TESE DE DOUTORAMENTO

UNRAVELLING ENVIRONMENTAL AND ECONOMIC CRITERIA FOR RESOURCE RECOVERY IN CENTRALISED AND DECENTRALISED WASTEWATER TREATMENT

Andrea Arias Cisterna

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Dña. Andrea Arias Cisterna

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Dona María Teresa Moreira Vilar, Catedrática de Enxeñería Química y Don Gumersindo Feijoo Costa, Catedrático de Enxeñería Química

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ABBREVIATIONS

AD	Anaerobic Digestion
aMBR	Anaerobic Membrane Bioreactor
AOB	Ammonium Oxidising Bacteria
AS	Activated Sludge
AWARE	Available Remaining Water Method
BES	Bioelectrochemical Systems
BW	Black Water
CC	Climate Change
CEPT	Chemical Enhanced Primary Treatment
CHP	Cogeneration Heat and Power
COD	Chemical Oxygen Demand
CW	Constructed Wetlands
ELAN	Autotrophic Nitrogen Removal
EP	Eutrophication Potential
ERBF	Enhanced Rotating Belt Filter
EROI	Energy return on investment
EWR	Environmental Water Requirements
FD	Fossil Depletion
FET	Freshwater Ecotoxicity
FS	Full Scale
FU	Functional Unit
GHG	Green House Gas
GW	Grey Water
HRAS	High Rate Activated Sludge
HRT	Hydraulic Retention Time
HTAD	High Temperature Anaerobic Digestion
HT	Human Toxicity
HWC	Human Water Consumption
IFAS	Integrated Fixed Film Activated Sludge
KW	Kitchen Waste
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory
LS	Laboratory Scale
MBBRs	Moving Bed Bioreactors
MBRs	Membrane Bioreactors
MET	Marine Ecotoxicity

MLE	Modified Ludzack-Ettinger
MLSS	Mixed Liquor Suspended Solids
MSW	Municipal Solid Waste
NEB	Net Environmental Benefit
NOB	Nitrate Oxidising Bacteria
NPV	Net Present Value
OD	Ozone Depletion
OLAND	Oxygen Limited Autotrophic Nitrification-Denitrification
OM	Organic Matter
OMPs	Organic Micropollutants
PC	Primary Clarifier
PEI	Potential Environmental Impacts
PES	Polyethersulfone
PMF	Particulate Matter Formation
PP	Pilot Plant
RBFs	Rotating Belt Filters
SBRs	Sequencing Batch Bioreactors
SC	Secondary Clarifier
SRTs	Solid Retention Times
TA	Terrestrial Acidification
TET	Terrestrial Ecotoxicity
TH	Thermal Hydrolysis
TMP	Transmembrane Pressure
TN	Total Nitrogen
UASB	Upflow Anaerobic Sludge Blanket
VER	Volume Exchange Ratio
VFM	Variable Flow Method
VRM	Vacuum Rotatory Membrane
WC	Water Consumption
WFN	Water Footprint Network
WWTPS	Wastewater treatment plants

ABSTRACT

The world population is growing year after year in a context of climate change and water scarcity. Within this framework, the wastewater treatment plants (WWTP) must be adapted to recover resources such as nutrients, energy or reclaimed water to be more sustainable from the environmental and economic point of view. With these objectives in mind, centralised and decentralised strategies for wastewater treatment have been explored in recent years under the focus of sustainability, especially when planning to implement alternative treatment schemes. As an element to highlight in this doctoral thesis, a comparative approach between two completely different realities (centralised vs. decentralised) of water treatment is provided, trying to go further in the proposal of innovative technologies. It is relevant that the different scenarios also consider different scenarios not only in terms of the population served but also in terms of their implementation in countries with different economic scenarios. Therefore, the main objective of this thesis is to evaluate centralised and decentralised systems from an environmental and economic perspective.

The first section of this thesis, called "*Improving centralised wastewater systems*", consists of five chapters in which different centralized schemes were evaluated from an environmental and economic perspective. In Chapters 2 and 3, the main changes in the wastewater schemes are proposed for the sludge line. Firstly, anaerobic digestion for different sizes in real sludge lines was addressed (Chapter 2). The main objective of Chapter 2 is to define the scale of the treatment plant from which it is convenient to configure the anaerobic digestion technology according to environmental and economic criteria, as well as to identify the existing barriers that prevent its widespread implementation. Continuing with the improvements in the sludge line, the purpose of Chapter 3 is to analyse alternatives to enhance the anaerobic digestion, the degradability of the sludge and to increase bioenergy production. Thus, two sludge pre-treatments were considered for the sludge line: (i) one related to the addition of chemicals and (ii) another based on increasing pressure and temperature. The new proposal was explored and compared with conventional systems in economic and environmental terms.

Subsequently, the wastewater line was modified (Chapters 4 and 5). A strategy based on organic matter (OM) recovery in primary treatment and nitrogen removal in secondary treatment by partial nitrification-anammox was compared with a conventional approach. In Chapter 4, a virtual plant designed for 100.000 population equivalents with different schemes based on the above-mentioned strategy was evaluated. The three innovative schemes are: (i) upflow anaerobic sludge blanket (UASB) followed by an integrated fixed activated sludge (IFAS), (ii) high rate activated sludge (HRAS) + IFAS, and, (iii) rotating belt filter (RBF) + chemically enhanced primary treatment (CEPT) + IFAS. These schemes were compared with a primary clarifier (PC) followed by a conventional activated sludge (CAS) with nitrogen removal. The main objectives are to study energy production, effluent quality and sludge production and how these factors can affect the environmental and economic profile of the wastewater line. Finally, in Chapter 5, the schemes were virtually modelled in two real plants located in Denmark and Spain. The existing configuration was replaced by two schemes: (i) HRAS + IFAS and (ii) Enhanced Rotating Belt Filter (ERBF) + IFAS. In this way, the main inputs and outputs of the different systems were estimated to calculate the environmental and economic profile. Finally, this section will conclude with the up-scale of a technology for nitrogen removal in the side stream (Chapter 6). The main objective is to establish the minimum scale to reliably estimate the environmental and economic indicators. In this way, the study can help as a guideline to address the evaluation of smaller units such as those of decentralized systems.

The second section of this thesis "*changing the paradigm of wastewater treatment*" seeks to highlight the importance of decentralised systems for resource recovery focusing on energy and water with the aim of demonstrating environmental and economic benefits. This section consists of two chapters (Chapter 7 and 8). The main objective of Chapter 7 is to evaluate the performance of a membrane plant for the recovery of irrigation water in Turkey. The wastewater plant is designed for 2,000 equivalent inhabitants. In this Chapter, an indicator called AWARE (available remaining water method) was applied to measure the water scarcity and the benefits of water reuse. Finally, Chapter 8 compares the approaches of decentralised and centralised systems from a citizen

perspective. The study aims to analyse the decrease or increase in water consumption and carbon footprint of a citizen living in a neighbourhood that incorporates a decentralized or centralized wastewater treatment system.

Keywords: energy production, innovative technologies, decentralised wastewater, economic indicators, life cycle assessment (LCA), reclaimed water

RESUMO

RESUMO

Durante os últimos anos, a poboación mundial experimentou un crecemento substancial e prevese que esta tendencia continúe. Isto implica que se deban xerar máis alimentos, enerxía ou auga potable, entre outros bens para satisfacer ás necesidades de dita poboación. Non obstante, o consumo de máis recursos implica unha xeración maior de residuos como, por exemplo, plásticos, augas residuais, desperdicios alimentarios, etc. Estes residuos deben ser tratados correctamente para evitar problemas relacionados coa contaminación do medio ambiente. No tratamento de residuos, as plantas de tratamento de augas residuais son factores chave para garantir a redución da descarga de contaminantes ó medio, xa que, se non son xestionadas correctamente, poden causar graves problemas ambientais e incluso a mortaldade das especies que viven nel. Así mesmo, estas plantas de tratamento deben facer fronte a novos desafíos como a eliminación de contaminantes cada vez máis complexos tales como fármacos, hormonas ou fragrancias sendo cada vez máis sustentables dende o punto de vista económico, ambiental e social. Neste contexto, o termo residuo debe ser substituído pola palabra produto. Polo tanto, as depuradoras deben traballar non para eliminar recursos senón para recuperalos, é o que se coñece como pensamento de economía circular. De esta maneira, o que se consegue é transformar ás xestoras de residuos en biorefinerías. O que se pretende nesta tese é abordar este cambio a través de diversas configuracións e distintos esquemas de tratamentos. Para abordar este cambio, a tese foi dividida en dúas seccións, as cales vanse explicar a continuación.

Capítulo 1: Introdución. Este primeiro capítulo pretende servir como marco teórico e explicar o motivo principal polo cal se decidiu realizar esta tese. O marco teórico engloba a problemática actual relacionada cos problemas de contaminación de augas, cambio climático ou aumento da poboación, entre outros. Definiuse o concepto de economía circular e como englobar ás depuradoras dentro deste termo, así como as estratexias que se poden levar a cabo para conseguir este propósito. Estes sistemas poden estar formados por distintas tecnoloxías que teñen distintas funcións dependendo da súa aplicación.

Como un dos obxectivos desta tese é abordar a parte ambiental e económica de distintos tratamentos de auga, describíronse en detalle as distintas ferramentas e indicadores que se poden empregar para este fin. Ademais, estudouse o que levan feito outros autores no ámbito das augas residuais en combinación con criterios ambientais e económicos. Dita información recóllese nunha revisión bibliográfica que se inclúe dentro deste capítulo. Finalmente, para concluír este primeiro capítulo, resumíronse os obxectivos que persegue cada capítulo que forma esta tese.

A primeira sección da tese titulada "melloras para sistemas centralizados" recolle un total de 5 capítulos. Esta parte do traballo foi financiada por un proxecto chamado Pioneer_STP (polas súas siglas en inglés *The Potential of Innovative Tecnologies to Improve Sustainability of Sewage Treatment Plants*), o cal ten como obxectivo principal, avaliar os desafíos relacionados co tratamento das augas residuais dende un punto de vista holístico que se centra en conceptos como a recuperación de enerxía, a xestión de lodos, a redución de custos e a mellora da calidade dos efluentes en sistemas centralizados de auga. Para cumprir con estes propósitos, propuxéronse e analizáronse novos esquemas de tratamento de augas. Ditos esquemas foron avaliados dende un punto de vista económico e ambiental durante esta tese (dende o capítulo 2 ata o 6). A información contida nestes capítulos, así como os principais resultados explícanse a continuación.

Capítulo 2. Avaliar as barreiras ambientais e económicas asociadas co escalado da dixestión anaerobia.

Este capítulo centrouse na avaliación da dixestión anaerobia para distintos tamaños de plantas reais de tratamento. Analizáronse catro liñas de lodos situadas en distintas localidades de España que van dende os 25.000 habitantes equivalentes (a máis pequena) ata 1.000000 habitantes equivalentes. A principal diferenza é que a liña máis pequena non ten dixestión anaerobia mentres que as outras tres configuracións de lodos si que dispoñen desta unidade. O principal obxectivo foi definir a escala de planta de tratamento a partir da cal convén configurar a tecnoloxía da dixestión anaerobia dacordo con criterios ambientais e económicos, así como, identificar as barreiras existentes que impiden unha implantación xeneralizada.

Os resultados deste traballo indicaron que esta tecnoloxía axuda a reducir os impactos ambientais e custos económicos debido á recuperación enerxética a través do biogás. A planta de tratamento que non ten dixestión anaerobia presentou custos de operación máis elevados relacionados co consumo enerxético e de químicos. Ademais, non se obtivo ningún beneficio ambiental asociado á recuperación enerxética. Cando a tecnoloxía se incorporou nesta liña obtivéronse beneficios non só ambientais senón tamén económicos. Se ben é certo, ó incorporar a unidade a complexidade tecnolóxica incrementa. Así mesmo, están as limitacións relacionadas co seu rendemento. Unha opción para mellorar este rendemento é a introdución de residuos orgánicos ou agrícolas no propio dixestor. Non obstante, a dixestión anaerobia considérase un proceso lento debido a hidrólise (primeira etapa deste proceso). Polo tanto, hoxe en día búscanse solucións tecnolóxicas para acelerar este paso. Estas solucións así como da súa problemática estudáronse no capítulo 3.

Capítulo 3. Avaliación comparativa dos indicadores ambientais e económicos das alternativas de xestión de lodos destinadas a mellorar a eficiencia enerxética e a recuperación de nutrientes.

Como se dixo anteriormente, o papel da xestión de lodos xoga un papel importante dentro do esquema de tratamento. Polo tanto, a finalidade deste estudo foi analizar alternativas para acelerar a etapa de hidrólise, mellorar a degradabilidade do lodo e incrementar a produción de enerxía. Para este propósito, estudáronse dous pre-tratamentos de lodos: i) o primeiro baseouse na adición de químicos e outro ii) fundamentouse no incremento de presión e temperatura (proceso termal) para xerar estas melloras. As novas propostas compraráronse cun sistema convencional (sen pre-tratamento de lodos). Tamén, se incluíron na análise dous tipos de tratamentos finais dos lodos: i) aplicación á agricultura e ii) incineración.

O sistema convencional mostrou unha xeración enerxética menor e unha cantidade de lodo maior en comparación coas liñas de lodos que incorporaron o tratamento químico e termal. Polo tanto, estas últimas obtiveron un perfil ambiental menor. En termos de custos, os pre-tratamentos asomaron custos maiores relacionados coa fase de construción pero que poden ser amortizados máis rápido debido á produción enerxética. A pesar de que no proceso termal hai máis consumo enerxético que no químico, o lodo está libre de patóxenos e pode ser aplicado á agricultura directamente. Isto implica unha redución maior en custos de operación.

Con respecto ó tratamento final dos lodos, a aplicación á agricultura presentou mellores impactos, aínda que se debe prestar atención á concentración de metais pesados e microcontaminantes, xa que en grandes cantidades poden presentar un problema de contaminación do medio no que se aplican.

Unha vez estudada a liña de lodos dos sistemas centralizados, o seguinte paso foi propor modificacións en toda a planta de tratamento (liña de augas e lodos). As tecnoloxías usadas para este fin definíronse nos capítulos 4 e 5.

Capítulo 4: Procura da eficiencia enerxética nas plantas de augas de tratamento: avaliación ambiental e económica de opcións innovadoras

Os esquemas de tratamento estudados durante este capítulo baseáronse nunha nova estratexia de tratamento das augas que consiste na recuperación de materia orgánica no tratamento primario e a eliminación do nitróxeno mediante un proceso de nitrificación parcial-anammox. Un total de catro configuracións deseñáronse para unha planta de 100.000 habitantes equivalentes. Os esquemas baseados na estratexia descrita anteriormente (recuperación de materia orgánica + proceso de nitrificación parcial anammox) comparáronse cun fronte a unha estratexia convencional.

Os novos esquemas incorporaron as seguintes tecnoloxías: i) UASB (pola súa abreviatura en inglés, *upflow anaerobic sludge blanket*) seguido dun proceso de nitrificación parcial-anammox denominado IFAS (pola súa abreviatura en inglés, *integrated fixed activated sludge*), ii) HRAS (pola súa abreviatura en inglés, *high rate activated sludge*) unido a un IFAS, e finalmente, iii) RBF + CEPT (polas súas abreviaturas en inglés, *rotating belt filter; chemical enhanced primary treatment*). O esquema convencional foi un clarificador primario acoplado a un sistema convencional de lodos (CAS polas súas siglas en inglés, *conventional activated sludge*) con eliminación de nitróxeno.

Os esquemas propostos mostraron ser máis eficientes en termos de enerxía recuperada a través do biogás. Non obstante, este incremento non xerou unha diminución do perfil ambiental en todas as configuracións, xa que o baseado en RBF + CEPT + IFAS presentou maiores impactos en comparación cos outros sistemas tanto innovadores (UABS + IFAS e HRAS + IFAS) como co convencional (PC + CAS). Este incremento debeuse á adición de químicos que aínda que produza unha mellora no perfil enerxético incrementa os impactos ambientais e económicos. A mellor configuración en termos xerais foi: UASB + IFAS seguido do esquema HRAS + IFAS. O esquema UASB + IFAS presenta a vantaxe de que non precisa dunha liña de lodos moi complexa, xa que a xeración dos lodos é moi baixa nesta configuración. Isto pode axudar a resolver a problemática relacionada coa cantidade de lodos que xeran estes sistemas, así como a súa xestión e posterior aplicación. Outra vantaxe dos sistemas innovadores é que a redución enerxética en aeración cando se incorpora o proceso IFAS pode chegar ata un 13%.

Este capítulo serve para comprender que non todos os esquemas innovadores implican mellores resultados. Neste sentido, cando se incorpora unha nova tecnoloxía, non só é necesario unha validación tecnolóxica senón tamén unha validación dende o punto de vista económico e ambiental. Polo tanto, a ferramenta do análise do ciclo de vida permite axudar a valorar estas opcións, o cal pode axudar á hora de tomar decisión ó deseñar unha depuradora.

Capítulo 5: Consecuencias ambientais e económicas ligadas á recuperación de enerxía por medio de novos esquemas en plantas de tratamento reais

Este capítulo está ligado ó anterior debido a que se busca reducir a xeración de lodo e aumentar a produción enerxética. Propuxéronse varios esquemas de tratamento de augas residuais para abordar os desafíos nomeados anteriormente. As modificacións van dende as etapas de modernización, onde se inclúen unidades novidosas en procesos convencionais ata concepcións completamente novas, pasando por modificacións substanciais do diagrama de fluxo. As principais diferenzas con respecto ó capítulo anterior é que agora os esquemas foron implantados en dúas plantas de tratamento reais situadas en distintos países europeos (España e Dinamarca). Para esta avaliación é necesario usar ferramentas para modelar, optimizar e seleccionar a mellor configuración de planta dende un punto de vista técnico, económico e ambiental. En ambas plantas, usáronse datos reais de fluxos de entrada nas depuradoras. Os datos de enerxía, eliminación ou consumo de químicos obtivéronse a partir do modelado, que se fixo co software Matlab, que é un dos máis comúns para modelar plantas de tratamento de augas. Este traballo xurdiu dunha colaboración e estadía na Universidade Técnica de Dinamarca (DTU).

Primeiro, modelouse a configuración real de cada planta de tratamento que consisten en un clarificador primario máis un sistema de lodos activos. Unha vez se modelaron esas dúas plantas, o sistema convencional (clarificador primario + sistema de lodos activos) modificouse por dúas opcións alternativas: (i) HRAS + IFAS e (ii) ERBF (pola súa abreviatura en inglés, *enhanced rotating belt filter*) + IFAS. Os datos obtidos no modelado usáronse para calcular os perfiles ambientais e económicos de cada configuración mencionada anteriormente. As novas configuracións demostraron ser mellores en termos de custos e aforro enerxético, o que propiciou que o perfil ambiental e económico fose menor que nas configuracións convencionais (PC + CAS).

Neste capítulo, o modelado demostrou ser unha ferramenta de cálculo eficiente da cal se poden obter datos válidos para calcular perfís ambientais e económicos para obter unha perspectiva xeral da planta. Neste caso, os novos esquemas axudaron a mellorar o nexo auga-enerxía e conseguir que as plantas de tratamento sexan máis eficientes de maneira integral facendo posible englobar estes sistemas de tratamento dentro da economía circular.

Capítulo 6: Escalado dunha tecnoloxía innovadora para a análise dos impactos ambientais e económicos

Esta primeira sección remata cun capítulo que ten como meta demostrar a escala na cal unha tecnoloxía innovadora proporciona datos e valores fiables para realizar o análise de ciclo de vida e os custos económicos, asegurando o avance na dirección da eco-eficiencia. Así mesmo, a importancia de medir a incerteza das ferramentas de cálculo reside na súa aplicación para sistemas descentralizados (cada vez máis pequenos).

Este estudo avaliou os impactos ambientais e económicos dunha tecnoloxía de eliminación de nitróxeno autótrofo (ELAN® polas súas siglas en español, eliminación autótrofa de nitróxeno) dende a concepción de laboratorio (1,5 L) ata a escala real (2 unidades de 115 m^3) pasando por dúas unidades a escala piloto (200 L e 1,2 m^3). As emisións indirectas relacionadas co consumo de enerxía foron a principal causa de impacto en todas as categorías excepto a eutrofización. Tamén se observou que a medida que a escala incrementa o impacto diminúe.

Á hora de avaliar a fiabilidade dos datos, este estudo proporcionou que a mínima á cal ten sentido aplicar a análise de ciclo de vida é de 200 L, mentres que para os indicadores económicos fixouse en 1 m^3 de volume de reactor. Polo tanto, se estas ferramentas se aplican a escalas máis pequenas a incerteza dos datos pode condicionar os resultados.

A segunda parte da tese doutoral consistiu en estudar os sistemas descentralizados e está formada por dous capítulos (Capítulo 7 e 8) resumidos a continuación. Esta sección está baseada nun proxecto europeo chamado Run4Life (polas súas siglas en inglés, *"Recovery and utilization of nutrients 4 low impact fertilizer"*). Este proxecto adopta o concepto de economía circular mediante a recuperación enerxética, auga ou biofertilizantes. Con isto o principal obxectivo do proxecto é intentar cambiar o obsoleto concepto fin de liña que se aplica para o tratamento de augas residuais. Para levar a cabo este obxectivo o desenvolvemento tecnolóxico combinarase con factores económicos e ambientais, así como unha avaliación de riscos non só para os compoñentes que poden ser perigosos para a saúde das persoas senón tamén para o medio ambiente.

