)	75,4	94,2
Other energy requirements		
Cleaning automation etc. (kWh)	4	6
Cost of other energy requirements (Euro)	0,005	0,006
Cost of lamp replacements		
Price of one lamp (Euro)	100	100
Required lamps	16	24
Total price lamps (Euro)	1600	2400
Lifetime of lamps (h)	12000	12000
Lifetime of lamps (yr)	1,4	1,4
Remaining lifetime (yr)	18,6	18,6
Required lamp replacement events	13,6	13,6

Reference:

1. Personal communication (Xylem Water Solutions Nederland B.V & WeUVcare)

Labour costs

It is assumed that each MATP can be operated by 2 operators that earn a gross salary of 50000 Euros per year each. Labour requirements and costs are estimated based on personal communication with sector specific companies.

Process optimization for increased RO recovery

Softener/biostablizer

Softener/biostablizer [SAC-SBA][1,2]		
Resin form	SAC	SBA
Resin Type	Amberlite-HPR1100 Na	Amberlite-HPR4580 Cl
Layout	[SAC-SBA]	
Vessel type	Amberpack sandwich	
Resin type	Amberlite-HPR1100 Na	Amberlite-HPR4580 Cl
Feed flow rate (m3/h)	93,7	93,7
Regeneration bypass (m3/h)	2,4	2,4
Design flow rate)m3/h)	100	94
Permeate flow rate (m3/h)	93,7	93,7
Design run time (h)	10	10
Regeneration time (h)	1,71	3,46
bed volume (m3)	8,3	5,25
Feed water pH	12	12
Product pH	7	7
Vessel pressure drop (bar)	1	
Safety factor	0,49	0,79
Resin info		
Volume (m3)	8,3	5,25
Effective ionic capacity (eq/L)	0,81	0,54
Ionic load (eq)	6686	7221
Regenerate	RO Brine+ 2,2 g/L NaCl	RO Brine+ 2,2 g/L NaCl
Regeneration dose (g/L)	80	80
Regeneration ratio (g/L)	485	485
Vessel type		
Linear Velocity (m/h)	23	
Internal bed area(m2)	4,426	
Vessel diameter (inside) mm	2374	
Vessel diameter (outside) mm	2400	
Inert resin		
Resin	Amberlite 14i	
Volume (m3)	0,747	
Height (mm)	169	
OPEX[1]		
Utility-service-Chemical cost (Euro/d)	24,55	
Specific Energy cost (Euro/m3)	0,01	
Resin replacement (Euro/h)	0,76	
Resin replacement (Euro/m3)	0,01	
Total OPEX (per m3)	0,02	
CAPEX[2]		
SAC (Euro/m3)	2000	
SBA (Euro/m3)	5500	
IEX vessel (plug and play)	200000	
Pumps (Euro)	3000	
Buffer tank (Euro)	4000	
Total net CAPEX (Euro)	447200	

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)

2. Personal communication (DuPont Water Solutions)

Reverse osmosis

General Info [1]	
Configuration	Double stages
Stage 1 (No. PV)	12
Stage 2 (No. PV)	6
Number of elements per PV	6
Total number of elements	108
Elements type [2]	ECO Pro-400
RO recovery (%)	95
General Operation [1]	
Feed flow rate (m3/h)	93,8
Permeate flow rate (m3/h)	89,06
Feed pressure (bar)	10,3
Average net driving pressure	6,7
Specific Energy (kWh/m3)	0,38
RO average flux (LMH)	22,2
Antiscalant (mg/l)	0
Flow properties [1]	
RO feed flow rate (m3/h)	93,8
Pressure (bar)	10,3
TDS (mg/L)	933,8
RO concentrate (m3/h)	4,64
Pressure (bar)	9,4
TDS (mg/L)	15224
RO permeate (m3/h)	89,06
Pressure (bar)	1
TDS (mg/L)	129,2
CIP cleaning [2]	
CIP frequency (events/yr)	20
CIP duration (h)	8
CIP chemicals	HCl/NaOH
CIP cleaning pH	2/12
OPEX [1]	
Energy, waste disposal and antiscalant (Euro/m3)	0,07
Cost of fouling (Euro/m3)	0,24
Cartridge filter (Euro/m3)	0,02
CAPEX [3]	
Single element (Euro/module)	600
108 Elements (Euro)	64800
Pressure vessel (Euro/module)	600
18 Pressure vessels (Euro)	10800
Feed Pumps (Euro)	200000
Feed buffer tank (Euro)	9000
Filtrate tank (Euro)	7200
Concentrate tank (Euro)	1800
Automation control, electrics (Euro)	100000
CIP (pumps, tanks etc.) (Euro)	30000
Valves, piping etc. (Euro)	15000