Capítulo 7: Estudo dunha planta de tratamento descentralizada localizada en Turquía baseada nun sistema de membranas para a recuperación da auga.

Os sistemas descentralizados xorden como unha alternativa ó concepto centralizado, que son implantados cerca do punto de xeración de auga residual para aliviar a dependencia co respecto ao esquema centralizado e facilitar a reutilización dos recursos cerca do punto de orixe. Con isto, o que se pretende minimizar son os impactos e custos de transporte, recolección e infraestrutura. A meta do capítulo 7 foi analizar dende o punto de vista ambiental unha planta de membranas para a recuperación de auga de rego en Turquía. A principal razón por escoller este país é que se ve gravemente afectado pola escaseza de auga sendo unha rexión con déficit, polo tanto a recuperación de auga para o rego é necesaria para contrarrestar o consumo de auga potable.

A planta de tratamento está deseñada para 2.000 habitantes equivalentes e os impactos ambientais incluíron a construción da planta, así como a operación. Neste capítulo para medir a escaseza de auga e o seu impacto á hora da reutilización utilizouse un indicador chamado AWARE (polas súas siglas en inglés "avaliable remaining water method"). Finalmente, tamén se realizou unha análise de sensibilidade para ver como afecta o tempo de retención de sólido ó funcionamento da unidade de membranas.

A auga de rego produciu créditos ambientais cando se aplicou para o rego das zonas verdes, o que verifica o seu beneficio de cara a mellorar os impactos das plantas. O alto consumo de enerxía na unidade de membranas mostrou os peores resultados en canto a operación da planta, mentres que a fase de construción (que habitualmente se considera non relevante) mostrou ter altos impactos ambientais debido á construción da membrana. Finalmente, non se observou ningunha influencia considerable do tempo de retención de sólidos no perfil ambiental.

Capítulo 8: Análise ambiental do servizo de tratamento de augas centralizado e descentralizado para a poboación que vive nunha mesma veciñanza

Neste capítulo comparouse a perspectiva centralizada e descentralizada dende o punto de vista do cidadán. A pregunta á que se respondeu é se a pegada de carbono e consumo de auga dun cidadán mellora se vive nun barrio que xestiona a auga residual das vivenda nun sistema descentralizado, ou pola contra, ten máis impacto. O estudo aplicouse a un barrio de Santiago de Compostela, onde a configuración existente comparouse con tres configuracións teóricas (unha centralizada e dúas descentralizadas).

Os resultados na análise ambiental mostraron mellores resultados que os sistemas centralizados debido á recuperación de auga e enerxía. O consumo enerxético nos fogares pódese reducir con estes sistemas, o que implica unha diminución da pegada de carbono. A demanda de auga de rego cóbrese na súa totalidade con auga rexenerada. Os sistemas descentralizados mostraron máis custos de construción, xa que son sistemas máis complexos. Porén, coa recuperación de recursos os custos operacionais son menores, o que implica un tempo de amortización menor. É certo que a incorporación destas configuracións tería sentido cando se trata dunha área residencial de nova construción e que se non se recuperan os recursos non aportaría beneficios ambientais e o impacto sería máis alto ca nas configuracións convencionais.

Capítulo 9: Conclusións xerais e perspectivas futuras

Neste capítulo intégranse os principais resultados e conclusións do traballo desenvolto na tese. Tamén se dividiu en dúas seccións (centralizada e descentralizada). En canto ós sistemas centralizados o mellor esquema resultou ser a configuración UASB + IFAS seguido no HRAS + IFAS. As configuracións baseadas no consumo de químicos tiveron peores resultados, polo tanto, sería recomendable non aplicalas. A primeira configuración ten a vantaxe de que a xeración de lodos é moi baixa, o que implica unha liña de lodos sinxela e con menores custos de construción.

As tecnoloxías descentralizadas mostraron perfís ambientais baixos debido á recuperación de recursos e á xeración de enerxía, créditos ambientais que poderían redundar en menores costes de operación e tempos de amortización dos equipos en comparación coas plantas centralizadas. A elección dun sistema centralizado ou descentralizado debe facerse tendo en conta parámetros como o número de habitantes, o fluxo a tratar, a economía da área (países en desenvolvemento ou desenrolados) ou o espazo dispoñible para a construción da planta. Non obstante, ambos esquemas (centralizado e descentralizado) deben buscar a recuperación de recursos e a redución de custos. Estes elementos xestores de residuos que hoxe en día considéranse como finais de liña, deberían incluírse dentro da economía circular nun futuro. É dicir, actuar como biorefinerías que xeran produtos que son necesarios para a poboación.

CHAPTER 1: **Introduction**

SUMMARY

The term sustainability is a word that is used globally in every area of society, but to some extent its indiscriminate use has lost its precise meaning. What we mean by sustainable development is not only the broad concept of environmentally friendly development but also comprises socio-economic consequences.

Social, economic and environmental dimensions of sustainability can be encompassed in what could be described as the guarantee of sufficient resources for the well-being and needs of present and future generations while preserving the availability of resources and integrity of the environment.

In the context of a world population that is growing every year, environmental problems such as global warming and water scarcity are becoming increasingly critical. For this reason, it is crucial to seek new alternatives in reducing energy demand, waste disposal as well as ensuring water quality. In this last aspect, wastewater treatment plants (WWTPs) play an important role.

Chapter 1 highlights the importance of including WWTPs within the philosophy of the circular economy. Two main strategies are explained and described: one focuses on centralised systems, while the other is based on decentralised wastewater schemes. Furthermore, throughout the thesis, these strategies will be evaluated from the environmental and economic point of view. Finally, the main goals and objectives of this thesis are presented.

TABLE OF CONTENTS-CHAPTER 1

1. GENERAL INTRODUCTION

1.1. Growing environmental concern about water as a finite resource

In recent years, the world population has increased, and it is estimated that this trend will continue at a high rate (Vergara-Araya et al., 2020). The increase in population will imply a growing demand for resources such as water, food or energy (Popp et al., 2014). The increase in resource consumption will put pressure on land, water resources and air quality. In addition, there has been a growing trend of research to promote biorefineries that exploit resources such as corn, wheat or barley for the production of first-generation biofuels that have the disadvantage of competing for land and resources in the food-fuel dilemma (Kumar et al., 2020). Finally, it is important to note that this population not only will consume more resources but also generate more waste such as wastewater, plastics or solids (Kehrein et al., 2020).

In this situation, one of the most serious problems that is becoming a global threat is water scarcity (Mekonnen and Hoekstra, 2016), which can be defined as the imbalance between water demand and availability and can have negative effects related to the ecosystems deterioration, salinity intrusion or soil alteration (Jiang, 2009). Thus, climate change (aridity or drought) with the population growth are the main contributors to water scarcity.

However, in addition to these two problems, there is another problem: water pollution. Currently, it is estimated that about 12% of the world population does not have access to basic sanitation, and about 29% has only a basic system (WHO/UNICEF, 2017). This implies that the nutrients present in wastewater can affect water quality and cause major environmental problems such as eutrophication, water toxicity or even species mortality (Kobetičová and Černý, 2019). To try to solve and ensure access to sanitation for the whole population, a large investment in wastewater treatment plants (WWTPs) has been made in Europe (Pereda et al., 2020).

WWTPs proved to be very effective in removing nitrogen, phosphorus, and organic matter. However, in recent years, emerging pollutants called organic micropollutants (OMPs) have emerged as one of the problems in the WWTPs. OMPs are defined as anthropogenic or natural substances that include personal care products, pesticides, drugs, hormones or pharmaceutical compounds, among others (Barbosa et al., 2016). OMPs can contaminate groundwater, soil or vegetables. WWTPs are not designed to remove these compounds and may promote their dispersion and distribution in the environment (Bellver-Domingo et al., 2017).

Beyond the issue of micropollutant removal, WWTPs are characterised by high energy consumption in the process of removing pollutants and generate a significant sludge production that must be managed correctly (Gu et al., 2018). Therefore, these problems can increase the costs of wastewater treatment. In addition, these systems may be considered environmentally and economically unsuitable. Therefore, it is necessary to improve these elements in order to have more sustainable systems.

As part of the effort to minimise environmental problems and ensure access to safe water and sanitation systems, the United Nations promoted the adoption of Agenda 2030 for Sustainable Development (United Nations, 2015). In this Agenda, 17 objectives were developed for the protection of people and the planet. With regard to water protection, Goal 6 refers to "clean water and sanitation", which specifies the improvement of water quality, waste minimisation, removal of OMPs and the recovery of wastewater products (United Nations, 2015). So, WWTPs should be adapted to the new demands of the population and must be improved. However, to achieve this purpose, the best way to develop wastewater treatment plants must be sought from an environmental, economic and social point of view, as well as the best treatment strategy to ensure global sanitation and product recovery.

The previous answers will be developed in the following sections of Chapter 1 where the change of philosophy of wastewater will be explained through the concept of circular economy (Section 1.2. Circular economy in the wastewater treatment sector). Then, two strategies to improve the wastewater treatment sector will be defined and explained (1.3. New wastewater treatment strategy for centralised systems and 1.4. Decentralised approach for the wastewater treatment sections). Then, the environmental strategy (1.5. Life Cycle Assessment methodology and its application to wastewater treatment) and economic strategy (1.6. Economic evaluation) will be explained, and finally the main objectives and motivations for this thesis will be summarised.

1.2. Circular economy in the wastewater treatment sector

As mentioned before, in the past, WWTPs have been considered end-of-pipe systems with the main objective of treating a waste and discharging it into the aquatic environment. This end-of-pipe model is known as "linear" economy, which consists of one-directional model in which resources are used to produce goods that are purchased and, finally, the goods are disposed of after a single use (Figure 1.1) (Esposito et al., 2017). Detrimental air environmental quality, long-term economic stability or unsustainability are the main problems of this type of system (Millar et al., 2019). Currently, and in order to try to solve these problems, end-of-pipe systems are being replaced by the approach of circular economy based on a circular flow model. The main objectives are to promote resource recovery, minimise environmental impact and, at the same time, encourage growth (Figure 1.2) (Andersen, 2007).

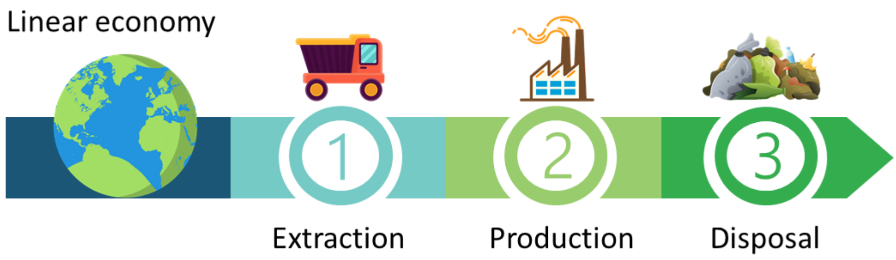

Figure 1.1. Linear economy philosophy

Figure 1.2. Circular economy perception

In this framework, the role played by the WWTPs (end-of-pipe elements) should be modified and adapted to this new "circular" philosophy. For this objective, the resource recovery can be a solution. Nutrients (nitrogen and phosphorus) reclaimed water and energy recovery are key factors in meeting this objective. In addition, the reduction of sludge production, as well as energy, have been under the focus for improvement (Leyva-Díaz et al., 2020). These goals can be achieved through the modification of the wastewater treatment strategy for centralised systems (section 1.3) or the implementation of decentralised wastewater treatment schemes (section 1.4). To better understand these concepts, both are explained in the following sections.

1.3. New wastewater treatment strategy for centralised systems

The conventional approach to wastewater treatment is based on large-scale systems in which wastewater is collected through an extensive sewer network. In general, this implies high construction and operational costs (Massoud et al., 2009). Moreover, the environmental impacts can increase in these wastewater schemes since they are characterised by high energy consumption and large sludge generation (Tang et al., 2020).

It is widely known that one of the hotspots in wastewater treatment is the energy consumption in aeration for the biological process (Gikas, 2017). Conventional nitrification-denitrification is based on aerobic and anoxic conditions. In the first stage (nitrification process), ammonium is oxidised to nitrate or nitrite and then both are reduced to dinitrogen gas (denitrification process). Therefore, in nitrification, there is an electricity consumption while in denitrification, organic matter is needed (Iannacone et al., 2019). Energy consumption can vary between 0.3 kWh/m^3 to 0.6 kWh/m^3 (Wan et al., 2016). Additionally, the C/N ratio should be higher than 5. An insufficient C/N ratio implies the addition of an external OM source which can increase the operational costs and more sludge production (Jiang et al., 2019). Finally, the sludge has a lower methanisation factor because only 30-50% of volatile solids are transformed into methane (Cao and Pawłowski, 2013).

So far, much effort has been devoted to exploring new technologies and wastewater alternatives with the main objective of making systems more sustainable and circular (Gu et al., 2018). In this situation, the Anammox process, in which ammonium is directly converted together with nitrite to dinitrogen gas, was a very significant advance in wastewater treatment. In this way, the energy in aeration can be reduced and an external source of OM is not necessary (Vázquez-Padín et al., 2009). Several technologies have been developed to use this pathway to remove nitrogen. Integrated fixed film activated sludge (IFAS) (Malovanyy et al., 2015a); autotrophic nitrogen removal (ELAN, eliminación autótrofa de nitrógeno, in Spanish) (Morales et al., 2015a) or SHARON-Anammox (Van Dongen et al., 2001) are some of them. However, these technologies do not work properly with a high percentage of solids or a high C/N ratio. Additionally, temperature and pH can be limiting factors (Xu et al., 2015).

The problems can be solved with the implementation of a new wastewater strategy that has been maintained in recent years. This approach consists of recovering OM in primary treatment and removing nitrogen with a partial nitrification-Anammox unit. When OM is applied in primary treatment, solids are removed and not incorporated into the secondary treatment. In addition, primary sludge is more biodegradable than secondary sludge, so the methanisation factor is also higher. This implies a greater production of biogas that can be transformed into electricity and heat, making the WWTPs more self-sufficient in terms of energy (Pérez-Elvira and Fernández-Polanco, 2012). New technologies such as rotating belt filters (RBFs), chemically enhanced primary treatment (CEPT) or high rate activated sludge (HRAS) have been included as primary treatments (Gu et al., 2018; Rahman et al., 2019; Ruiken et al., 2013) and others more widespread such as upflow anaerobic sludge blanket (UASB) (Malovanyy et al., 2015b).

In addition to this technology substitution and change of strategy in the wastewater line, the sludge line was also improved. As mentioned above, research has been conducted in recent years on how to maximise energy production in wastewater treatment plants to make systems carbon neutral. Today, the environmental and economic advantages of the anaerobic digestion (AD) unit have been proven in great detail (Gianico et al., 2015). However, not all wastewater treatment plants include this type of treatment. AD process consists of four steps: hydrolysis, acidogenesis, acetogenesis and methanogenesis. The first step (hydrolysis) is a limiting step due to polymer and extracellular membrane protections (Dai et al., 2016). On the one hand, the main reason for not implemented the AD at all levels is associated with sludge production. That is, in small wastewater treatment plants, the sludge generated is not sufficient to guarantee the use of biogas. To solve this problem, the co-digestion process can be applied in WWTPs. This method is based on the addition of a solid waste rich in organic matter (Gu et al., 2020) and can improve the performance of the AD process to increase biogas production. Alternatively, the integration of a pre-treatment in the sludge line may also favour the process as this type of pre-treatments aim to accelerate the hydrolysis step, and to improve sludge dewatering.

Sludge pre-treatments are divided into four main types: thermal, chemical, physical and biological (Abelleira-Pereira et al., 2015; Neumann et al., 2016). Thermal pre-treatment consists of the solubilisation of complex organic matter by increasing temperature and pressure (Serrano et al., 2015). Optimal temperatures can range from 150 °C to 180 °C, while pressure varies from 600-2500 kPa (Elalami et al., 2019). This unit is used to work in cycles of about 30 or 60 min, and depending on the characteristics of the process, can achieve an energy increase of about 51% (Bougrier et al., 2008).

Biological processes are based on enzymatic hydrolysis or the addition of fungi/bio-surfactants (Zhen et al., 2017). The addition of these compounds works best at thermophilic temperature because the increase in this variable promotes the hydrolysis of the raw materials (Ge et al., 2010). The increase of methane can fluctuate between 25% and 69% (Bolzonella et al., 2012). Regarding chemical pre-treatments, when there is a chemical addition, alkaline and acidic chemicals are the most studied (Khiewwijit et al., 2015a). However, these alternatives can cause problems of precipitation or inhibition, so their addition must be done very carefully. For this reason, other methods have been studied such as free ammonia (Wei et al., 2018) or oxidation with ozone or H_2O_2 (Chacana et al., 2017; Yu et al., 2018). The main problems of this type of pre-treatment are that ammonia can inhibit the AD process and oxidation requires a lot of energy and a large consumption of chemicals. Thus, these factors can penalize these pre-treatment schemes when introduced into the sludge line.

Finally, physical pre-treatment can be divided into high pressure, lysis, microwave and ultrasound. High pressure is similar to thermal pre-treatment but only by increasing operating pressure. There are several publications in which this pre-treatment can achieve a methane enhancement of 60-80% (Engelhart et al., 2000; Khiewwijit et al., 2015b). Lysis is a simple pre-treatment that causes partial cell destruction and improves the biogas field by about 15-26% (Dohányos et al., 1997). As for microwave and ultrasound methods, in addition to improving biogas and sludge dewatering, they can also help eliminate pathogens in the sludge. However, these processes can be energy-intensive, so biogas yield should be higher than in the other scenarios. However, the development of these pre-treatments is still in the laboratory or pilot plant (Feng et al., 2009; Neumann et al., 2016).

Although there are a wide range of sludge pre-treatments, they all have the same objectives, namely, to improve biogas production for more independent energy systems and to improve sludge dewatering. In this way, operational costs related to sludge management can be significantly reduced. Therefore, the new design of the WWTP can include all these concepts. However, it is true that many of these technologies are still under development and more research is needed to ensure that these assumptions are fulfilled.

1.4. Decentralised approach for wastewater treatment

As mentioned above, wastewater treatment is constantly changing to seek different approaches that are more sustainable. Within this framework, the decentralised system of wastewater treatment has gained strength in recent years (Hophmayer-Tokich, 2006). These systems are based on the separation of wastewater generated at different points in a household. Black water (BW) is generated in toilets, while grey water (GW) refers to water from laundry, showers, sinks or dishwashers (Ashok et al., 2018). Finally, kitchen waste (KW) is organic waste and can be treated with the BW or separately.

The main advantages of these systems compared to the centralised perspective are flexibility and the elimination of long sewer systems (Leigh and Lee, 2019). In addition, water reuse and nutrient recovery are increased due to source separation. While BW and KW are more appropriate for energy and nutrient recovery, GW is used in irrigation because the concentration of pollutants is very low (Kobayashi et al., 2020). These systems are also more appropriate for rural areas and developing countries because investment and maintenance costs can be economically more viable than conventional systems (Machado et al., 2017; Zeng et al., 2017).

Decentralised systems combine technologies that are applied in conventional wastewater treatment plants and more innovative ones. Regarding GW, several technologies have been studied in recent years (Ashok et al., 2018; Boyjoo et al., 2013). The most applicable are constructed wetlands (CW) due to the simplicity of operation and low energy consumption that implies lower operating costs (Garfí et al., 2017; Wu et al., 2015). In simple terms, this technology is considered as a complex natural bioreactor in which iterations occur between plants, soil and sediments (Corroto et al., 2019). The type of vegetation, substrate, microorganism and physicochemical parameters are key factors for its application (Corroto et al., 2019; Hijosa-Valsero et al., 2011). However, these systems require a large land area, which can be troublesome for their implementation (Arden and Ma, 2018).

In this context, membrane bioreactors (MBRs), moving bed biofilm bioreactors (MBBRs) and sequencing batch reactors (SBRs) have emerged as an alternative for treating GW. These systems are more compact, so the land use is lower than in GW and provide a high-quality effluent. However, these technologies are more operationally complex and electricity consumption is higher than in CWs (Cecconet et al., 2019; Jabri et al., 2020). The operation of these units consists of a combination of aeration and non-aeration periods. The main difference is that in MBR and MBBR there is a membrane integrated in the unit (Komesli and Gökçay, 2014), while in SBR the removal of OM or nutrients is accomplished (Vázquez-Padín et al., 2010a). The effluent can be used for irrigation of green areas, street washing or filling toilets (Chen and Wang, 2009) (Figure 1.3).

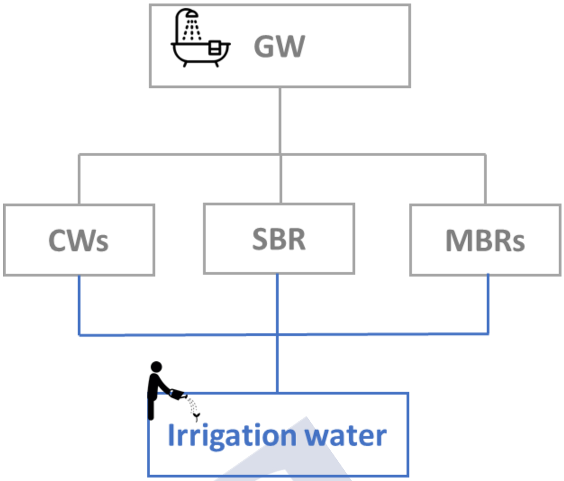

Figure 1.3. Most commonly used technologies for the treatment of greywater. Abbreviations: GW: grey water, CWs: constructed wetlands, SBR: sequencing batch reactors and MBR: membrane bioreactors.

BW and KW can be treated together or separately, but the goal is the same (nutrients and energy recovery). In terms of BW, it is important to distinguish the type of toilets that can be implemented in a house. Conventional toilets are the most common and vacuum toilets (new systems). In conventional toilets, water consumption is high, between 6-8 L per flush, while in vacuum toilets it is approximately 1-2 L per flush (Gao et al., 2019). This implies that in vacuum toilets wastewater is more concentrated and the production of biogas will be higher than in conventional toilets. However, vacuum toilets entail energy consumption and the noise generated by each flush can be very annoying (Bisschops et al., 2019).

The main technologies used to treat this type of wastewater are UASB (Kujawa-Roeleveld et al., 2006) and anaerobic membrane bioreactors (anMBR) (Pretel et al., 2016). Both are characterised by the transformation of OM into biogas, which is valorised into electricity and heat. In addition, high temperature anaerobic digestion (HTAD) has been developed in recent years to treat BW. The main difference with the other technologies is that this reactor works at temperatures of about 70 °C. This means that the water is free of pathogens and can be applied directly to agricultural irrigation (Zhang et al., 2020). AnMBRs and UASB can work at ambient or mesophilic temperature (about 35 °C). However, when these units work at ambient temperature, they may have problems with dissolved methane in the effluent, so this factor should be taken into account when applying them (Allegue et al., 2020).

In addition to energy, nutrient recovery is carried out in this type of wastewater (BW and KW). Within this framework, the struvite unit is the most studied method of phosphorus recovery and consists of a physical-chemical separation in which magnesium salts are added to facilitate struvite precipitation (Ishii and Boyer, 2015). In this unit, the pH is a key parameter and must be controlled in a range between 8-9 (Liu et al., 2008). In addition, many different types of reactors have been studied by different authors to achieve the best phosphorus recovery (Le Corre et al., 2009; Rahaman et al., 2014).

Other technologies focus on nitrogen recovery, such as stripping methods or bioelectrochemical systems (BES). Stripping methods and subsequent sorption in sulphuric or nitrous acid are highly energy-dependent (Bisschops et al., 2019). Bioelectrochemical processes could separate different types of ions such as NO_3^-, NO_2^- or NH_4^+ (Kuntke et al., 2018). The total ammonia nitrogen is concentrated by the influence of an electrical current and transported to the cathode. Nitrogen is then recovered through stripping. However, in this method, there is no chemical addition (Bisschops et al., 2019). The main technologies for treating BW and KW are summarised in Figure 1.4.

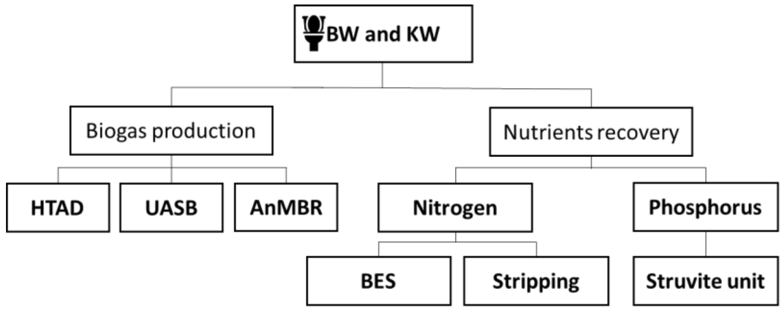

Figure 1.4. Main technologies for the treatment of BW and KW. Abbreviations: HTAD: high temperature anaerobic digestions; UASB: upflow anaerobic digestion; AnMBR: anaerobic membrane bioreactor; BES: bioelectrical systems

The previous section defined and explained a strategy that can be applied to centralized systems, as well as the processes and technologies used. In this section, an overview of decentralized systems was discussed. However, both strategies should ensure the concept of a circular economy and be sustainable (combination of environmental, economic and social factors). In this sense, the methodology of life cycle assessment (LCA) will be explained and implemented in these systems through the chapters of the thesis. Furthermore, this methodology will be combined with economic indicators to obtain a global vision of these technologies and treatment strategies when implemented to treat wastewater.

1.5. Life cycle assessment (LCA) methodology and its application to wastewater treatment

This methodology is defined as the quantification of the environmental impacts related to a product or process during its entire life cycle, including the extraction of raw materials until disposal or waste management (end of life) (ISO 14040, 2006). Four main steps are involved in this procedure: i) goal and scope definition; ii) life cycle inventory (LCI); iii) life cycle impact assessment and iv) interpretation. The main goals of these objectives are explained briefly below and can be summarised in Figure 1.5.

Goal and scope: the objective of the study is established, as well as the definition of the system boundaries. In this step, it is important to remark the delimitation of the boundaries. The methodology can take different approaches such as: cradle-grave, cradle-gate, gate-gate or gate-grave. In the first analysis, all flows are included (production of raw materials, transformation, use and waste), while the other studies only cover a part of the transformation of the material or the process. In addition, this section selects and includes the functional unit (FU) that should reflect the main function of the system or process under study (ISO 14040, 2006).

Life cycle inventory (LCI): This is characterised by the most crucial and time-demand stage. All inputs and outputs that take part in the system or process under evaluation are quantified. The different data that make up the system are referred to the FU (ISO 14040, 2006).

Life cycle impact assessment (LCIA): the inventory data are transformed into environmental impacts. There are different software packages for this transformation, such as Gabi or SimaPro. In this section, there are five steps where three of them (selection of impact categories, classification and characterisation) are mandatory and two (normalisation and weighting) are optional (ISO 14040, 2006).

Interpretation: this phase is carried out based on the interpretation of the main findings from the LCI and LCIA stages. Therefore, it is possible to identify the critical points, but also to propose measures or possible improvements to the system or process (ISO 14040, 2006).

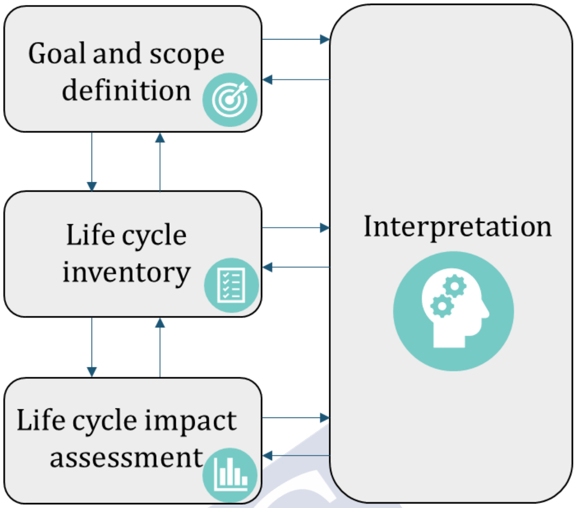

Figure 1.5. Life cycle assessment approach (ISO 14040, 2006)

This approach was widely applied from its early stages to the wastewater sector. If a quick search is made with the Scopus by selecting life cycle assessment and wastewater treatment plants as keywords, more than 530 documents have been published. In the first studies, the LCA methodology was applied in a simplified way to WWTPs. The main objectives were focused on comparing different scenarios for improving these elements, identifying hotspots or analysing different alternatives for sludge disposal (Lundin and Morrison, 2002; Tillman, 2000; Tillman et al., 1998). With the evolution of wastewater treatment schemes and technologies, LCA was also applied to more complex cases and more indicators such as construction phase, economic indicators or eco-efficiency were considered (Foley et al., 2010; Harclerode et al., 2020; Lorenzo-Toja et al., 2016b). In this framework, it is interesting to know the most common FUs, system boundaries, the used data or the most common environmental categories applied in WWTP LCAs.

As mentioned above, there are several publications that use LCA methodology to calculate the environmental profile in wastewater treatment schemes. Concerning FU, the most common are 1 m^3 of treated wastewater (Corominas et al., 2013; Pasqualino et al., 2011; Piao et al., 2016). Beyond the selection of volume as the most common option, there are others such as equivalent population (most used in decentralised systems) (Kalbar et al., 2013; Machado et al., 2007; Remy and Jekel, 2012) and kg of removed nutrients (Hauck et al., 2016; Zhu et al., 2013). However, with the trend towards more sustainable and circular systems, the FUs should be adapted to this new philosophy. In this sense, nutrients or energy recovery are proposed as FU (Roldán et al., 2020; Singh and Goldsmith, 2020).

In general, in WWTPs, the system boundaries are usually defined only as an operational phase because they account for more impact than the others, which can be considered insignificant (Álvarez et al., 2020; Rahman et al., 2018). Thus, in the operational phase, sludge management, electricity and chemical consumption and emissions into the atmosphere, among others, are collected to calculate environmental impacts. However, other studies have highlighted the importance of the construction phase, for instance Lorenzo-Toja et al. (2016a) estimated that the construction phase had an impact of about 35% in the climate change category. In addition, other authors reported the relevance of this phase (Arzate et al., 2019; Ioannou-Ttofa et al., 2017). In all these studies, the WWTPs function as an independent unit without taking into account the sewer network. In this sense, Risch et al. (2015) analysed the operation and construction phases of sewer including pipelines, road rehabilitation or civil works. The results showed that sewer has the worst impact on more environmental categories than the construction and operation of wastewater treatment plants. In addition, other authors also evaluated the importance of sewer (Morera et al., 2020; Petit-Boix et al., 2014). However, when there are not enough data, the inclusion of sewer can generate great uncertainty in the inventory and the outcomes of the analysis may be controversial. For this reason, it is not included in all LCA studies.

Once the system boundaries and FU have been defined, the next step is to conduct the LCI. The LCI data can be defined as primary data (real) or secondary data (bibliographic or estimated by modelling). Moreover, the missing data can be completed with databases such as Ecoinvent or Industry data, among others. WWTPs have been operated long before undergoing an environmental assessment, in this situation, real data of

WWTP construction and operation are used (Lijó et al., 2017). Furthermore, in innovative technologies it is more difficult to obtain real data, so the second option is the most used (Roldán et al., 2020; Taboada-Santos et al., 2020).

Traditionally, when the environmental impacts are calculated in a WWTP, the most representative categories are climate change (CC); eutrophication potential (EP) and toxicity categories (Zang et al., 2015). The CC category is related to electricity consumption due to fossil CO_2 emissions that can contribute to increasing greenhouse gas (GHG) emissions. In a context where WWTPs should aim to be energy neutral, this category becomes particularly relevant. EP is related to the discharge of nutrients such as nitrogen and phosphorus and can create problems related to excessive algae growth (Gallego et al., 2008). Finally, toxicity categories comprising human toxicity (HT), freshwater ecotoxicity (FET), marine ecotoxicity (MET) and terrestrial ecotoxicity (TET) have been gained importance in recent years and are associated with heavy metals or micropollutants (Li et al., 2019; Niero et al., 2014). It is important that the methodology, as well as the impact categories, are consistent with the objective of the study.

1.6. Economic evaluation

The estimation of the economic costs is a parameter to take into account to determine the viability in the operation of the treatment plants. Costs can be divided into capital costs and operating costs. Capital costs are related to the materials that have been manufactured by the different units of the process, including maintenance costs, among others. Operating costs are associated with sludge management, personnel, chemicals and electricity consumption. Beyond these costs, environmental prices can be estimated when calculating the costs associated with environmental impacts (De Bruyn et al., 2018). Environmental costs are considered as indirect costs. Therefore, indirect costs can be added to direct costs to have a more complete economic evaluation. In general, the most studied costs are operational because there is less uncertainty than in construction costs.

There are several authors who include life cycle cost (LCC) in the wastewater treatment sector. When searching the words life cycle cost and wastewater treatment plants in the SCOPUS database, more than 274 results have been identified. The first publications were related to the comparison of costs between different technologies at different levels (Tsagarakis et al., 2003; Uluatam, 1991). As in the case of the LCA approach, costs were more integrated into the WWTP with the evolution of these wastewater treatment elements.

Zessner et al. (2010) analysed construction and operating costs for different sizes of wastewater treatment plants in the economic framework of the Danube countries. The estimate, depending on the country, Austria presented the highest annual cost about 250 €/pe. Additionally, they concluded that the annual average price in terms of wastewater treatment was at least to 90 €/pe, which may be a high cost for the population. It is therefore important to reduce the cost associated with wastewater treatment in order to ensure the viability of sanitation. Mburu et al. (2013) also investigated the total costs of wetland-based treatment. The reduction in costs compared to traditional wastewater treatment plants was considerable. They established around 13.2 to 13.7 €/pe depending on the type of wetland. Instead of expressing costs based on the number of users or population equivalent, it is common to find results in terms of flow (cubic meters).

Rodriguez-Garcia et al. (2011) studied WWTPs of different sizes in Spain. The results ranged between 0.127-0.311 €/m^3 depending on whether the plants have nutrient removal. In the same country, Lorenzo-Toja et al. (2016b) also estimated the operational costs in different WWTPs and the conclusions were similar, around 0.044 to 0.344 €/m^3. Opening the scope to other countries, Li et al., (2017) evaluated different wastewater treatment configurations and priced between 0.705 and 0.892 yuan/t. For India decentralised plants with different UASB configurations were also evaluated (Khalil et al., 2008). In terms of resource recovery, the benefits associated with reclaimed water, energy or nutrients also was also conducted (Carr et al., 2011; Shen et al., 2015).

Other authors were focused on comparing different technologies but without their incorporation into a WWTP. For instance, Zepon et al.,

(2018) analysed tertiary technologies and sludge management alternatives or different nitrogen removal technologies (Jafarinejad, 2017). Moreover, other authors studied the total costs of a given technology. Pretel et al., (2016) estimated that an anMBR unit can be values between 0.03 to 0.12 €/m^3. In the case of MBR technology, there are more studies that estimated higher values of 0.08 to 0.25 €/m^3 (Gil et al., 2010). More recently, for decentralised technologies, Resende et al. (2019) studied wetland costs, between 1.55 $/$m^3$ to 0.84 $/$m^3$. However, it is important to note that the economic indicators can change considerably from country to country and over the years. These changes are related to changes in electricity, personnel or chemicals, among others. In this thesis, the costs will be adapted to the different configurations and countries and will be calculated taking into account the possible deviations.

1.7. Thesis outline: objectives and structure

The main goal of this doctoral thesis was to analyse and compare different wastewater treatment configurations from an environmental and economic point of view to provide insights on the sustainability of existing and innovative schemes of wastewater treatment. With this in mind, the thesis was structured in 2 sections: one for centralised systems and other for decentralised schemes. Section I is developed in 5 chapters, whereas Section II is composed by 2 chapters. Finally, the main conclusions of this thesis will be summarised in Chapter 9.

Chapter 1 presents the state of the art in the wastewater treatment sector. The main objective is to have a general idea about the problems of the wastewater sector, the importance of the circular economy and the different schemes that can be implemented to improve WWTPs. Moreover, the methodological tools used in this thesis will be explained to better understand its application.

Section I: Improving centralised wastewater systems. In this section different schemes and technologies were evaluated to improve the energy-water nexus from an environmental and economic point of view. Chapters 2 and 3 are focused on technologies for improving the sludge line at different sizes. In Chapters 4 and 5, wastewater treatment schemes will be changed. New configurations will be explored from an environmental and economic point of view to try to search more efficient configurations. In Chapter 5, two real WWTPs in different countries will be analysed and compared with the existing plant. The main objective of this work is to achieve more efficient systems that do not depend on the electricity grid, as well as to improve the quality of the effluents. Finally, the last chapter that takes part in this section (Chapter 6) has as objective to assess the scale-up of a technology focused on nutrient removal. The main reason for evaluating this technology is to verify the reliability of the LCA and economic indicators in small scale as, for example, in decentralised systems. In this way, these results can serve to have a reference when the decentralised systems (section II) are studied.

Section II: Changing the paradigm of wastewater treatment. This section is focused on the evaluation and implementation of different decentralised configurations. In Chapter 7, a wastewater treatment plant based on a MBR is going to analyse for recovering reclaimed water in Turkey which is a country with water deficit. Additionally, the construction phase will be included in the analysis to know how affect the construction in the decentralised wastewater schemes. In Chapter 8, two decentralised schemes are studied at neighbourhood level and compared with a centralised system with the main objective to know if the carbon footprint and water consumption of a person who lives in a decentralised area increase or decrease in comparison with a person that decide to live in a centralised zone. In this case, the chapters cover the concept of recovery (energy and water) but also from the inhabitant perspective.

Conclusions. The conclusions chapter aims to give a holistic and integrated view of the main findings and justifies the main contributions of the study. First, a comparison between different centralized schemes will be evaluated to show which is the best configuration in terms of energy, effluent quality and sludge production. Finally, in the decentralized schemes, the main findings and advantages of these systems in terms of energy and water recovery will be summarized.

1.8. References

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IMPROVING CENTRALISED WASTEWATER TREATMENT SYSTEMS

CHAPTER 2: **Identifying environmental and economic barriers associated with the scale of operation in the anaerobic digestion process**

SUMMARY

WWTPs are the most widely used environmental management systems to ensure that water pollution is properly managed. Since energy costs are the largest factor in operating costs, new installations are designed under energy optimisation parameters. The AD unit allows the valorisation of the organic load into bioenergy. However, not all WWTPs incorporate this technology in the sludge line since a minimum scale plant is required to guarantee stable and profitable operation of the unit. Small treatment plants imply a certain oversizing of electromechanical equipment, so that the unit consumption in such plants is relatively high. In large treatment plants, the design and sizing are optimized to achieve greater control over energy consumption. With the decentralized context gaining momentum, it is important to assess the viability of AD in small plants.

In this chapter, four different sludge lines with different plant sizes were evaluated from an environmental and economic point of view. The sludge lines range from 25,000 to 1,000,000 of equivalent inhabitants, although the small sludge line has no AD unit. A gate-to-gate approach was selected to perform the LCA. According to the results obtained in Chapter 2, the environmental impacts of the AD technology are not correlated with the size of the plant, so that not only medium and large-scale plants report environmental and economic benefits, but also smaller ones, provided that the premise of biogas flow valorisation into bioenergy is met. Moreover, the AD technology can be improved with the addition of agrowaste that can enhance the organic load in anaerobic digestor and improve the yield of biogas production and the eco-efficiency. This alternative allows to improve the technological, economic and environmental viability of the process.

Andrea Arias*[a], Gumersindo Feijoo[a], María Teresa Moreira[a]. "What is the best scale for implementing the anaerobic digestion according to environmental and economic indicators?" Journal of Water Process and Engineering. 2020, Vol. 32, 101235. https://doi.org/10.1016/j.jwpe.2020.101235. [a]CRETUS institute. Department of Chemical Engineering. Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain

TABLE OF CONTENTS-CHAPTER 2

2.1. INTRODUCTION

As mentioned in Chapter 1, the WWTPs are essential actors for the treatment of wastewater prior to its discharge into the environment (Pan et al., 2015). In this context, the configuration of new facilities is undergoing a process of dynamic change through the implementation of technologies that entail lower environmental impacts and economic costs (Gude, 2015). In general, the high costs related to sludge management and low energy production are two key factors that penalise the operation of WWTPs.

In terms of operational costs, sludge production can imply about 50% of the total costs in a WWTP (Lorenzo-Toja et al., 2016b). Among the different technologies of sludge treatment, the most widely implemented alternative is constituted by a thickening unit followed by homogenisation and dewatering units (Rodriguez-Garcia et al., 2011). In this scheme, sludge is treated as a waste, so there are no environmental or economic benefits. The most widely used alternative in WWTPs for the valorisation of biogas is the AD process. Moreover, the solid fraction can be used as fertiliser (Karagiannidis and Perkoulidis, 2009). However, not all WWTPs integrate this sludge treatment scheme, which is attributed to the need for a minimum size of the treatment plant to ensure stable and cost-effective operation of the unit. In the context of population growth, in which new treatment plants are planned to treat the wastewater of newly built dwellings with limited centralised services, there is an undeniable interest in assessing the viability of the AD technology at different sizes.

In this framework, it is interesting to combine an environmental approach with the economic or costs analysis associated with wastewater and sludge treatments (Nelson et al., 2008). Bearing in mind that this unit has significant benefits, the question arises as to why it is not a universal and undeniable option for any type of treatment plant. In addition, it is important to compare sludge lines lacking an AD unit with schemes that incorporate this technology in order to validate or rule out its implementation. With this in mind, the main goal of Chapter 2 was to evaluate the implementation of the AD unit not only on a technological basis, but also on the economic and environmental advantages that this unit may have in different plant sizes. Consequently, it is important to define the scale of the treatment plant from which it is convenient to set up the AD technology according to environmental and economic criteria and to identify the existing barriers that impede a generalised implementation.