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)

2. (Shang et al. 2011)

3. (Hitsov et al. 2018)

Fouling and cleaning in reverse osmosis

Feed Channel pressure drop increase [1]	
Initial Normalized feed pressure drop (NPD0) (kPa)	120
Average Normalized feed pressure drop (NPD) (kPa)	140
Unit cost of electricity (Euro/kWh)	0,1
Pump efficiency	0,6
Feed flow rate (m3/d)	2248,8
Permeability reduction [1]	
Virgin membrane permeability (m/s/kPa)	3,6
Average fouled membrane permeability (m/s/kPa)	2,88
Permeate flow rate (m3/d)	2138
Design flux(constant value) (LMH)	20,2
Membrane replacement [1]	
RO module (Euro/stuck)	600
Total number of membrane replaced	108
Life time (year)	5
Annual cost of module replacement (Euro)	10800
CIP cleaning [1]	
Chemicals:	
C HCl (pH=2)	0,36
C NaOH (pH=12)	0,4
Cost of acid (Euro/kg)	100
Cost of base (Euro/kg)	100
Acid volume (m3)	2,88
Base volume (m3)	2,88
Number of CIP per year	20
Waste disposal	
Waste cost factor (Euro/m3)	0,6
Down-time cost (Due to CIP)	
Water production capacity (m3/d)	1800
Profit loss due to down-time (Euro/m3)	0,1
CIP solution heating	
Pump efficiency	0,9
Ambient Temperature (C°)	10
T base (C°)	35
T acid (C°)	35
Specific heat capacity kJ/kg/C°	4,2
Density (kg/m3)	1000

Reference:

1. (Jafari et al. 2021)

Degasifier

Designed for saturated stream [1]	
CO2 removal (%)	70
Type	Tank/Agitator
Design flow (m3/h)	89,6
Residence time (min)	10
Agitation power intensity (W/m3)	50
Mixing tank volume (m3)	14,9
Specific Energy (kWh/m3)	0,01
OPEX	
Energy cost (Euro/m3)	0,01
CAPEX [1]	
Tank (Euro)	7000
Agitator (Euro)	2500
Instrumentations (Euro)	2375

Reference:

1. Personal communication (Global Water and Energy)

Ion exchange (mixed-bed) resin column

Ion exchange mixed-bed permeate polishing [1] [SAC|SBA]		
Resin form	SAC	SBA
Layout	[SAC|SBA]	
Vessel type	Amberpack sandwich	
Resin type	AMBERLITE™ HPR1200 H	AMBERLITE™ HPR4200 Cl
Feed flow rate (m3/h)	89,6	89,6
Regeneration bypass (m3/h)	1,7	
Design flow rate)m3/h)	90	90
Permeate flow rate (m3/h)	87,9	87,9
Design run time (h)	20	20
Regeneration time (h)	3,76	
Bed volume (m3)	2,52	5,77
Feed water pH	9,8	
Product pH	7,9	
Vessel pressure drop (bar)	1,34	
Safety factor	0,95	0,95
Resin info		
Volume (m3)	2,52	5,77
Ionic load (eq)	3113	3114
Regenerate	HCl	NaOH
Regenerate dose (g/L)	80	80
Regeneration ratio (g/L)	165	370
Vessel type		
Linear velocity (m/h)	36	
Vessel diameter (inside) mm	1780	
Vessel diameter (outside) mm	1800	
Inert resin		
Resin	AMBERLITE™ 14i	
OPEX[1]		
Utility-service-chemical cost (Euro/d)	389	
Specific utility and chemical cost (Euro/m3)	0,18	
Resin replacement (Euro/h)	1,03	
Resin replacement (Euro/m3)	0,01	
Total OPEX (per m3)	0,2	
CAPEX[2]		
SAC (Euro/m3)	2000	
SBA (Euro/m3)	5000	
IEX vessel (plug and play) (Euro)	120000	
Pumps (Euro)	3000	
Buffer tank (Euro)	4000	
Total net CAPEX (Euro)	333665	

Reference:

1. Water Application Value Engine (WAVE) software (DuPont Water Solutions)
2. Personal communication (DuPont Water Solutions)