2.2. MATERIALS AND METHODS

2.2.1. Goal and scope definition

Environmental and economic indicators of the different sludge lines of real WWTPs, all of them located in Spain, were evaluated. Four plant sizes were selected: i) one small (Scenario 0: 25,000 equivalent inhabitants); ii) two medium (Scenarios 1 and 2: 200,000 and 400,000 equivalent inhabitants, respectively) and, finally, iii) one large (Scenario 3: 1,000,000 equivalent inhabitants). The plants have different wastewater treatment flows, from 6,250 m^3/d for S0 to 213,410 m^3/d for S3. The plants are mainly based on the activated sludge process to remove OM. The small plant does not have a primary treatment, but only a pre-treatment to remove greases and solids, while the medium and large plants have a primary treatment to remove solids and OM. For all scenarios, a composting unit for the sludge was considered as a management option as a biofertiliser. The main differences correspond to the sludge line scheme.

The small plant (S0) consists of a thickening unit, a homogeniser and, finally, a filtration unit with a dewatering band filter. It is therefore a basic sludge line without an AD unit. The other plants have an analogous configuration, except for the fact that they include an AD unit of different size, coupled to a cogeneration heat power (CHP) unit to transform biogas into electricity (Figure 2.1).

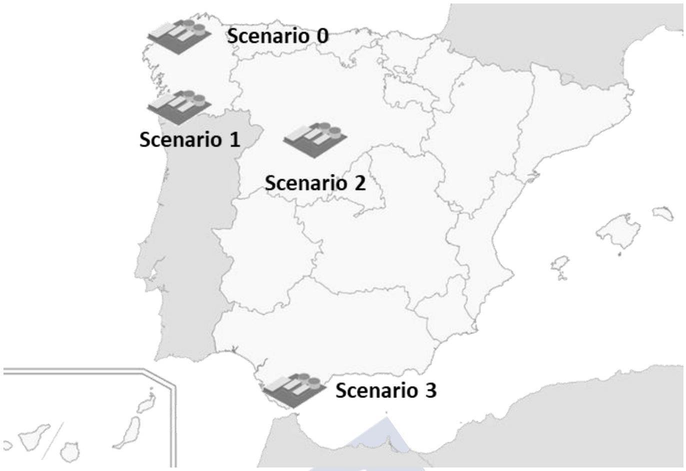

Figure 2.1. Different WWTP localisation considered in this study

2.2.2. Functional unit

In this case, the study is focused on biogas production, but it is not possible to choose 1 kWh of energy produced because the small plant does not have an AD unit, which would not allow the comparison of different types of plant. For this reason, 1 ton of mixed sludge was selected as FU, according to other publications related to the topic of sludge management (Dong et al., 2014).

2.2.3. System boundaries

To make the environmental assessment of the different sludge lines, only the impacts associated to the operational phase were taken into account. The environmental impacts related to the construction and decommissioning phases can be considered non-significant. This is because the operation of the facility is considered more relevant to the impact categories than the other phases (Lassaux et al., 2007; Lundie et al., 2004). All mass and energy flows of the different sludge lines were quantified. Figure 2.2 shows the system boundaries for the sludge lines.

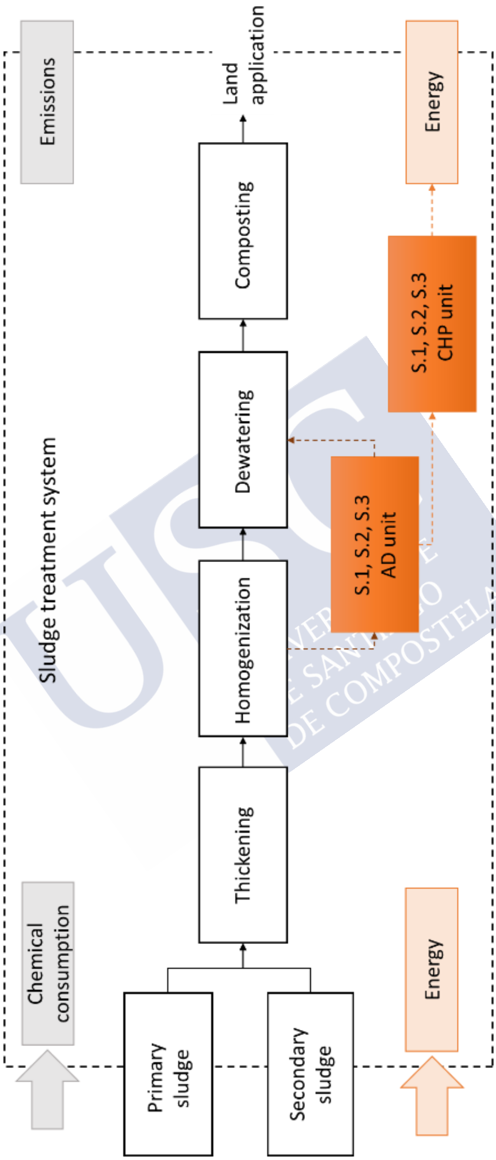

Figure 2.2. System boundaries of the different case studies

2.2.4. Life cycle inventory approach

Inventories were performed with primary (real data coming from the different sludge lines) and secondary data (estimated and bibliographic data). Primary data are associated to the characteristics of sludge such as nitrogen, phosphorus or heavy metals, electricity consumption and biogas production of the different plants. Secondary data comprise air emissions from the AD unit or sludge applied in agriculture (De Vries et al., 2012). In addition, the data were completed with the Ecoinvent 3.5 database (Weidema et al., 2013). Several simplifications were considered to make a more reliable LCI. All these data are presented in Table 2.1 (main inputs to the sludge lines) and Table 2.2 (main outputs to the systems).

The Spanish electricity mix has been updated with the most recent scenario according to the annual report of Red Eléctrica Española (REE, 2018). In addition, transmission losses associated to the electricity were taken into account. Euro 4 trucks with a capacity between 16 and 32 t were selected for the transport of chemicals and sludge. An average of 25 km was selected as a medium distance (Hospido et al., 2004).

Biogas losses were estimated at 1.5% of the total biogas production (Lijó et al., 2017) and air emissions associated with the application of sludge to the soil as fertiliser and to the composting plant were calculated according to the literature (Boldrin et al., 2009).

Table 2.1. Main inputs to the different systems (FU: 1 ton of mixed sludge)

	S0	S1	S2	S3
Inputs from the technosphere				
Materials and fuel				
Influent				
TS (kg)	200	200	200	200
VS (kg)	137	137	137	137
COD (kg)	220	220	220	220
TN (kg)	6.83	6.83	6.83	6.83
TP (kg)	10.74	10.74	10.74	10.74
Electricity consumption				
Thickening (kWh)	11.95	4.07	3.69	5.80
Homogenization (kWh)	9.43	3.21	2.91	4.57
AD (kWh)	–	84.93	70.99	103.79
Dewatering (kWh)	74.69	25.43	23.09	36.23
Composting (kWh)	2.81	4.01	3.35	4.90
Chemical consumption				
Dewatering				
Polyelectrolyte (kg)	2.30	0.72	1.66	2.16
Transport				
Polyelectrolyte (kg·km)	57.35	81.62	68.22	99.74
Sludge (kg·km)	13.50	9.5	11.25	7.75
Land application				
Agricultural machinery (kg)	0.54	0.38	0.45	0.31

Table 2.2. Main outputs to the different systems (FU: 1 ton of mixed sludge)

	S0	S1	S2	S3
Outputs to the environment				
Emissions to air				
AD				
CH_4 (kg)	–	0.70	2.37	2.97
CO_2 (kg)	–	1.14	3.84	4.82
H_2S (kg)	–	0.02	0.08	0.09
Composting unit				
CH_4 (kg)	2.41	3.43	2.78	4.19
CO_2 (kg)	1.41	2.01	1.68	2.46
N_2O (kg)	0.40	0.57	0.48	0.70
NH_3 (kg)	0.38	0.54	0.46	0.66
Land application				
N_2O (kg)	1.03	1.47	1.22	1.79
NH_3 (kg)	0.80	1.13	0.96	1.38
Emissions to water				
Land application				
NO_3^- (kg)	3.40	4.33	3.62	5.29
PO_4^{3-} (kg)	1.15	1.63	1.37	2.00

Table 2.2 (cont.). Main outputs to the different systems (FU: 1 ton of mixed sludge)

	S0	S1	S2	S3
Outputs to the environment				
Emissions to soil				
Land application				
Cr (mg)	22.30	14.65	14.65	23.73
Fe (mg)	12284	12268	12284	12283
Cu (mg)	139	197	395	126
Zn (mg)	300	395	297	319
As (mg)	9.30	10.25	12.20	17.90
Hg (mg)	0.44	0.62	0.62	8.59
Pb (mg)	44.39	33.49	33.45	94.84
Outputs to the technosphere				
Cogeneration				
Avoided electricity (kWh)	–	43.74	147.58	185.56
Avoided heat (kWh)	–	39.41	132.83	167.00

2.2.5. Life cycle impact assessment and interpretation

SimaPro 9.0 was the environmental software for calculating the environmental impact assessment. Two methods were selected to determine the most representative impact indicators. EP was calculated using CML 2001 method (Guinée, 2002), while CC, TA (terrestrial acidification), PMF (particular matter formation), HT, OD (ozone depletion), TET, FET, MET, FD (fossil depletion) and WC (water depletion) were calculated with the ReCiPe Midpoint (H) v1.1 method (Huijbregts et al., 2017).

Given the enormous importance of chemical organic matter (COD) concentration in the characterisation of effluent discharge, the selection of two methods is based on the approach to calculate COD-related impacts. In CML 2001, the impact associated with the COD concentration of the effluent has characterisation factor, whereas in the ReCiPe Midpoint (H) v1.1 method, COD is not considered in EP category. The inclusion of this parameter is considered particularly relevant in accordance with Spanish legislation, so both impact assessment methods have been selected (ECC, 1991).

2.2.6. Economic indicators

As in the case of environmental indicators, only operating costs were selected as economic indicators. These costs are related to the consumption of chemicals, electricity and the management of the sludge. Biogas and biofertilisers were considered as benefits (environmental credits). The main reason for considering construction costs as not significant is that they represent a minor contribution to the total operating costs. The infrastructure of WWTP has a useful life of between 25 and 50 years, so the costs can be amortised (Termes-Rifé et al., 2013).

2.3. RESULTS AND DISCUSSION

2.3.1. Environmental profile of the different sludge lines

In global terms, the environmental impacts were calculated for the different treatment schemes and scales, as shown in Table 2.3. All results are presented by FU (1 ton of mixed sludge). As expected, the largest plant could have the worst environmental profile based on inventory data of energy consumption and sludge generation. However, this is not the case, the small plant presents worse environmental results than the larger plant in categories such as OD, HT and FD. The main contributors to the environmental impact of the different plants are explained below.

The impacts of the largest plant are due to the direct air emissions related to CH_4, H_2S and CO_2 (Table 2.2), compounds present in the biogas losses that occur in the AD unit. Emissions in the PMF category are related to the indirect electricity emissions from non-renewable energy sources. Finally, impacts on ecotoxicity (TET, MET and FET) and EP categories are associated with the presence of heavy metals in the sludge. These values are higher in the operation of the largest plant than in the others. The presence of heavy metals in the sludge represents the need for careful monitoring to comply with the values recommended in the legislation.

It should be noted that the worst environmental profile of the plant lacking the AD technology is due to the indirect emissions associated with the polyelectrolyte consumption. Chemicals are necessary to ensure a satisfactory degree of sludge dewatering prior to the composting unit (Table 2.1). When the AD technology is applied, the sludge has better dewatering characteristics, so there is no need to add as much polyelectrolyte as in the case of sludge without an AD unit.

Table 2.3. Environmental results of the different sludge lines studied for the impact categories under assessment (FU: 1 ton of mixed sludge). Abbreviations: IC: impact categories S0: small plant; S1. and S2: medium plant S3: large plant

Impact categories	S0	S1	S2	S3
CC (kg CO_2 eq)	353.59	512.09	431.02	620.99
OD (kg CFC-11 eq)	$4.48 \cdot 10^{-3}$	$6.39 \cdot 10^{-3}$	$5.35 \cdot 10^{-3}$	$7.80 \cdot 10^{-3}$
TA (kg SO_2 eq)	2.61	3.64	2.72	3.99
EP (kg PO_4^{3-} eq)	4.69	6.64	5.48	8.00
HT (kg 1,4-DCB eq)	2.07	2.40	0.09	0.22
PMF (kg PM_{10} eq)	0.40	0.54	0.33	0.49
TET (kg 1,4-DCB eq)	226.86	300.22	165.19	256.04
FET (kg 1,4-DCB eq)	3.80	4.99	2.54	3.99
MET (kg 1,4-DCB eq)	3.88	4.95	1.97	3.25
WC (m^3)	0.37	0.36	0.17	0.21

The most representative categories in a WWTP are CC due to energy consumption, EP that is associated with the discharge of nutrients such as nitrogen or phosphorus, and, finally HT, which is caused by the heavy metals or chemical consumption, is important because it entails harmful effects to the human health (Rodriguez-Garcia et al., 2011). However, as the main objective of this work is to study the AD unit and the most dependent category is the CC, for this reason, this category was studied in more detail. The impact on the AD unit is related to biogas loss (Lijó et al., 2017), therefore, the first analysis was to compare how it affects the environment if biogas loss is zero.

The results are shown in Figure 2.3. In S0 (small plant), the environmental results did not change, because there was no AD unit. However, in the other plants they are even lower than S0. This means that, if the AD unit works properly and no biogas losses occur, the environmental impacts can be drastically reduced. Considering the potential benefit of the AD unit in the sludge line, a sensitivity analysis was performed for the smallest plant.

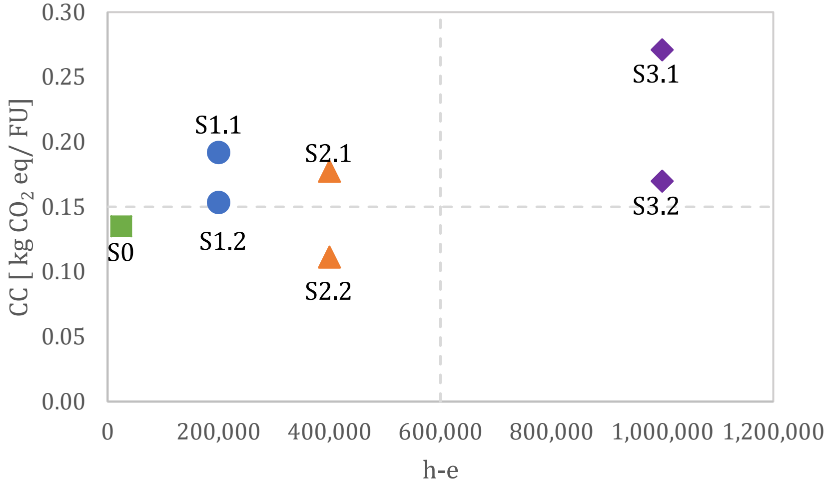

Figure 2.3. Comparison of the CC outcomes with and without biogas losses (FU: 1 ton of mixed sludge). Symbols: S0 (□): small plant; S1.1 and S2.1(o): medium plant without biogas losses; S1.2 and S2.2 (Δ): medium plant with biogas; S3.1 (◊): large plant with biogas losses; S3.2(◊): large plant without biogas losses.

2.3.2. Assessment of the feasibility of the anaerobic digestion (AD) unit

Considering the interest in implementing small-sized AD units, this section considers two main objectives: (i) whether or not the AD unit improves the environmental profile in S0, and (ii) to study the importance of energy recovery in sludge treatment.

The study was carried out for the CC category because this category is more sensitive and is directly related to energy consumption and biogas losses. As in the previous scenarios, biogas losses are estimated at 1.5% of the total biogas production. The results are shown in Figure 2.4. The integration of the AD unit shows an improvement in the environmental profile of around 10%. This positive effect is due to the production of biogas that allows a partial autonomy of the use of energy from the grid. This also means that, from an environmental point of view, the AD technology will be appropriate for this plant size.

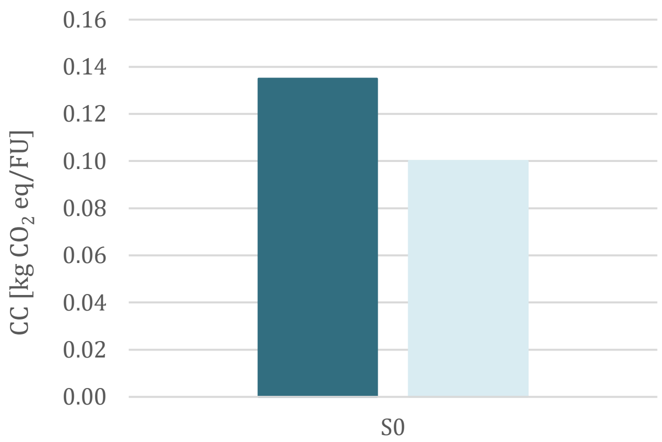

Figure 2.4. Sensitivity analysis of the small plant with and without AD technology

Some authors evaluate the incorporation of technologies such as UASB or AnMBRs in small communities (less than 2,000 equivalent inhabitants) because they can have benefits such as biogas production, which can make these small plants self-sufficient in terms of energy (Kujawa-Roeleveld et al., 2006; Pretel et al., 2016). However, for the treatment of primary and secondary sludge in this type of plant, extensive information on the operational limit in terms of size is not available. Pavan et al. (2007) studied the efficiency of AD technology with a population equivalent range of 1,000 to 3,000 inhabitants. However, this sludge was mixed with municipal solid waste. Therefore, for the AD technology to be appropriate on a smaller scale, it would be necessary to operate with a higher organic load, such as mixing sewage sludge with agricultural waste. The need to implement a cogeneration system suitable for smaller digester sizes should also be considered to ensure biogas valorisation.

This comment points out a recurrent situation in many WWTPs, where biogas is produced and burned directly in a torch. In this sense, it is important to highlight the role of energy production in achieving the water-energy nexus. The results of this analysis (two medium and one large plants) are shown in Figure 2.5. If biogas is not used in the WWTPs, not only can the environmentally impacts increase, but also the operating costs. In S1, environmental impacts may increase by 10%; in the case of the other plants, this increase in impacts is even greater: about 33% in S2 and 28% in S3. These results show the importance of biogas valorisation, which is crucial in the eco-efficiency profile of WWTPs.

Figure 2.5. Sensitivity analysis of the different plant sizes considering or not the energy use. Symbols: (Δ) without energy recovery; (o) with energy recovery

2.3.3. Energy benefit in the different sludge lines

To evaluate the energy benefit of different sludge lines, an indicator called Energy Return on Investment (EROI) was calculated. This indicator is useful to calculate the energy produced in the sludge line in relation to the energy consumed in the sludge line itself. If the indicator is higher than 1, the plant has a positive energy balance, which makes it energy self-sufficient. However, if the indicator is less than 1, the plant is not energy efficient. EROI indicator is represented by Eq.1 (Bisinella de Faria et al., 2015):

$$\text{EROI} = \frac{\text{Electricity produced}}{\text{Electricity consumed}} \quad [1]$$

Therefore, if the small plant has no AD unit, its EROI value is zero because there is no electricity production. However, when the AD unit is incorporated into the sludge line, the EROI value changes and is approximately 0.13, this means that about 13% of the electricity can be supplied by the biogas transformed into electricity. As for the other plants, the EROI values for medium-sized plants are 0.39 (S1) and 1.41 (S2). Finally, the value for the large plant is 1.19 (S3). According to these values, for S2 and S3 it is not necessary to consume energy from the grid in the sludge line. In addition, the management of the plant is crucial to have a satisfactory sludge-energy nexus. The difference between S1 and S2 are very significant when both plants are considered medium-sized plants. Therefore, to recover biogas and energy the plants must be properly managed.

2.3.4. Economic analysis of the different sludge lines

From the perspective of economic analysis, operational costs are different from those obtained by considering the environmental impacts (Figure 2.5). Consequently, the large plant (S3) presents the best economic results with an approximate value of 50 €/ton of mixed sludge, followed by the medium-sized plants with approximate values of 50-71€/ton of mixed sludge. The use of biogas in the plant itself can result in a benefit of between 11 and 9 €/ton of mixed sludge. These values are very important for reducing the operational costs. In the small plant lacking the AD unit, costs are higher (107 €/ton of mixed sludge). This can result in about 30% more in overall operating costs. In addition, in the small plant, there is a higher consumption of polyelectrolyte to achieve adequate sludge dewatering. Thus, if only the consumption of chemicals is compared, the operating costs increase by 98% compared to the rest of the plants.

Finally, operating costs related to sludge disposal are higher in the medium and large plants. This makes sense because the amount of sludge that needs to be managed, especially in the larger plant. The trend in small plants may change when the AD unit is incorporated into the sludge scheme. If the biogas is recovered and used in the plant, the total operational costs can be reduced by 10%. This reduction is not only due to the biogas production, but it also to the reduction of polyelectrolyte consumption, which also reduces the indirect emissions associated with the chemical consumption. Thus, the AD technology reduces operating costs, and the largest plant presents the most favourable costs.

Despite the positive economic indicators, Kalbar et al., (2012) argue that the AD technology cannot be implemented at all scales because the amount of sludge must be sufficient and guaranteed. In this sense, there are other residues such as agricultural, livestock or food waste. If this type of waste is introduced in the AD unit, the production of biogas will be higher, and the benefits will increase between 0.05 and 0.20 €/kWh of electricity generated. The range is very different because, as already mentioned, the type of waste is very important. For example, manure cannot have an acceptable efficiency in the AD unit due to the amount of water it contains (Vasco-Correa et al., 2018). In addition, these economic costs take into account the benefits of using sludge as biofertiliser.

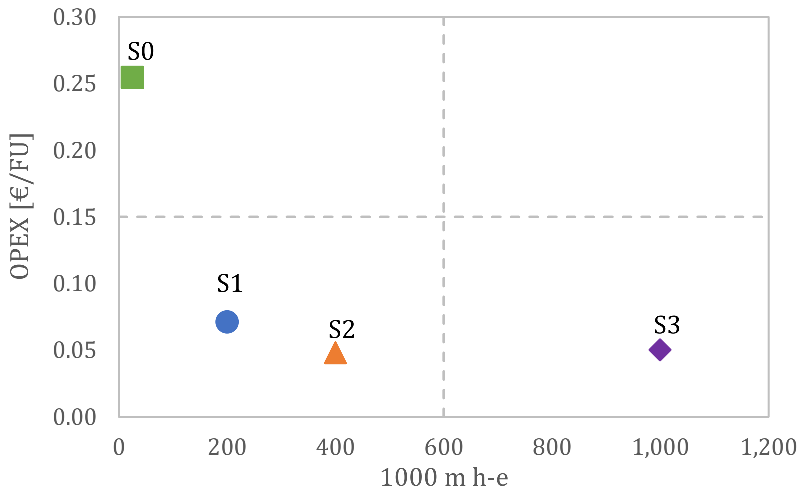

Figure 2.5. Comparison of the economic results from the different plant sizes (FU: 1 ton of mixed sludge). Symbols: □ small plant, o medium plant (scenario 1), Δ medium plant (scenario 2), ◊ large plant

In other words, the savings from not having to purchase mineral fertilisers, which can be around 50% of the total costs of fertilisers

(Frank, 1998). As mentioned above, a plant size of 25,000 equivalent inhabitants cannot be considered as a small plant. It is true that, in this case, the use of resources such as biogas or biofertilisers have a high variability costs and are more limited. This is because it is difficult to quantify the benefits of these products because sometimes the technology is not appropriate and does not allow the transformation of biogas into energy or the use of biofertilisers in agriculture (Borrion et al., 2012).

2.4. CONCLUSIONS

The AD technology proved to be a viable alternative in sludge treatment due to the generation of a green energy and a quality digestate that can be used in agriculture. However, this technology is not integrated in all plant sizes and is attributed to the need for a minimum scale. This study showed that the AD unit is a suitable environmental and economic alternative for sludge treatment, regardless of the plant size. Moreover, the use of biogas in the plant itself can improve the eco-efficiency of the WWTPs due to less dependence on the energy from the grid. This means less CO_2 emissions associated with non-renewable energy. In addition, the technological feasibility of the AD technology can be guaranteed in small plants as sewage sludge management could be combined with agricultural solid waste, which also implies a higher organic load in the digester and increased biogas production.

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CHAPTER 3: **Benchmarking environmental and economic indicators of sludge management alternatives aimed at enhancing energy efficiency and nutrient recovery**

SUMMARY

The main objectives of a WWTP are to remove the pollutants present in the wastewater, reduce the volume of sludge and improve the energy efficiency. The sludge treatment has a relevant role within the overall management scheme and can imply the largest share in operational costs. Considering the sludge treatment as a key factor to improve in a WWTP, the main goal of this Chapter is to evaluate different alternatives and strategies for sludge management and treatment from the perspective of LCA, with special emphasis on those options that reduce the environmental impacts and economic costs.

Two pre-treatments (one chemical and another thermal) and two post-treatments (composting unit followed by land application or incineration) were evaluated to improve the efficiency of the AD unit in terms of operation (biogas production and digested sludge), environmental and economic indicators. According to the results obtained, both sludge pre-treatments alternatives proved to be an adequate alternative to improve biogas production without negatively affecting environmental and economic impacts. If the final disposal of the digestate is analysed, its application to the soil as a biofertiliser is recommended, since it presents a better environmental profile than incineration. Nevertheless, soil application must be conducted under controlled conditions, avoiding exceeding the soil oversaturation, not only due to the potential eutrophication problems, but also to the presence of heavy metals that can lead to toxicity problems.

Andrea Arias*[a], Gumersindo Feijoo[a], María Teresa Moreira[a]. "Benchmarking environmental and economic indicators of sludge management alternatives aimed at enhanced energy efficiency and nutrient recovery". Journal of Environmental Management 2020 Nov 4; 111594. https://doi.org/10.1016/j.jenvman.2020.111594. [a]CRETUS institute. Department of Chemical Engineering. Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain

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3.1. INTRODUCTION

Nowadays, solving the WWTP problems associated with the sludge production and the electricity consumption are key factors to improve the eco-efficiency of these elements. The basic wastewater treatment schemes are based on the AD process, in which OM is transformed into biogas for heat and electricity recovery. As mentioned in Chapter 1, the hydrolysis stage conditions the reaction time, thus, it is important to foster this phase so that solid substrates are more accessible to anaerobic bacteria. The technical feasibility of different sludge hydrolysis processes based on chemical, thermal, biological and mechanical processes have been evaluated (Abelleira-Pereira et al., 2015). From the broad range of alternatives, it is important to identify the environmental and economic impacts associated with the most recommended technologies in order to check whether their implementation contributes to improving the energy efficiency of the treatment based on biogas production and operating costs.

In this context, the LCA methodology has been used to analyse and evaluate the environmental profile of different sludge management schemes (Hong et al., 2009; Tarantini et al., 2007). In addition, other authors have included thermal hydrolysis (TH) as a pre-treatment to improve sludge biodegradability and biogas production (Mills et al., 2014). Regarding the post-treatment alternatives for the management of biosolids after AD, thermal pyrolysis, land application or incineration processes have been explored (Cao and Pawłowski, 2013; Hospido et al., 2005; Murray et al., 2008). Recently, Dong et al. (2014) compared four post-treatment techniques: i) composting; ii) thermal drying-incineration; iii) co-combustion in a power plant and iv) cement manufacture for sludge treatment, but from an energy perspective.

Some documents focusing on sludge lines have considered the combination of environmental and economic perspectives (Murray et al., 2008; Xu et al., 2014). Despite the previous interest of these works, they are individual evaluations of different process. In a context in which new technologies are being developed to improve biogas production and sludge biodegradability, it is pertinent to study how these technologies are integrated into the sludge line, and the possible advantages or disadvantages compared to conventional schemes. At present, there is no report in the literature that addresses the integration and benchmarking of sludge management systems from an environmental and economic point of view.

The main objective of this study is to answer the questions posed above. In this sense, a conventional sludge line lacking sludge pre-treatment was compared with a modified scheme implementing two sludge pre-treatments: i) alkaline chemical pre-treatment and ii) TH. In this context, it is very important to discern how these processes are adapted to the existing sludge lines and to check their influence on the environmental and economic profile. After the AD process of sludge, its final disposal should also be considered on the basis of two premises: i) resource recovery as fertiliser for its application in agricultural soil after a composting stage and ii) recovery of its calorific potential in an incineration stage.

3.2. MATERIALS AND METHODS

3.2.1. Description of the different sludge lines and scope of the study

The goal of this study is to evaluate the environmental and economic indicators of three sludge scenarios that differ in the type of pre-treatment implemented in a facility designed for 500,000 equivalent inhabitants. Scenario 0 (S0) is the basic sludge line of a WWTP, which consists of a thickening unit followed by an AD unit and a dewatering system. Biogas is converted into energy and heat in a CHP unit. The electricity production will be used in the plant to power other systems and not be so dependent on the electricity from the grid. In addition, the heat produced in the CHP unit will be used to maintain the temperature in the AD unit.

In this baseline scenario, two pre-treatments to improve biogas production were evaluated: chemical (S1) and TH (S2). In addition, two techniques for sludge disposal were evaluated: a composting unit followed by land application and, alternatively, an incineration unit. It is important to note that, according to the European legislation Directive 86/278/CEE, thermally treated sludge can be applied directly to agricultural land, avoiding the need to include a composting unit in this scenario. That is, air emissions and energy consumption related to the composting plant are not considered in S2.

In order to explain the differences of the different pre-treatment alternatives in more detail, a description of both process units is presented below. The alkaline chemical pre-treatment (S1) allows the solubilisation of the sludge and the enhancement of its specific surface to facilitate access to anaerobic microbes. This implies an increase in the COD concentration and, consequently, the yield in the biogas production. This process can be carried out at room temperature thanks to the use of different chemicals such as potassium hydroxide (KOH), sodium hydroxide (NaOH) or calcium oxide (CaO) (Ariunbaatar et al., 2014). In this case, the chemical selected to improve biogas was KOH because it provided better results compared to other chemicals. The rationale behind this is the inhibitory effect of NaOH on the AD process, while CaO can cause operational problems related to precipitation of carbonates and phosphates (Kim et al., 2003). The TH pre-treatment (S2) is based on the disintegration of the floc structures under high pressure and temperature: 10 atm and 175 °C (Pérez-Elvira and Fernández-Polanco, 2008). After treatment, an increase in soluble COD and potential increase in biogas production is observed.

Regarding the system boundaries, the sludge line comprises the different pre-treatments, the AD process and the final disposal (Figure 3.1), considering only the impacts associated with the operational phase, in agreement with other works (Corominas et al., 2013). The FU was defined as 1 ton of mixed sludge (which represents the contribution of primary and secondary sludge) because it is the main objective of this study, which is consistent with other LCA studies on sludge management (Houillon and Jolliet, 2005; Suh and Rousseaux, 2001).

3.2.2. Life cycle inventory (LCI) for the different sludge pre-treatments

Considering that the key objective is to ensure greater production of biogas and therefore bioenergy, the production of electricity has been considered as an avoided product. This means that the environmental benefits of electricity production from biogas valorisation are considered as environmental credits. Moreover, taking into account the composting of the digestate or its incineration, the impacts related to these stages were calculated, such as direct emissions into the atmosphere or emissions into water, among others. In this study, LCI was developed with secondary data associated with the characteristics of the sludge generated in a conventional WWTP, and, bibliographic data related to chemical consumption, biogas losses and electricity consumption (Boldrin et al., 2009; Cano et al., 2015; Lijó et al., 2017). Finally, this secondary data was completed with the Ecoinvent v.3.5 database. The inventories are presented in Tables 3.1 and 3.2. Table 3.1 corresponds to the main inputs to the systems whereas Table 3.2 represents the main outputs. In addition, several simplifications have been considered for background data.

Electricity: the Ecoinvent v3.5 database has been updated for 2018 with data from the annual report of Red Eléctrica Española (Spain) (REE, 2018). The medium voltage electricity used in the WWTPs was modelled including transformation from high voltage (Lorenzo-Toja et al., 2016b); thus electricity transmissions losses in this process were also included (Dones et al., 2007).

Chemical consumption: the amount of polyelectrolyte used in the dewatering unit was calculated according to Tchobanoglous et al., (1998), assumed equivalent to cationic resin as reported in the Ecoinvent 3.5 database (Wernet et al., 2016). The amount of KOH required for alkaline hydrolysis was estimated based on the work of Kim et al. (2015).

Emissions to air (N_2O and NO_3) and **water** (NO_3^- and PO_4^{-3}) from the agricultural application were taken into account in the final impact (Bruun et al., 2006).

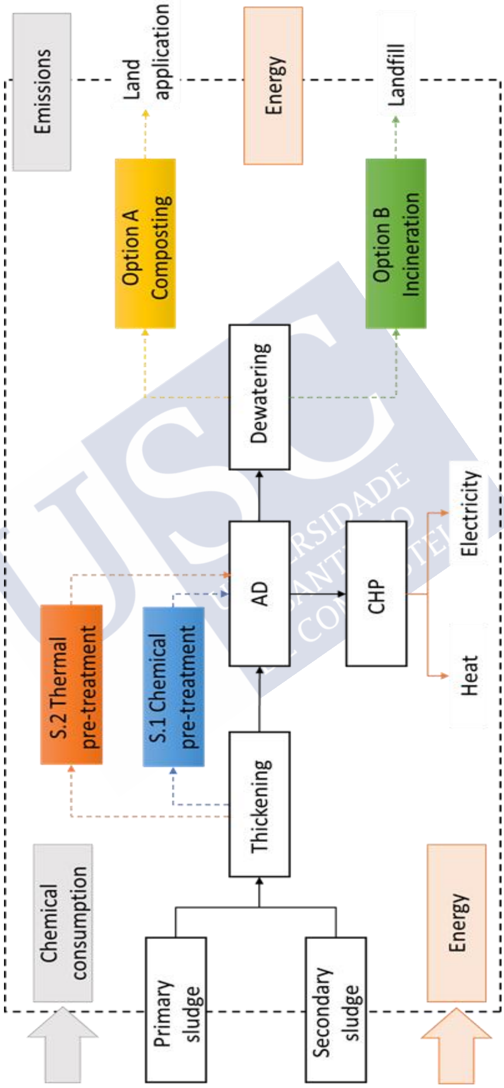

Figure 3.1. General system boundaries of the different case studies considered

Table 3.1. Main inputs to the different systems (FU: 1 ton of mixed sludge). Scenario a) composting plant; Scenario b) incineration plant

	Scenario 0	Scenario 1	Scenario 2
Inputs from the technosphere			
Materials and fuel			
Influent			
TS (kg)	100	100	100
VS (kg)	70	70	70
COD (kg)	126	126	126
TN (kg)	3.70	3.70	3.70
TP (kg)	6.90	6.90	6.90
Electricity consumption			
Thickening (kWh)	21.67	21.67	21.67
TH (kWh)	–	–	12.5
Chemical pre-treatment (kWh)	–	0.97	–
AD (kWh)	15.85	10.39	10.39
Dewatering (kWh)	5.46	3.58	3.58
Composting (kWh)[a]	1.30	1.30	–
Incineration (kWh)[b]	16.80	16.80	16.80
Chemical consumption			
Pre-treatment			
KOH (kg)	–	9.63	–
Dewatering			
Polyelectrolyte (kg)	1.60	1.60	1.60
Transport			
Polyelectrolyte (kg·km)	40	40	40
KOH (kg·km)	–	24.20	–
Sludge (kg·km)[a]	9.15	6.97	6.97
Ashes (kg·km)[b]	1.45	1.45	1.45
Landfill			
Amount of ashes (kg)[b]	$5.79 \cdot 10^{-2}$	$5.79 \cdot 10^{-2}$	$5.79 \cdot 10^{-2}$
Land application			
Agricultural machinery(kg)[a]	0.37	0.28	0.28

Table 3.2. Main inputs to the different systems (FU: 1 ton of mixed sludge). Scenario a) composting plant; Scenario b) incineration plant

	Scenario 0	Scenario 1	Scenario 2
Outputs to the environment			
Emissions to air			
AD			
CH_4 (kg)	0.43	0.69	0.87
CO_2 (kg)	0.84	1.35	1.70
H_2S (kg)	0.01	0.02	0.03
Composting unit [a]			
CH_4 (kg)	0.53	0.53	–
CO_2 (kg)	13.78	13.78	–
N_2O (kg)	$8.88 \cdot 10^{-3}$	0.01	–
NH_3 (kg)	0.26	0.26	–
Land application [a]			
N_2O (kg)	8.24	4.71	5.23
NH_3 (kg)	4.93	3.88	4.31
Emissions to water			
Land application [a]			
NO_3^- (kg)	5.50	3.14	3.49
PO_4^{3-} (kg)	4.26	2.43	2.43
Emissions to soil			
Land application [a]			
TN (kg)	2.00	2.00	2.00
TP (kg)	7.94	7.94	7.94
Cr (mg)	22.34	22.34	22.34
Fe (mg)	5676	5676	5676
Cu (mg)	603.49	603.49	603.49
Zn (mg)	754.49	754.49	754.49
As (mg)	9.21	9.21	9.21
Hg (mg)	0.95	0.95	0.95
Pb (mg)	51.07	51.07	51.07
Outputs to the technosphere			
Cogeneration			
Avoided electricity (kWh)	109	123.48	152.81
Avoided heat (kWh)	98.83	105.04	137.53

3.2.3. Environmental and economic indicators for the sludge pre-treatments

The different impacts were evaluated through two methods. EP was calculated using the CML 2001 method (Guinée, 2002), while CC, OD, TA, PMF, HT, TET, FET, MET, and FD were calculated using the ReCiPe Midpoint (H) method (Huijbregts et al., 2017). As in previous chapters, the main reason for choosing two methodologies is based on how to estimate the impact of the COD contribution.

Costs can be divided into operational and capital costs. The costs of construction, equipment or maintenance were calculated based on bibliographic data (Tables 3.3 and 3.4). In addition, in the operational costs, the disposal of sludge, electricity, chemical consumption and staff costs were included. Biogas that is transformed into electricity and heat was considered a benefit. In other words, the share of the total electricity from biogas will cover a fraction of the total requirements of the plant. The value of this electricity production is shown in Table 3.2. Thus, considering the price of electricity in Spain, this electricity production will be deducted from the total cost of electricity (Mills et al., 2014). Furthermore, in order to share the same FU as in the LCA methodology, the total costs are estimated per 1 ton of mixed sludge. The costs are represented by the Net Present Value (NPV) defined in Eq.1, where **n** is the time of useful life while **i** is the discount rate adjustment for inflation equal to 5% (Hermelink and Jarger, 2015).

$$\mathrm{NPV} = \mathrm{CAPEX} + \sum_{n} \frac{OPEX}{(1+i)^n} \qquad [1]$$

In addition, in this study, it is important to calculate the payback time according to Eq.2, where M_s. represents the mass of sludge production in a year (ton/year); C_d is the value related to the costs of the final disposal of the sludge in €/ton; ΔE is the difference in the electricity (production in the sludge line (kWh/year); C_e: costs of electricity is associated with the price of electricity and C represents the total capital costs.

$$\text{Payback time} = \frac{Ms\ x\ Cd + \Delta E\ x\ Ce - C}{Investment\ cost} \quad [2]$$

Table 3.3. Inventory data for operational costs

Economic item	Unit	Value	Source
Specialized worker	€/year	50,000	Longo et al., 2017
Unit cost of electric energy	€/kWh	0.12	Morales et al., 2015
Unit cost of polyelectrolyte	€/kg	1.8	Longo et al., 2017
Unit cost of KOH	€/kg	0.65	Carrere et al., 2012
Unitary cost for sludge composting and application	€/ton	90	Longo et al., 2017
Unitary cost for sludge incineration	€/ton	354	Hong et al., 2009

Table 3.4. Inventory data for the construction and maintenance costs

Economic item	Unit	Value	Source
Thickening unit	€	185,162	Mills et al., 2014
Anaerobic digestion unit	€	403,114	Mills et al., 2014
Cogeneration unit	€	386,098	Mills et al., 2014
Dewatering + silo unit	€	265,903	Mills et al., 2014
Chemical pre-treatment unit	€	60,000	Diamantis et al., 2013
TH pre-treatment unit	€	410,850	Mills et al., 2014
Composting unit	€	385,500	Chen, 2016
Incineration unit	€	1,925,000	Panepinto et al., 2016
Project timeframe	y	20	Mills et al., 2014
Interest rate	%	5	Longo et al., 2017
Maintenance costs for civil works	€	0.17	Hernández et al., 2006
Maintenance costs for electro-mechanic elements	€	1.24	Hernández et al., 2006

3.3. RESULTS AND DISCUSSION

3.3.1. Main parameters and life cycle results of the different sludge scenarios

Table 3.5 presents the main variables and parameters associated with the scenarios considered in terms of energy consumption, biogas production from the primary sludge of the clarifying unit and from the secondary sludge of the activated sludge process in terms of methane and electricity, as well as the reduction in the volume of sludge.

S2 (TH pre-treatment) presents the best results in terms of biogas production, followed by S1 (chemical pre-treatment). These pre-treatments can improve electricity production between 6% and 11% compared to the baseline scenario. In addition, the degradability of sludge improves by 30% when a pre-treatment is included in the sludge line. For S0 (AD only) and S1 (chemical pre-treatment), energy consumption is very similar as it must take into account that the amount of electricity associated with the dosing and mixing of chemicals is minor. The energy consumption of the TH pre-treatment is approximately 14% higher than in the other options. When the energy balance takes into account the final management of the sludge (incineration or composting followed by land application), composting presents a better energy balance than the incineration unit, which translates into differences of around 25% for this parameter. Finally, heat is used entirely to maintain the temperature of the AD unit at 35°C.

Table 3.5. Variables and operational parameters associated to the scenarios considered including final disposal of the sludge (FU: 1 ton of mixed sludge). Scenarios: a) composting and land application; b) incineration.