Renewable energy integration

Electricity recoverable with anaerobic sludge digestion

Unit	Assumptions & Calculations	Value	Unit	Reference
Influent	Total influent	100	m3/h	
	COD []	750	mg/l	(Henze and Comeau 2008)
		750000	mg/m3	
		0,75	kg/m3	
	COD load	75	kg	
	Energy content COD	17,8	kJ/gCOD	(Heidrich et al. 2011)
	Chemical energy in influent COD	1335000	kJ	
		1335	MJ	
Primary COD capture	COD capture	0,6	%	(Wan et al. 2016)
	COD into primary sludge	45	kg	
CAS	COD load	30	kg	
	COD oxidised into CO_2	0,6	%	(Winkler et al. 2013)
	COD into secondary sludge	12	kg	
AD	COD load	57		Primary + secondary sludge
	COD converted into biogas	0,5	%	(Khiewwijit et al. 2016), Mesophilic-35^{o}C
		29	kg	
	CH_4 content biogas	0,65	%	(Frijns et al. 2013)
	COD converted into CH4	19	kg	
	Theoretical max CH4 production	0,35	Nm3 CH4/kg COD	(Frijns et al. 2013)
	Produced CH4	6	Nm3 CH4	
	Calorific value CH4	35,9	MJ/Nm3	(Frijns et al. 2013)
	Produced energy	233	MJ	
	Recovery efficiency CH4	17	%	
CHP	Electricity conversion efficiency	0,4	%	(Verstraete and Vlaeminck 2011)
	Electricity produced	93	MJ	
	1 MJ=	0,000278	MWh	Conversion factor
	Electricity produced	0,026	MWh	
		0,001	MW	(Wh)/(h) = (W)
		2,59E-05	kWh	per 100m3 WWTP effluent
		2,59E-07	kWh/m3	
		6,21E-04	kWh/d	
	Recovery efficiency electricity	7,0	%	

Required photovoltaic area

The calculations are based on the Photovoltaic Geographical Information System (PVGIS) of the European Commission and an assumed ratio of 1 kWp PV capacity per 10m^2 installed PV area. (European Commission 2020).

PVGIS results	
Provided inputs	
Latitude/Longitude (Delft)	52.010/4.349
Horizon	Calculated
Database used	PVGIS-SARAH
PV technology	Crystalline silicon
PV installed (kWp)	1
System loss (%)	14
Simulation outputs	
Slope angle (°)	39
Azimuth angle (°)	3
Yearly PV energy production (kWh)	1024,66
Yearly in-plane irradiation (kWh/m2)	1262,75
Year-to-year variability (kWh)	47,61
Changes in output due to:	
Angle of incidence (%)	-3
Spectral effects (%)	1,74
Temperature and low irradiance (%)	-4,39
Total loss (%)	-18,85
	Energy producible (kWh/yr)
Delft: 1 kWp PV capacity	1024,66
	PV module sizing (m2)
1 kWp PV capacity	10
Field size Johan Cruyff Arena (m2)	7140

List of Publications

Journal Articles:

- Kehrein, Philipp, Mark van Loosdrecht, Patricia Osseweijer, Marianna Garfí, Jo Dewulf, and John Posada. "A Critical Review of Resource Recovery from Municipal Wastewater Treatment Plants – Market Supply Potentials, Technologies and Bottlenecks." Environmental Science: Water Research & Technology 6, no. 4 (2020): 877–910. https://doi.org/10.1039/C9EW00905A.

  This publication received the award *"Best paper from 2020 published in the Environmental Science journals of the Royal Society of Chemistry".*

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Oral presentations:

- Kehrein, P. (2019). A strategical planning and assessment framework to conceptually design a municipal WWTP from a resource recovery perspective. 12th IWA International Conference on Water Reclamation and Reuse, 18th of June, Berlin, Germany.

- Kehrein, P. (2019). Exploring Resource Recovery Potentials for the Aerobic Granular Sludge Process via mass and energy balances for COD and P recovery. 3rd IWA Resource Recovery Conference. 10th of September, Venice, Italy.

- Kehrein, P. (2019). From Wastewater Treatment Plants Towards Water Resource Recovery Facilities. Águas do Tejo Atlântico - Caminho da Inovação 2019, 26th of September, Lisbon, Portugal.

- Kehrein, P. (2019). From Wastewater Treatment Plants Towards Water Resource Recovery Facilities. EU cluster event water, 22nd of October, Girona, Spain.