	Scenario 0	Scenario 1	Scenario 2
Energy consumption [a] (kWh)	61.08	54.71	64.94
Energy consumption [b] (kWh)	221.88	214.77	224.94
Biogas production (m^3)	54.04	57.44	75.21
Methane yield (m^3 CH_4/kgVS feed)			
Primary sludge	0.30	0.33	0.38
Secondary sludge	0.20	0.28	0.31
Electricity production (kWh)	109	123.48	152.81
Energy balance (kWh) [a]	-47.92	-68.77	-87.87
Energy balance (kWh) [b]	112.88	91.23	72.31
Heat production (kWh)	98.83	105.04	137.53
Sludge production (kg/d)	22876	17435	17435

The environmental profile is reported in terms of various impact categories (Table 3.6). The results show that the environmental impacts are very different depending on the category considered. In the case of chemical pre-treatment, greater environmental impacts are observed in categories such as TA, PMF and TET due to the indirect emissions associated with chemical production. However, when the TH and chemical pre-treatment are implemented in the sludge line, the avoided electricity may increase due to the greater amount of biogas, provided that the valorisation of biogas entails lower dependence of grid electricity. In addition, in S2a (TH pre-treatment), the composting plant is not necessary because, according to Directive 86/278/CEE, thermally-treated digested sludge can be applied directly to agriculture. In addition, impacts related to atmospheric emissions associated with the composting unit can be avoided (Table 3.2). However, it is very difficult to know the overall environmental impact of these pre-treatments due to the much larger impacts of the post-treatments.

In energy-dependent categories such as CC, OD or FD, the incineration unit has greater impacts than the composting unit followed by land application. This is due to the large amount of electricity consumed in this process. Indirect emissions are related to fossil CO_2 and N_2O from the coal electricity production. Conversely, scenarios with composting followed by land application present worse environmental profile than the incineration process in toxicity-related categories due to the presence of heavy metals in the sludge. In this case, only the heavy metals in the sludge were considered since the routine measurement of micropollutants is not carried out due to the complexity of the necessary equipment, sample preparation and costs. If the pathogens or micropollutants were included in this study, the toxicity categories would probably be the most affected, considering the application of the sludge to the soil. However, although, for the toxicity impact categories, the impact would be higher, the environmental profile in overall terms will not change as incineration continues to be the main factor with the greatest weight in the energy-dependent categories. As far as toxicity is concerned, it is important to be aware that when the TH pre-treatment is applied, the sludge can be considered sterilised. In this sense, the pathogens present in the sludge would be removed and its application would be safe.

Table 3.6. Characterisation results for the different scenarios evaluated in this study (including post-treatment) for 1 ton of mixed sludge. a) composting plant; b) incineration plant

I.C	**Scenario 0**		**Scenario 1**		**Scenario 2**	
	A	B	A	B	A	B
CC	40.82	160.50	42.56	162.22	29.41	167.66
OD	$8.3 \cdot 10^{-5}$	$9.1 \cdot 10^{-6}$	$9.6 \cdot 10^{-5}$	$9.2 \cdot 10^{-5}$	$1.1 \cdot 10^{-5}$	$9.3 \cdot 10^{-5}$
TA	0.48	0.77	0.48	0.76	-0.08	0.73
EP	25.77	0.20	25.77	0.20	25.77	0.20
HT	115.62	4.17	115.60	4.15	115.32	3.88
PMF	0.03	0.29	0.03	0.29	-0.05	0.27
TET	43.80	173.75	43.76	173.71	31.58	162.20
FET	6260	3.24	6260	3.24	6260	3.24
MET	4877	4.30	4877	4.30	4877	4.30
FD	-3.10	31.43	-3.28	31.50	-5.29	29.34

In order to discern the contribution of pre-treatment to the overall impact, two analysis were proposed. The main environmental categories in WWTPs are CC and EP (Rodriguez-Garcia et al., 2011). However, the EP category is more affected by the sludge disposal and, in this case, was not taken into account. For this reason, the CC category was evaluated for the different scenarios (conventional, chemical pre-treatment and TH pre-treatment). In addition, the main sub-systems that contributes to the environmental profile were evaluated in this category.

For the CC category (especially relevant in processes depending on energy production and use), S1 presents the best environmental results because chemical pre-treatment does not require much energy followed by the TH pre-treatment. The worst scenario is the conventional one (Figure 3.2) because the biogas production is lower than in the other scenarios. Although the conventional scenario has lower energy consumption due to the lack of pre-treatment unit, biogas production is lower than in the other scenarios, which results in worse environmental profile. Furthermore, considering the CC impact of the sub-systems, of each scenario (Figure 3.2), the AD unit has the worst environmental impacts due to CH_4, CO_2 and H_2S emissions (Table 3.2) while the impact of chemical pre-treatment is considered negligible in this category. However, for S2 the impact of the energy consumption for the TH pre-treatment represents 6% in this category.

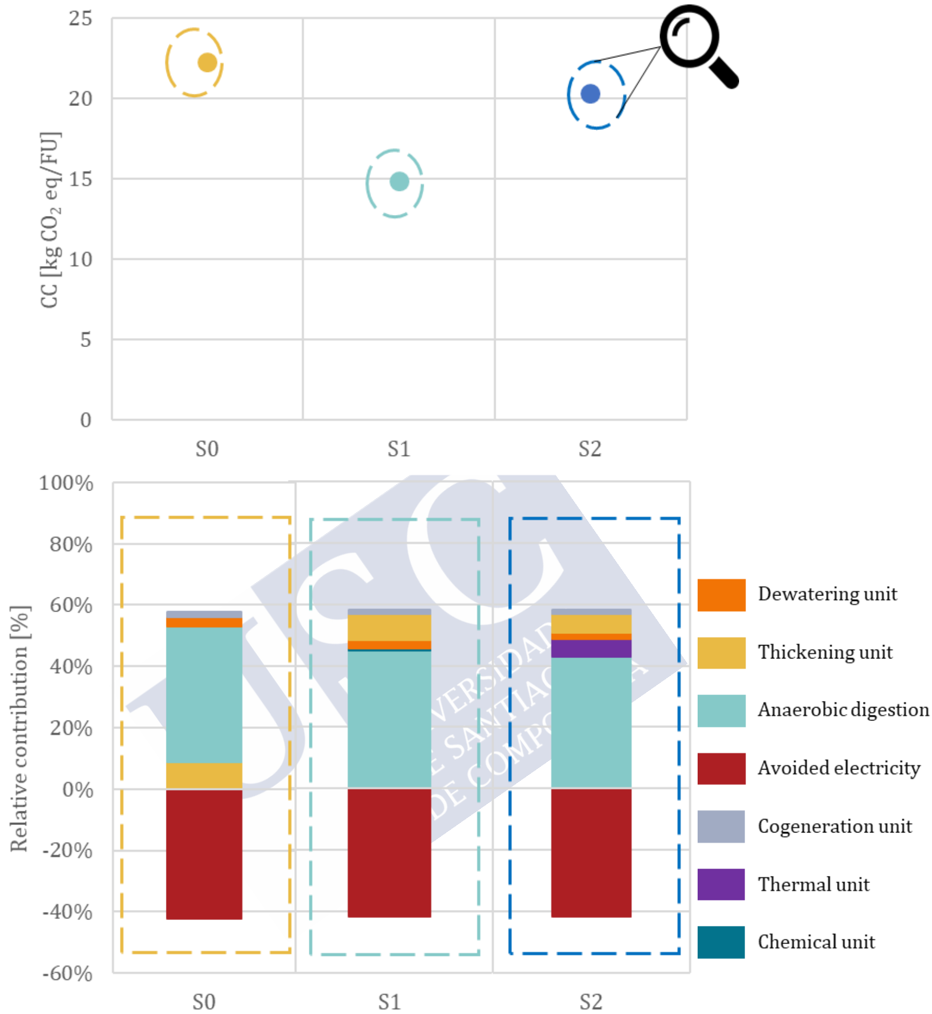

Figure 3.2. Environmental results for the climate change (CC) category for the different scenarios analysed. S0 (conventional scenario); S1 (chemical pre-treatment) and S2 (TH pre-treatment)

3.3.2. Economic evaluation of the different sludge pre-treatments

The choice of one type of pre-treatment or another should be studied from a technological, environmental and economic perspective, taking into account the size of the plant and the local conditions of each country or region. The results of the economic evaluation were classified as investment costs and operating costs (electricity consumption, chemicals, personnel and sludge management) and are presented in Table 3.7.

The best scenario in terms of operational costs is S2 without composting unit, as sludge can be applied directly to agricultural soil. In addition, biogas production is higher than in the other scenarios. This may imply more benefits related to electricity consumption, around 10% higher than chemical pre-treatment and 28% compared with conventional scenario. It is true that the electricity cost increases in S2 due to the implementation of TH plant (by 20%). Moreover, for the chemical pre-treatment, the costs related with the chemical consumption also rise in comparison with the other scenarios by around 68%, but the improvement in biogas production can offset these costs. When estimating overall costs, the high costs are distributed between sludge disposal and personnel (Table 3.7). Personnel costs should not decrease, so the best option is to reduce the final sludge disposal. In this sense, when pre-treatment (chemical or thermal) is included in the sludge line, the volume of sludge is lower than in the conventional case due to improved dewatering. Overall, the cost can be reduced by 27%.

In terms of construction costs, S2 is more expensive than the other scenarios because the TH plant is more expensive than the chemical pre-treatment (Table 3.4). In S0, the construction costs can decrease between 4% (chemical pre-treatment) and 17% (TH pre-treatment). The main responsible for the construction costs is the incineration plant, which can imply an increase of 79% of the costs in comparison with a composting plant. In this context, it is important to calculate how biogas production and the amount of sludge affect the total payback time in the different scenarios considered (Table 3.7).

As in the case of operational and construction costs, it is more difficult to amortise the investment for the scenarios when the incineration unit is integrated. Moreover, the conventional scenario presents the worst results in terms of amortisation. The best results are obtained for the TH plant because a composting unit is not necessary and biogas production is higher than in the other scenarios. In addition, S1 also has better results than the conventional scenario. That is to say, although construction costs will be higher at the beginning, it is easier to pay off these costs in the sludge lines with a sludge pre-treatment due to the lower sludge production and the increase on biogas production.

When biogas is transformed into electricity, the total operational costs can be reduced by 47% for the chemical pre-treatment and 44% for the TH process. In addition, depending on the size of the plant (medium or large), the energy demand of the network would be reduced (from 60% to 11%). This value may result in an additional benefit from 9.1 €/kW·ton to 18.6 €/kW·ton (Ma et al., 2011).

Table 3.7. Economic analysis of each scenario considered, including the post-treatment scenario. A composting plant; B incineration plant (cost are reported in Euro per 1 ton of mixed sludge)

	Scenario 0	Scenario 1	Scenario 2
Construction ^{A} (€)	76.79	79.63	96.20
Construction ^{B} (€)	149.51	152.35	168.92
Payback time ^{A} (y)	24	16	6
Payback time ^{B} (y)	47	35	31
Personnel (€)	101.80	147.61	147.61
Chemical (€)	2.88	9.14	2.88
Electricity (€)	7.17	6.40	7.63
Composting ^{A} (€)	104.4	72	28
Incineration ^{B} (€)	410.64	283.2	283.2

3.3.3. Sensitivity analysis for the different pre-treatment processes

An exhaustive analysis of the different sludge management alternatives must consider the analysis of sustainability according to the most relevant categories than affect the operation of a WWTP. Considering the relevance of the CC category, a sensitivity analysis was conducted to assess the influence of three main parameters in this category: i) biogas leaks; ii) energy demand for the sludge pre-treatment processes; and iii) energy demand for the AD unit. These parameters ranged from -20% to 20%. This is to say, five scenarios were considered: i) base case; ii) -10% and -20% decrease in energy consumption and biogas losses; and iii) 10% and 20% increase in biogas losses and energy consumption. The results for both scenarios are shown in Figure 3.3. As for the biogas leaks and the AD unit, the impacts are very similar in both units. Considering a 20% of variation in biogas leakage, a 10% variation was observed in the CC impact category (Figures 3.3a and 3b). However, biogas losses decrease in Scenario 1 (chemical pre-treatment) because biogas production is lower than Scenario 2 (TH pre-treatment).

Therefore, it is very important to ensure proper maintenance of anaerobic digesters to avoid biogas losses that can be detrimental to the environment. If the biogas losses are compared with the energy required in the pre-treatment units, the low percentage in the pre-treatment units verifies that they imply non-significant shares in the environmental impact of the plant.

a)

b)

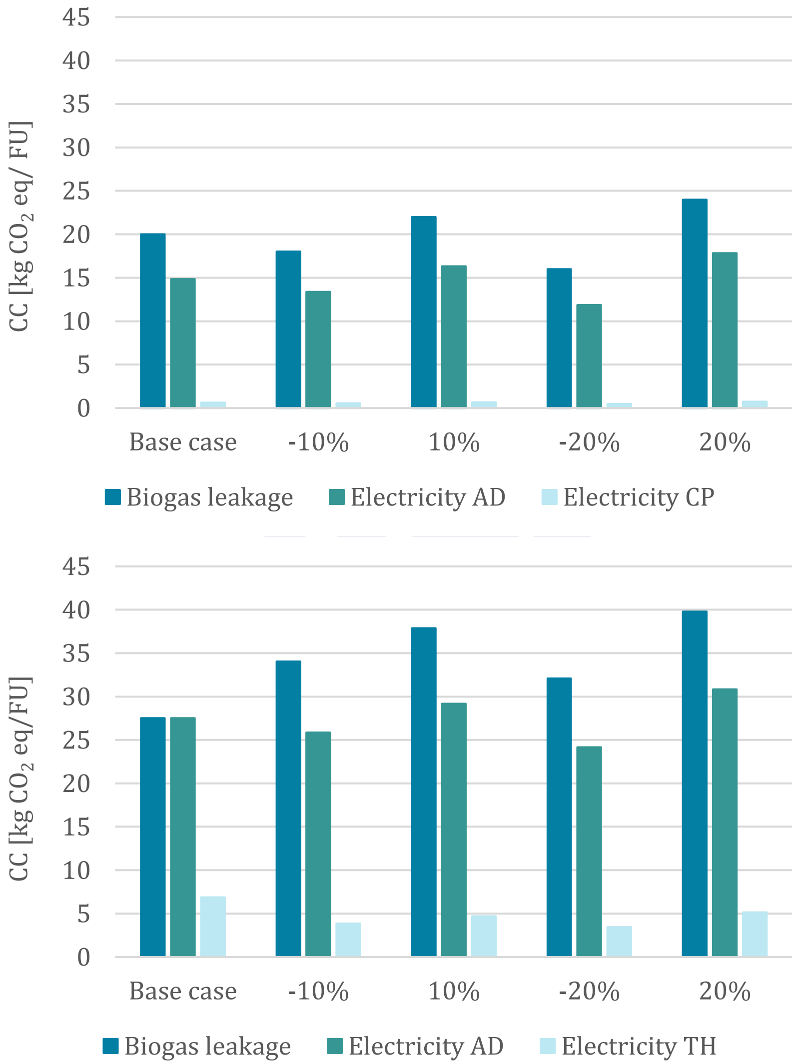

Figure 3.3. Main responsible parameters of the environmental impact for the scenarios considered: a) Scenario 1 (chemical pre-treatment); b) Scenario 2 (TH pre-treatment). FU: 1 ton of mixed sludge

3.3.4. Evaluation of the efficiency in different scenarios under evaluation

Improving biogas production is the main objective for incorporating this type of pre-treatments. The EROI is an indicator to evaluate the efficiency of WWTPs or sludge lines (Colosi et al., 2015), which was previously defined in Chapter 2.

As incineration is an energy-intensive process, in this case the post-treatment is considered to affect the efficiency of the final sludge. Incineration presents poor values in terms of efficiency, with values that range from 0.23 for the baseline and 0.24 for Scenarios 1 and 2. This means that about 80% of the electricity should be supplied by the grid. Therefore, it is very difficult to balance the environmental impacts. However, when the composting is considered followed by land application, the trend changes and estimations are indicative of energy-sufficiency.

The highest score corresponds to Scenario 2 (TH pre-treatment): 2.34 followed by Scenario 1 (chemical pre-treatment): 2.24 and conventional case (1.78). These results do not mean that the sludge line may imply environmental credits on the CC category because direct air emissions are also relevant due to the GHG emissions.

3.3.5. How to improve the efficiency of a WWTP

There are several methods that have been developed to improve dewatering and biogas production. In this chapter, two methods were analysed. However, other chemical and thermal pre-treatment alternatives have been described as follows. Wei et al., (2018) considered the application of free nitrous acid as a pre-treatment to improve biogas production and sludge dewatering by 16% and 14%, respectively. Other chemical pre-treatments of the sludge, such as Fe (II) activated persulfate or Fenton oxidation, showed an increase in biogas production of between 12% and 50% (Ra et al., 2010), which is comparable to the results reported in this study: 13%.

As the sludge pre-treatment of TH, the type of sludge plays an important role. There are two possibilities: (i) the sludge can be a mixture of primary and secondary sludge or (ii) the segregated streams of primary and secondary sludge can be treated separately. Pérez-Elvira and Ferdández-Polanco (2012) evaluated these two options and showed that methane production can rise by about 32% if the sludge was treated separately whereas the increase was only 17% for mixed sludge.

However, it is important to note that other types of pre-treatments such as ultrasound, microwaves and electrokinetic disintegration have also been proposed to improve biogas production. Riau et al., (2015) studied ultrasonic pre-treatment applied to activated sludge (secondary sludge) and concluded that methane production can be enhanced by 42%. Martín et al., (2015) also studied ultrasonication as a pre-treatment for mixed sludge and concluded that methane production can be increased by 95%. In addition, Appels et al., (2013) and Ebenezer et al., (2015) studied microwave pre-treatment of activated sludge and reported biogas improvement between 20% and 60%. Finally, the enhancements observed for electrokinetic disintegration as pre-treatment were variable: 40% for mixed sludge (Rittmann et al., 2008) and 100% for activated sludge (Salerno et al., 2009). A priori, these pre-treatments can produce more biogas than the pre-treatments considered here. However, it would be important to compute the electricity or chemical consumption associated with these alternatives in order to get a complete picture and avoid biased conclusions.

Finally, in terms of improving the efficiency of WWTP, the final disposal of sludge is very significant. This study evaluated the most common options that have gained importance in recent years (Kelessidis and Stasinakis, 2012). These alternatives were evaluated from an environmental point of view by other authors and compared with other post-treatments such as pyrolysis or wet oxidation (Dong et al., 2014; Houillon and Jolliet, 2005). However, post-treatments such as incineration, pyrolysis or oxidation showed worse environmental profile in terms of energy consumption than land application. For this reason, land application is the alternative chosen for the final disposal of the sludge as environmental credits and derived from its fertilisation potential.

3.4. CONCLUSIONS

The AD process is nowadays the most widespread process for the management of sewage sludge as it allows the production of bioenergy and the stabilisation of the sludge. Even though it is a mature and widely implemented technology, it is necessary to improve the process performance by increasing the biogas yield so this energy can be used in the plant itself. In this context, several pre-treatments have proven to have beneficial effects on biogas production: 12% (for chemically enhanced precipitation) and 30% (for TH). Additionally, the degradability of sludge and life cycle environmental impacts are significantly improved. Although construction costs increase when the sludge pre-treatment is incorporated into the sludge line, the payback time is reduced compared to the conventional configuration. This implies that amortisation of these sludge lines is more feasible compared to the conventional case. Finally, the land application of the sludge has a better environmental and economic profile than the incineration unit. However, the presence of heavy metals must be controlled and measured to avoid toxicity impacts in this sludge disposal scheme.

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CHAPTER 4: **Pursing energy self-sufficient in wastewater treatment plants: environmental and economic assessment of innovative options**

SUMMARY

Nowadays, WWTPs should no longer be considered as end-of-pipe systems but should be approached by integrating standards of technological performance but also environmental, economic and social indicators. In this framework, it is necessary to address the energy-water nexus for the selection of the most appropriate technology. Targeting increased biogas yields, the recovery of OM in the primary treatment emerges as interesting alternative. For this purpose, new technologies such as RBFs or HRAS and other not so new as UASB has been implemented as primary treatment in the water line.

Chapter 4 aims at identifying the life-cycle environmental impacts and economic costs associated to four configurations: three schemes focus on recovering OM in the primary treatment and one conventional using the LCA methodology. Despite the fact that the technological and operational complexity is noteworthy for OM-oriented process, lower environmental impacts were estimated for technologies such as UASB and HRAS. However, not all schemes based on OM recovery have environmental benefits and special attention should be paid to aspects associated with the chemical and energy consumption, as well as land occupation, which may be limiting variables to implement these technologies.

Andrea Arias*[a], Gumersindo Feijoo[a], María Teresa Moreira[a]. "Linking organic matter removal and biogas yield in the environmental profile of innovative wastewater treatment technologies". Journal of Cleaner Production 2020 Vol.247, 124292. https://doi.org/10.1016/j.jclepro.2020.124292.

[a] CRETUS institute. Department of Chemical Engineering. Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain

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4.1. INTRODUCTION

In general, current WWTPs meet environmental requirements in terms of organic matter, nitrogen and phosphorus removal. However, it is becoming increasingly evident that wastewater technologies must address more complex challenges such as the safe removal of emerging contaminants such as recalcitrant compounds and pathogens, as well as efficient operation with less resource consumption (Barbosa et al., 2016; Gu et al., 2018).

As was mentioned in Chapter 1, one of the hotspots in wastewater treatment is the energy consumption in aeration for the biological process (Gikas, 2017). In this framework, the Anammox process has several advantages such as the reduction of oxygen requirements, therefore, the energy requirement for aeration can be reduced. In addition, the extraordinarily low biomass yield of 0.12 kg VSS/ kg $N_{removed}$ means low sludge generation (Morales et al., 2015b). There are several schemes that have been developed in recent years, such as IFAS, SHARON or CANON (Malovanyy et al., 2015a; Van Dongen et al., 2001; Vázquez-Padín et al., 2010b). Although the strategy is the same for different technologies, the main difference between technologies is that PN-Anammox can be implemented in a single or two stages. However, these technologies encounter limitations in the case of streams with a large percentage of solids or a high C/N ratio (Xu et al., 2015).

In this context, it is necessary to recover OM in primary treatment. In Chapter 1, these technologies such as HRAS, RBF, CEPT or UASB (Jimenez et al., 2015; Lotti et al., 2015) were explained. The choice of one or another technology and its combination depend on several factors. For example, the energy consumption associated with UASB implies its implementation in hot climates (Bdour et al., 2009) or RBF can be combined with technologies such as HRAS and CEPT but not with the Anammox process due to the high solid content (Ruiken et al., 2012).

Sludge management is another decisive element in the operation of WWTPs according to the circular economy approach. Although the most applied methods are incineration and land application (Kelessidis and Stasinakis, 2012; Tomei et al., 2016), other options such as gasification, thermal process or supercritical water oxidation have been explored such as sludge disposal alternatives (Garrido-Baserba et al., 2015). Its use as an additive in cement production (Bertanza et al., 2016) or its conversion into granular activated carbon or bio-oil are also of interest (Kacprzak et al., 2017; Mu'Azu et al., 2019). Several authors demonstrated that the application of sludge in agriculture is a low-cost valorisation option that can provide nutrients to the soil (Pradel and Aissani, 2019; Raheem et al., 2018). However, heavy metals and other uncontrolled harmful substances may cause surface and groundwater pollution problems. This implies that their concentrations must be monitored to ensure that the discharge of heavy metals present in the sludge complies legislation requirements; otherwise, it will be necessary to implement treatment technologies to handle these streams safely (Cieślik et al., 2015).

With regard to these new systems, it is important to study if there are more environmentally friendly and economic than conventional systems. In order to assess the sustainability of these schemes, LCA showed to be a good methodology since it has been widely used for evaluating and comparing the environmental profile of different technologies or wastewater treatment schemes (Bertanza et al., 2017; Rashidi et al., 2018). In addition, environmental methodology was combined with economic impacts to look for more efficient and economic options (Garrido-Baserba et al., 2014; Piao et al., 2016).

In this framework, the main goal of Chapter 4 is to evaluate environmentally wastewater treatment schemes based on recovering OM followed by a partial nitrification-Anammox process to remove nitrogen and verify whether these schemes are more efficient than conventional system from an environmental and economic point of view.

4.2. MATERIALS AND METHODS

4.2.1. Description of the wastewater schemes and scope of the study

The main goal of this study is to evaluate four different wastewater schemes: i) three innovative based on OM recovery, and ii) one conventional from and environmental and economic perspective. These schemes are implemented in a virtual WWTP designed for 100,000 equivalent inhabitants with a flow rate of 20,700 m^3/d and COD concentration of 500 mg/L (Wan et al., 2016). The four scenarios are described in detail below.

Scenario 1 consists of the combination of UASB and IFAS in the water line. In this case, the amount of sludge generated in UASB and IFAS is lower than in the CAS. For this reason, the sludge line consists of a thickening unit, dewatering followed by a composting unit, and finally, the sludge is applied in agriculture (Figure 4.1). Scenario 2 is based on the HRAS sequence followed by an IFAS unit in the water line. The sludge line consists of a thickening unit followed by an AD process and, finally, a dewatering unit. As in the previous scenario, the sludge is treated in a composting unit and then applied in agriculture (Figure 4.2). Scenario 3 has an identical scheme for the sludge line as Scenario 2. However, the primary treatment involves RBF and CEPT coupled to the IFAS unit (Figure 4.3).

Finally, for comparative purposes, the conventional system (Scenario 4) consists of a PC and then a biological treatment based on AS process with a prolonged aeration and nitrogen removal (Figure 4.4). The sludge line is the same as in Scenarios 2 and 3.

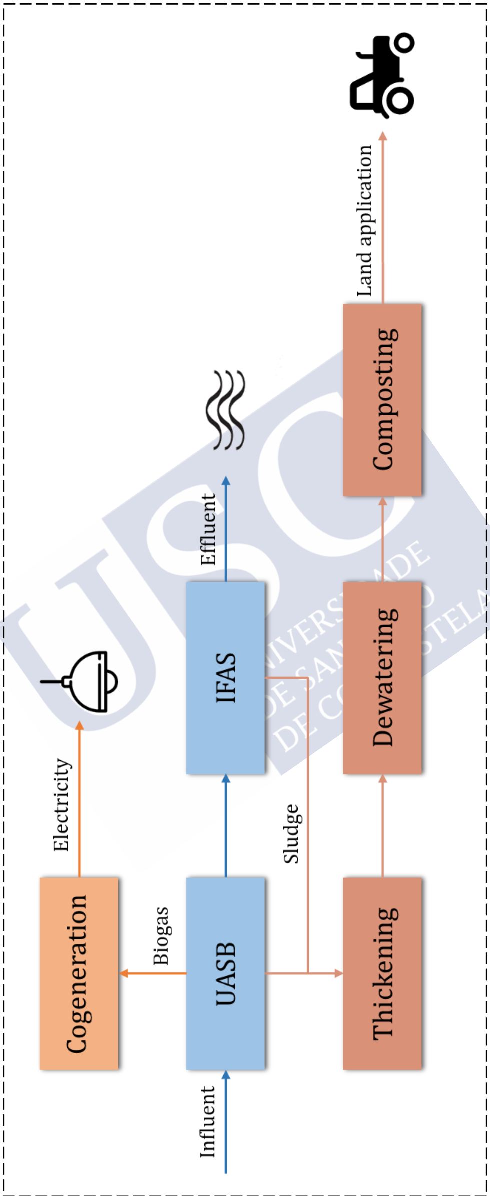


Figure 4.1. System boundaries of Scenario 1

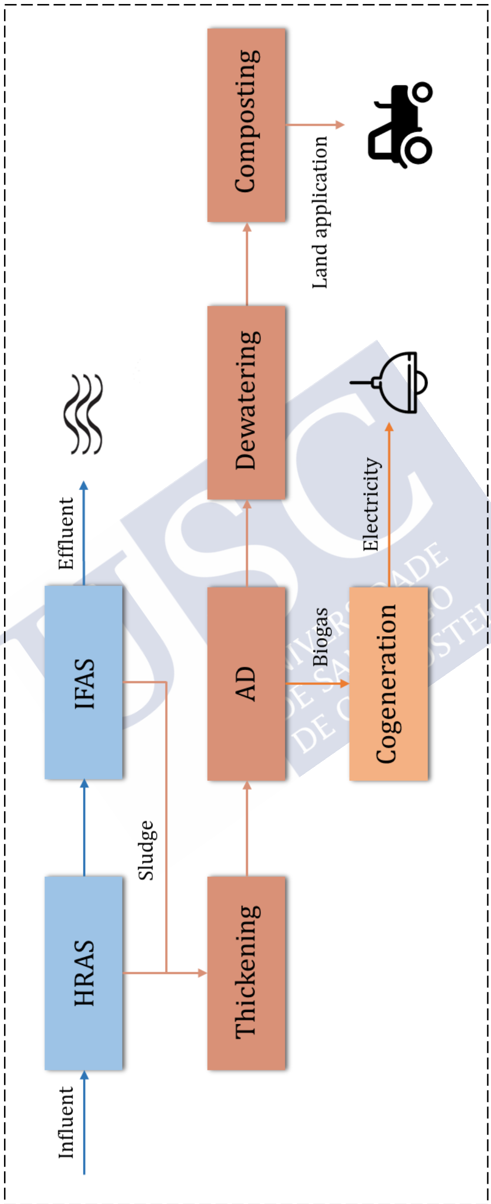


Figure 4.2. System boundaries of Scenario 2

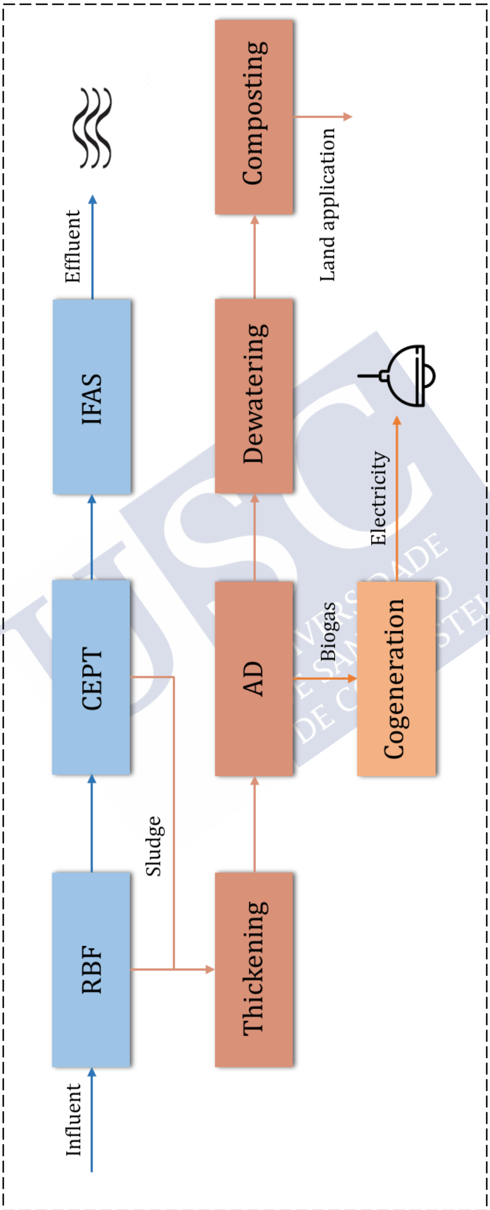


Figure 4.3. System boundaries of Scenario 3

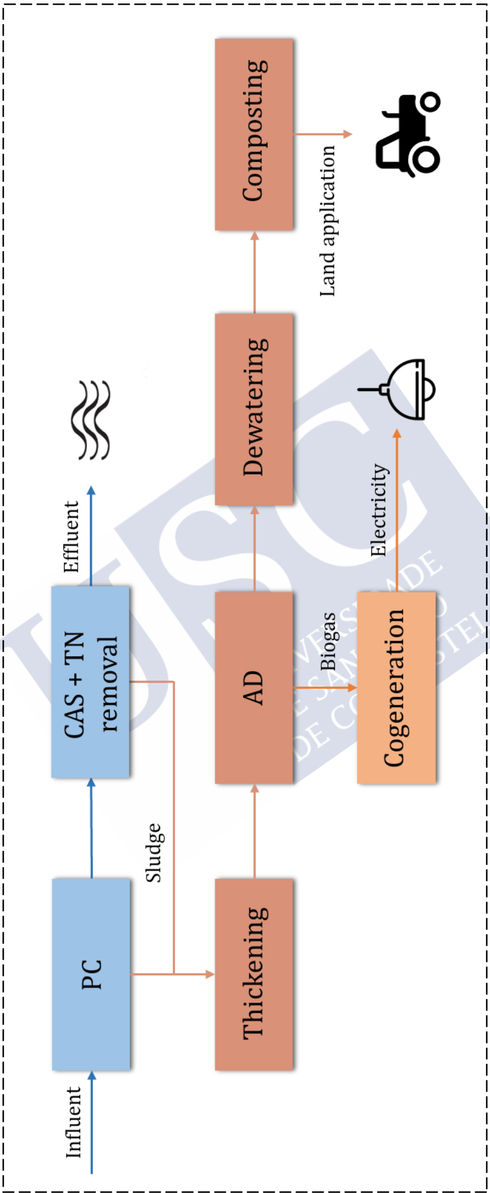


Figure 4.4. System boundaries of Scenario 4

4.2.2. Inventory data acquisition for the new wastewater configurations

In this study, only the environmental impacts associated with the operational phase were evaluated. Although sewerage impacts contribute significantly to negative effect (Petit-Boix et al., 2014), the operational phase is the main cause of the environmental impacts. System boundaries were defined as the operation of the different scenarios that are defined in the previous section (Figure 4.1 to Figure 4.4). The simplest FU selected could be 1 m^3 of treated wastewater. However, bearing in mind that the objective is to improve the efficiency of the WWTPs, 1 kWh of energy produced was selected as FU.

LCI was carried out with estimated data related with the different technologies considered in the scenarios such as sludge, wastewater characteristics or consumption of chemicals, among others. In addition, the estimated data were completed with bibliographic data associated with the air emissions and heavy metals contained in the solid digestate (Hijazi et al., 2016; Lorenzo-Toja et al., 2016b) and the Ecoinvent v3.5 database (Wernet et al., 2016). The data used to build the inventories are presented in Table 4.1 (main inputs to the system) and Table 4.2 (main outputs to the system). Moreover, several simplifications have been considered to complete the inventory information. These simplifications are presented below:

Transport: the distance for chemical and sludge distribution was selected as 25 km (Hospido et al., 2004). Moreover, trucks Euro 4 with a capacity between 16 to 32 t were selected as transport vehicles (Lorenzo-Toja et al., 2016b).

Consumption of chemicals in the sludge line: the amount of polyelectrolyte consumed in the dewatering unit was 5-8 kg polymer/ 1000 kg of dry matter (Tchobanoglous et al., 1998).

Air emissions from the compost unit: these emissions were calculated according to the type of composting plant selected. In this case, the open windrow activate ventilation process was selected as a composting process (Boldrin et al., 2009).

Electricity: Ecoinvent 3.5 database was updated to the 2018 Spanish country mix (REE, 2018). Moreover, the transmission losses associated with electricity transport were taken into account (Dones et al., 2007).

Table 4.1. Summary of the inventory data for the four scenarios considered. FU: 1 kWh of produced energy

	S1	S2	S3	S4
Inputs from technosphere				
Materials and fuel				
Influent				
COD (g)	2632	2632	2941	8333
TN (g)	101.70	99.40	112.12	305.07
TP (g)	25.14	24.57	27.71	75.41
Cr (mg)	25.92	25.37	28.21	79.92
Mn (mg)	709.95	694.92	772.59	2189
Fe (mg)	14695.95	14384.92	15992.65	45312.50
Co (mg)	7.51	7.35	8.18	23.17
Ni (mg)	50.65	49.58	55.12	156.17
Cu (mg)	1997.27	1955	2173.50	6158.25
Zn (mg)	830.89	813.31	904.21	2591.92
As (mg)	28.16	25.57	30.65	86.83
Cd (mg)	2.00	1.96	2.18	6.17
Hg (mg)	1.35	1.32	1.47	4.17
Pb (mg)	46.68	45.69	50.79	143.92
Electricity consumption				
Pre-treatment (kWh)	0.08	0.08	0.09	0.24
UASB (kWh)	0.05	-	-	-
RBF (kWh)	-	-	0.60	-
CEPT (kWh)	-	-	0.06	-
HRAS (kWh)	-	0.16	-	-
PC (kWh)	-	-	-	0.16
IFAS (kWh)	0.86	0.84	0.95	-
CAS (kWh)	-	-	-	3.40
Thickening (kWh)	$3.46 \cdot 10^{-2}$	0.03	0.04	0.10
AD (kWh)	-	0.25	0.17	0.75
Dewatering (kWh)	$2.79 \cdot 10^{-3}$	$2.72 \cdot 10^{-3}$	$3.07 \cdot 10^{-4}$	$5.67 \cdot 10^{-4}$
Composting (kWh)	$5.29 \cdot 10^{-2}$	$5.71 \cdot 10^{-2}$	$5.83 \cdot 10^{-2}$	$1.59 \cdot 10^{-1}$

Table 4.1. (cont.). Summary of the inventory data for the four scenarios considered. FU: 1 kWh of produced energy

	S1	S2	S3	S4
Chemical consumption				
CEPT				
$FeCl_3$ (kg)	-	-	0.59	-
Dewatering				
Polyelectrolyte (kg)	$1.89 \cdot 10^{-4}$	$1.85 \cdot 10^{-4}$	$2.08 \cdot 10^{-4}$	$3.50 \cdot 10^{-5}$
Transport				
Polyelectrolyte (kg·km)	$4.72 \cdot 10^{-3}$	$4.62 \cdot 10^{-3}$	$5.21 \cdot 10^{-3}$	$5.67 \cdot 10^{-4}$
$FeCl_3$ (kg)	-	-	14.89	-
Sludge (kg·km)	2.81	2.72	2.77	7.39
Land application				
Agricultural machinery (kg)	$1.13 \cdot 10^{-1}$	$1.09 \cdot 10^{-1}$	$1.11 \cdot 10^{-1}$	$2.96 \cdot 10^{-1}$

Table 4.2. Summary of the inventory data for the four scenarios considered. FU: 1 kWh of produced energy

	S1	S2	S3	S4
Outputs to the environment				
Emissions to air				
AD				
CH_4 (kg)	$4.74 \cdot 10^{-3}$	$4.64 \cdot 10^{-3}$	$4.59 \cdot 10^{-3}$	$4.88 \cdot 10^{-3}$
CO_2 (kg)	$9.37 \cdot 10^{-3}$	$9.17 \cdot 10^{-3}$	$8.84 \cdot 10^{-3}$	$9.63 \cdot 10^{-3}$
H_2S (kg)	$1.65 \cdot 10^{-4}$	$1.62 \cdot 10^{-4}$	$1.56 \cdot 10^{-4}$	$1.70 \cdot 10^{-4}$
Composting unit				
CH_4 (kg)	$5.96 \cdot 10^{-3}$	$5.51 \cdot 10^{-3}$	$2.67 \cdot 10^{-3}$	$6.38 \cdot 10^{-3}$
CO_2 (kg)	1.27	1.71	$5.69 \cdot 10^{-1}$	1.36
N_2O (kg)	$9.42 \cdot 10^{-5}$	$8.71 \cdot 10^{-5}$	$4.55 \cdot 10^{-5}$	$1.01 \cdot 10^{-4}$
NH_3 (kg)	$1.28 \cdot 10^{-2}$	$1.18 \cdot 10^{-2}$	$6.19 \cdot 10^{-3}$	$1.37 \cdot 10^{-2}$
Land application				
N_2O (kg)	$3.05 \cdot 10^{-4}$	$5.35 \cdot 10^{-5}$	$1.48 \cdot 10^{-4}$	$3.27 \cdot 10^{-4}$
NH_3 (kg)	$2.52 \cdot 10^{-4}$	$4.41 \cdot 10^{-5}$	$1.22 \cdot 10^{-4}$	$2.70 \cdot 10^{-4}$
Emissions to water				
NO_3^- (kg)	$2.04 \cdot 10^{-2}$	$3.57 \cdot 10^{-3}$	$9.84 \cdot 10^{-3}$	$2.18 \cdot 10^{-2}$
PO_4^{-3} (kg)	$2.56 \cdot 10^{-3}$	$4.47 \cdot 10^{-4}$	$1.19 \cdot 10^{-3}$	$2.56 \cdot 10^{-3}$
Emissions to soil				
COD (kg)	1.24	1.15	$5.57 \cdot 10^{-1}$	1.33
TN (kg)	$1.30 \cdot 10^{-2}$	$1.20 \cdot 10^{-2}$	$6.26 \cdot 10^{-3}$	$1.39 \cdot 10^{-2}$
TP (kg)	$8.35 \cdot 10^{-2}$	$7.71 \cdot 10^{-2}$	$3.88 \cdot 10^{-2}$	$8.37 \cdot 10^{-2}$
Cr (mg)	79.16	77.49	86.15	244.10
Fe (mg)	20118	19692	21894	62032
Cu (mg)	2138	2093	2327	6595
Zn (mg)	2674	2617	2910	8245
As (mg)	32.65	31.96	35.53	100.67
Hg (mg)	3.35	3.28	3.65	10.33
Pb (mg)	181	177.17	196.97	558.08

4.2.3. Impact assessment methodology and economic evaluation

Environmental impacts and their corresponding prices were quantified through the SimaPro 9.0 software. Two methods were selected to measure the most representative impacts of the different scenarios considered. EP was calculated with the CML 2001 method (Guinée, 2002) whereas CC, PMF, HT, OD, FD, TA, TET, MET, FET and WC were calculated with the ReCiPE Midpoint (H) v1.1 (Huijbregts et al., 2017). Moreover, these impact categories were transformed into their environmental prices. However, not all categories have their transformation into costs, for this reason, WC and EP were not included in this study (De Bruyn et al., 2018).