- Kehrein, P. (2021). Resource Recovery Potentials & Bottlenecks. PAO Techniek en Management. Cursus: Circulaire Waterzuivering: de laatste inzichten, 10th of June, Amersfoort, Netherlands

Curriculum vitae

Philipp Kehrein was born on the 18th of January, 1985 in Wuppertal, Germany. In 2005 he completed his high school diploma at the "Gymnasium an der Siegesstraße" in Wuppertal. After a 3 year traineeship in hotel management in Frankfurt am Main and collecting work experience abroad, Philipp followed his interests in sustainable development and began to study in this field. He obtained a B.Sc. in Environmental Management from the Justus Liebig University Giessen in 2013. Afterwards he began the double degree European Master in Environmental Science - Soil, Water, Biodiversity and obtained his M.Sc. from the University of Hohenheim and the University of Copenhagen in 2016.

Already during his multidisciplinary studies he became passionate about waste management solutions and urban water management. During an internship at the federal state agency for environmental technologies in Baden-Württemberg, Germany (Umwelt-Technik BW), Philipp learned about circular economy solutions by reporting on the opportunities and challenges of ecological industrial parks. He became fascinated by the concept of converting waste streams into resources that can be reused. Trying to link his growing interests in (i) waste management; (ii) circular economy; (iii) urban water management; and (iv) sustainable development, Philipp conducted his master thesis at the University of Copenhagen in the field of energy and biochar recovery from sewage sludge via gasification and pyrolysis. He focussed on the carbon sequestration potential of biochar and the use of syngas as a renewable energy carrier to estimate the global warming potential of both processes.

Looking for opportunities to learn more about technological solutions that allow a circular resource flow within the urban water cycle and measuring the sustainability of those systems, Philipp started his PhD as an early stage researcher in the Marie Skłodowska-Curie actions (MSCA) funded research project SuPER-W (Sustainable Product, Energy, and Resource Recovery from Wastewater) in 2016. The research was conducted at the Biotechnology and Society Group in the Department of Biotechnology of the Delft University of Technology as a PhD student with Prof. dr. P. Osseweijer as promotor. It included a six months research stay at the Sustainable Systems Engineering group led by Prof. dr. ir. Jo Dewulf, Department of Green Chemistry and Technology, Gent University in 2019. In addition to his research activities, Philipp completed several trainings within the European Training Network (ETN). Since then, Philipp has worked on the research presented in this thesis and became afterwards a Postdoctoral researcher at TU Delft in January 2021 in the WATER-MINING research project.

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"A person who is nice to you, but rude to the waiter, is not a nice person."

Dave Barry

Have being a waiter for many years myself, I truly believe that all people directly or indirectly involved in this dissertation are nice to waiters.

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A very positive contribution that I am very thankful for came from my co-promoter **Jo Dewulf** who supported me especially during my research stay at his STEN group at Gent University. From you **and your fantastic Sustainable Systems Engineering group** I learned a lot about how to properly assess innovative processes and how to conceptualize resource recovery systems. I really enjoyed the good atmosphere in the STEN group and the good vibes among the colleagues. All in all Ghent University made important contributions to the work described in this dissertation.

The involvement of my co-supervisor **Anna Garfí** (Universitat Politècnica de Catalunya) was another very positive experience. Your availability and support especially at the beginning of the project was of great importance and I appreciate your contributions despite the distance.

Moreover, it is important to highlight the various contributions to chapter 4 that was a great team effort and collaboration that I could have never realised by myself. First I want to thank my friend **Morez Jafari** for being an absolutely reliable partner in this project. You were humble enough to leave me the first authorship although we led the project together and you would have deserve it equally. My big thanks goes also to **Marc Slagt** (DuPont Water Solutions) who showed great commitment for this project and widened my perspective. I learned a lot from our numerous meetings and am thankful for the many insights from an application point of view that is sometimes difficult to get in academia. I am also grateful for the involvement and contributions from **Emile Cornelissen** (KWR, UGent). Your critical view and friendly contributions certainly shaped this chapter and improved the results significantly.

I also want to thank **Ralf Lindeboom** and **Lisa Scholten** (both TU Delft) who took time to meet and discuss about different water management aspects. I really benefitted from your experiences and insights and appreciate your time and insight sharing.

A very special thanks goes to all the colleagues that were part of the **BTS group** between 2016-2020. Our lunch meetings have been an anchor point in my daily PhD life and the discussions we had were always a great inspiration. I am lucky to call several of you my friends now and hope we manage to stay in touch way beyond this project.

I feel the same for all the great **colleagues within the SuPER-W project** especially **Gijs Du Laing** (UGent) for creating and managing this great project. It was a great group of young researchers, professors and partners and I believe we established something special. I hope to keep in touch with all the special colleagues I had the pleasure to work with during this project.

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