Operating and construction costs (OPEX + CAPEX) were selected as direct economic indicators, while environmental prices were quantified such as indirect indicators. Operational costs were related to sludge management, electricity, staff and chemical consumption. Regarding capital costs, construction, maintenance and depreciation costs were included.

4.3. ENVIRONMENTAL AND ECONOMIC RESULTS

4.3.1. Environmental and economic approach for the four studied scenarios

The environmental results are presented as a comparison between the different scenarios considered (Table 4.3). The best scenarios are Scenario 1 (UASB + IFAS configuration) followed by Scenario 2 (HRAS + IFAS configuration) because there is more electricity production than in the others. In addition, the consumption of chemicals in these wastewater units (primary technologies) is zero. However, Scenario 3 (RBF + CEPT + IFAS scheme), which is a new scheme, has a high environmental impact, even higher than in the conventional system in several categories. These environmental impacts are due to the indirect emissions associated with the chemicals production. Thus, the addition of chemicals to improve biogas production, it is not a good option from an environmental point of view. In eutrophication and toxicity categories (EP, FET and MET), which depend on the quality of effluent, Scenario 4 (conventional system) presents the worst results. The eutrophication impact is associated with the discharge of effluent into the aquatic environment as it contains N, P and COD. In addition, the integration of the IFAS unit can decrease by 13% the electricity consumption associated with aeration. This decrease in electricity improves the environmental profile because it entails lower fossil CO_2 emissions (Table 4.3).

The environmental impacts obtained were transformed into their corresponding environmental costs, which are considered as indirect costs additionally to construction and operational costs (Table 4.4). Scenario 3 presents the worst environmental prices with an increase about 52% in comparison with Scenario 4 and 80% more than Scenarios 1 and 2. The main categories that cause this negative effect are OD and TET. These categories are influenced by indirect chemical consumption emissions where Scenario 3 is worse than the other scenarios considered.

Concerning the operational costs, Scenario 1 followed by Scenario 2 are the most advisable due to electricity production is higher than in the other wastewater schemes considered. Therefore, in Scenario 1, where there is no AD unit in the sludge line, the incorporation of UASB shows that it is a good option for treating wastewater and generating electricity. The worst-scenario in terms of operating costs is Scenario 3 due to the consumption of chemicals to improve the AD process. In addition, in this scenario, two units are included to eliminate OM, so electricity consumption is higher than in the other cases. Operating costs increase by 16% compared to the conventional case and by 32% compared to the other innovative schemes.

Table 4.3. Environmental results of the different wastewater treatment schemes for the impact categories under assessment (FU: 1 kWh of energy produced)

Impact Categories	S1	S2	S3	S4
CC (kg CO_2 eq)	0.75	0.86	3.63	2.06
OD (kg CFC-11 eq)	$4.04 \cdot 10^{-7}$	$4.72 \cdot 10^{-7}$	$1.00 \cdot 10^{-5}$	$1.16 \cdot 10^{-6}$
TA (kg SO_2 eq)	0.03	0.03	0.02	0.04
EP (kg PO_4^{3-} eq)	0.07	0.07	0.09	0.20
HT (kg 1,4-DCB eq)	0.01	0.02	0.11	0.06
PMF (kg PM_{10} eq)	$4.06 \cdot 10^{-3}$	$4.08 \cdot 10^{-3}$	$5.67 \cdot 10^{-3}$	$7.27 \cdot 10^{-3}$
TET (kg 1,4-DCB eq)	0.53	0.69	5.21	2.36
FET (kg 1,4-DCB eq)	0.05	0.07	0.12	0.21
MET (kg 1,4-DCB eq)	0.06	0.07	0.15	0.23
WC (m^3)	$3.20 \cdot 10^{-3}$	$4.20 \cdot 10^{-3}$	0.03	0.01

In terms of construction costs, the most unfavourable scenario is Scenario 3 because there is an extra unit in comparison with the other scenarios followed by Scenarios 1 and 2. Although energy production is higher in these scenarios, the technology is more complex than in the conventional scenarios. For this reason, also depreciation costs are lower in the conventional scenario. The integrated analysis of environmental, operational and construction costs show that Scenario 3 is the worst-case, about 51% more than conventional and when compared with innovative schemes the difference increased up to 87% and 85% in Scenarios 1 and 2, respectively.

Table 4.4. Operational and construction costs of the different wastewater schemes considered (FU: 1 kWh of energy produced)

Costs	S1	S2	S3	S4
Operational costs				
Electricity	0.12	0.17	0.23	0.36
Chemical consumption	$3.31\cdot10^{-4}$	$3.31\cdot10^{-4}$	0.35	$6.30\cdot10^{-4}$
Sludge management	$9.87\cdot10^{-3}$	$9.77\cdot10^{-3}$	$9.84\cdot10^{-3}$	$1.64\cdot10^{-2}$
Staff	0.15	0.15	0.16	0.28
Lab. costs	$3.91\cdot10^{-3}$	$3.91\cdot10^{-3}$	$4.37\cdot10^{-3}$	$7.43\cdot10^{-3}$
Maintenance	0.09	0.09	0.10	0.18
Other costs	0.03	0.03	0.04	0.06
TOTAL OPEX (€)	0.40	0.45	0.89	0.90
Construction costs				
Pre-treatment	$3.24\cdot10^{-3}$	$3.24\cdot10^{-3}$	$3.62\cdot10^{-3}$	$6.16\cdot10^{-3}$
UASB	$7.57\cdot10^{-3}$	-	-	-
HRAS	-	$2.51\cdot10^{-3}$	-	-
CEPT	-	-	$4.68\cdot10^{-3}$	-
RBF	-	-	$2.08\cdot10^{-2}$	-
PC	-	-	-	$1.70\cdot10^{-2}$
Cogeneration unit	$1.33\cdot10^{-3}$	$1.33\cdot10^{-3}$	$1.48\cdot10^{-3}$	$2.52.10^{-3}$
IFAS	0.28	0.28	0.31	-
CAS	-	-	-	0.09
Thickening	$1.16\cdot10^{-3}$	$1.16\cdot10^{-3}$	$1.30\cdot10^{-3}$	$2.21\cdot10^{-3}$
AD unit	-	$7.57\cdot10^{-3}$	$8.46\cdot10^{-3}$	$5.66\cdot10^{-3}$
Dewatering	$5.47\cdot10^{-3}$	$5.47\cdot10^{-3}$	$6.12\cdot10^{-3}$	0.01
Composting unit	$2.98\cdot10^{-3}$	$2.98\cdot10^{-3}$	$3.33\cdot10^{-3}$	0.01
TOTAL CAPEX (€)	0.30	0.30	0.36	0.14
DEPRECIATION COSTS (€)	0.30	0.31	0.36	0.12
TOTAL INDIRECT COSTS (€)	5.45	6.95	47.81	22.63
TOTAL COSTS (€)	**6.45**	**7.99**	**49.42**	**23.79**

4.3.2. Environmental perspective for each wastewater treatment configuration

To better understand the contribution of the impact that each unit that formed the wastewater treatment scheme can create, the environmental impacts are studied individually for each scenario. As in the case before, the results are calculated on the basis of the FU (1 kWh of energy produced).

In Scenario 1 (UASB + IFAS), the main contributor to all impact categories except TA and PMF is the IFAS unit. This impact is related to the indirect emissions associated with the electricity consumption for the CC, OF or FD categories. In categories such as EP, TET, FET and MET, the impact is associated with the discharge of the effluent into the environment. The negative effect is related to the presence of heavy metals in the wastewater. Their bioaccumulation potential can affect wildlife and vegetation over time (Figure 4.5a). In TA and PMF categories, the main contributor to the impact is the composting unit. Air emissions associated with this unit are the cause of the impact on this process. The value of these emissions is presented in Table 4.3 (materials and methods section). In the CC category, the impacts are more distributed: 40% IFAS unit, 30% composting unit and 24% UASB unit. The impact of the UASB unit is related to the atmospheric emissions of CH_4, H_2S and CO_2 (Table 4.3; material and methods section). However, the UASB impact is very small and even negligible in some categories such as FET, MET or TA. Finally, the impacts of the other units such as dewatering or cogeneration can be considered non-significant (Figure 4.5a).

The results for Scenario 2 (HRAS + IFAS) are shown in Figure 4.5.b. As in the previous scenario, the main contributor to the impact in all categories except TA and PMF is the IFAS unit. As explained above, the impact is associated with the direct emissions related to the effluent discharge and the indirect emissions associated to the electricity consumption in this unit. In this case, the AD unit represents a negative effect between 3% in FET category and 30% in CC category, which is mainly attributed to biogas losses (Table 4.3; materials and methods section). The composting unit is the main contributor to the negative effect on TA and PMF (as in Scenario 1) and the effect is caused by the air emissions. As for HRAS, which is the new unit in this configuration, the impact ranges from 11% in FD to 1% in the TA category. Finally, the effect of dewatering or thickening unit can be considered negligible (Figure 4.5.b).

Figure 4.5.c shows the contribution per subsystem in Scenario 3 (RBF + CEPT + IFAS). In this scenario, the results change (Table 4.3). The CEPT unit is the main contributor to the impact in all categories except TA, CC and EP. This is due to the amount of chemicals used to achieved greater OM recovery. The impacts are associated with indirect emissions related to the production of the chemical used in this process ($FeCl_3$). In the TA category, air emissions caused in the composting unit are the main factor contributing to the negative effect (53%). In the EP category (as in the previous scenarios), the discharge of the effluent into the aquatic environment is detrimental for the environmental score. In the RBF, which is the new unit in this configuration, the main impact ranges from 13% in the OD category to 5% in the MET category. In CC, the AD unit contributes around 46% of the total impact, followed by the CEPT unit. Finally, other units such as dewatering, thickening or pre-treatment have an impact that can be considered non-significant.

Finally, Figure 4.5.d presents the results for the conventional scenario (PC + CAS with nitrogen removal). The activated sludge reactor is the worst unit in terms of environmental impact in all categories except TA and PMF. The negative effect of this unit is associated with the high electricity consumption and the direct emissions when the effluent is discharged into the environment. In the TA and PMF categories, the composting unit is the main contributor to the impact. As in the previous scenarios, the impact is related to the air emissions that occur in this process when the compost is produced. The PC unit has a negligible impact such as dewatering, cogeneration or thickening units.

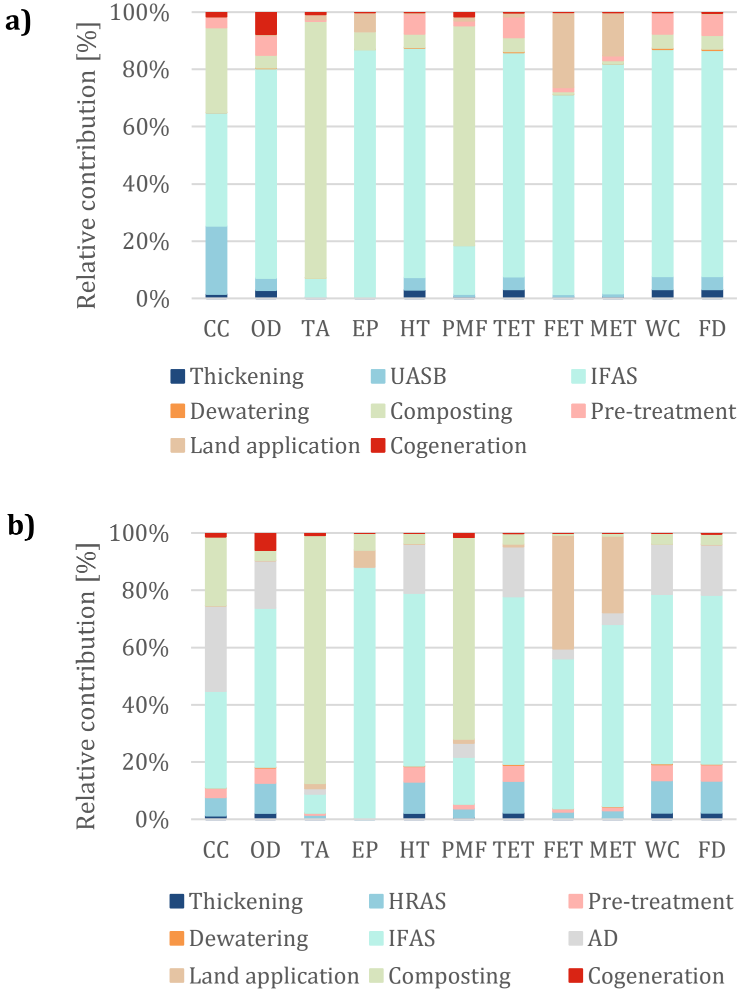
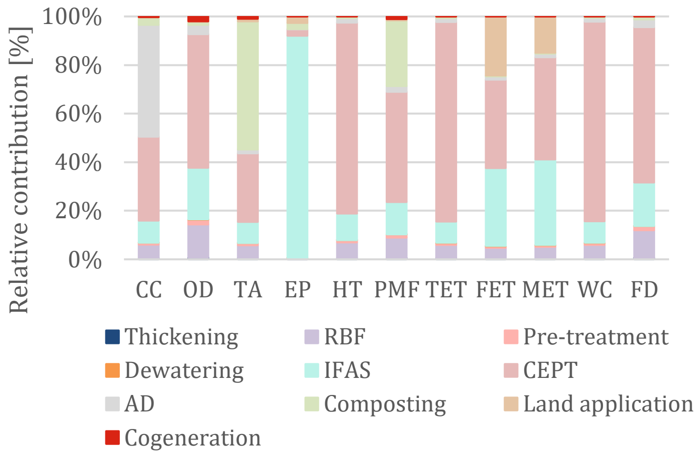


Figure 4.5. Environmental impacts for the different scenarios considered: a) Scenario 1 (UASB + IFAS); b) Scenario 2 (HRAS + IFAS); c) Scenario 3 (RBF + CEPT + IFAS); d) Scenario 4 (conventional case)

c)



d)

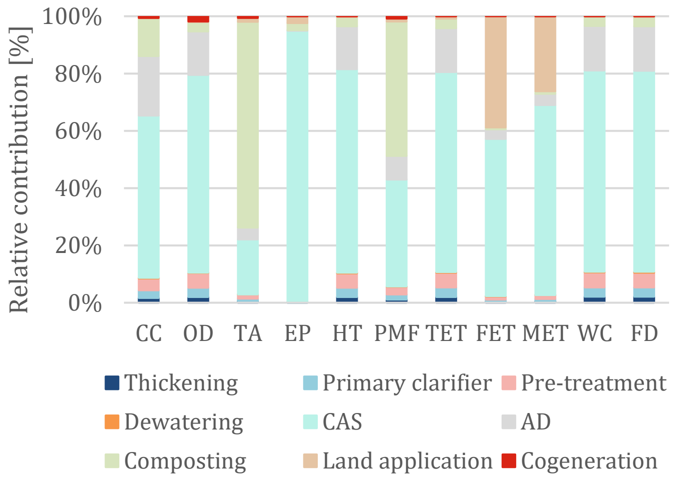


Figure 4.5 (cont.). Environmental impacts for the different scenarios considered: a) Scenario 1 (UASB + IFAS); b) Scenario 2 (HRAS + IFAS); c) Scenario 3 (RBF + CEPT + IFAS); d) Scenario 4 (conventional case)

4.4. DISCUSSION

4.4.1. Improving wastewater treatment efficiency in the WWTPs

The main objective of the new wastewater configurations is to achieve more efficient systems, mainly through the recovery OM to enhance the biogas yield. For studying the efficiency of these new configurations, the EROI indicator (as described in Chapter 2) was calculated for all the scenarios.

In this study, the best results in terms of energy are presented in Scenario 1 (UASB + IFAS) with an efficiency around 92%. This means that only 8% of the electricity will be needed to treat wastewater. Scenario 2 has an efficiency around 71% due to the incorporation of AD unit, which increases energy consumption. The worst cases in terms of energy are Scenario 3 (52%) and Scenario 4 (40%). This means that when the OM is consumed in the denitrification process, the sludge has low biodegradability and the methanisation factor is lower than in the primary sludge (Cano et al., 2015), so that the plant with a conventional scheme needs about 80% of energy from the grid. This implies more environmental and economic impacts in terms of energy.

To determine how biogas affects the environmental profile to the configurations considered in this study, a comparison (with and without biogas reuse) was conducted. The category most affected by energy consumption is CC due to the fossil CO_2 emissions (Stocker et al., 2013). Consequently, this category was selected to perform the study. In addition, 1 kWh of electricity production cannot be chosen as FU because energy will not be used in the plant. Accordingly, 1 m^3 of treated wastewater was defined as FU to make a reliable comparison. The results are shown in Figure 4.6.

Environmental impacts decrease considerably when the energy is used in the plant itself, around 15% in the conventional scenario and 31% in Scenario 1 (UASB + IFAS). Thus, new wastewater treatment configurations make sense when energy is valorised from biogas. If biogas is not reused, the conventional scenario shows environmental values very similar to Scenario 1 and even lower than Scenario 2. This is because the PC unit consumes less electricity than the HRAS unit. In this sense, it is important to highlight the importance of biogas recovery as energy to decrease the environmental profile of the WWTPs. It is evidenced that the plants are not self-sufficient in terms of energy production. Nowadays, co-digestion of waste is evaluated to develop more energy-sufficient treatments. This alternative consists of feeding the AD unit with solid waste from households, restaurants or food factories (Luostarinen et al., 2009). Depending in the type of substrate incorporated into the AD unit, biogas yield can increase by 25-60% if it is the organic fraction of municipal solid waste (MSW) or chicken waste (Mata-Alvarez et al., 2014; Wang et al., 2014). It is true that not all wastes are suitable for improving the biogas yield. For instance, the manure substrates show lower biogas production due to the high water content (Atandi and Rahman, 2012).

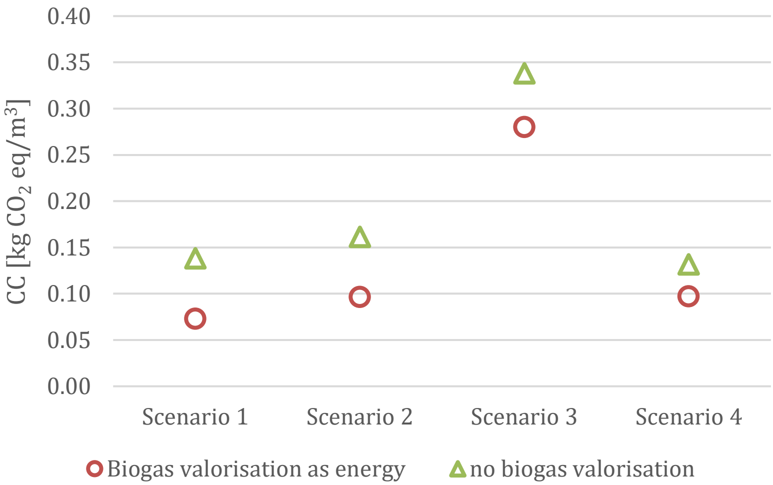


Figure 4.6. Differences between considering or not considering biogas production in the different wastewater schemes (FU: 1 m^3 of treated wastewater). Symbols: (o) biogas valorisation as energy; (Δ) no biogas valorisation.

4.4.2. How conventional and new technologies influence the effluent quality

Eutrophication is a real problem caused by the discharge of WWTP effluents into the aquatic environment. As was mentioned in Chapter 1, this problem is attributed to the amount of nutrients such as nitrogen or phosphorus that are presented in the wastewater (Lehtoranta et al., 2014). Eutrophication causes excess algae growth by decreasing the amount of oxygen present in the water and triggering mortality of aquatic species (Schindler, 2010). Therefore, it is important to improve technologies in terms of nutrient removal.

In this study, two nitrogen removal technologies were compared from an environmental perspective: i) one conventional (CAS + TN) based on the nitrification-denitrification process and ii) one more new (IFAS) based on a partial nitrification-Anammox. Both technologies are studied in terms of the effluent discharge. An indicator called Net Environmental Benefit (NEB) was defined and calculated for these technologies. This indicator analyses the difference between the potential environmental impacts (PEI) caused and avoided by WWTPs (Godin et al., 2012), and is calculated using Equation [2].

$$\mathrm{NEB} = [\mathrm{PI_{NO}} - \mathrm{PI_{TW}}] - \mathrm{PI_{SLC}} \qquad [2]$$

When PI_{NO} represents the scenario without treatment, PI_{TW} means the scenario with the treatment and, finally, PI_{SLC} corresponds to the impact caused by a WWTPs during its life cycle.

The results for IFAS and CAS with nitrogen removal are very similar, with a difference of about 1%. This is because conventional systems are highly optimised and implemented. However, IFAS is a technology that needs more research to improve its efficiency in terms of nitrogen removal. In this sense, other types of technologies were used to improve the discharge of effluents, highlighting the MBR that are characterised by a high effluent quality. If these units are compared with the IFAS and conventional technologies, MBR unit would present better results in terms of nitrogen removal (Komesli et al., 2007).

4.4.3. Economic aspects focused on energy recovery

When the energy production is improved through biogas, the operating costs can be reduced by 50% due to the benefits associated with its production (Scenarios 1 and 2). However, the costs are not reduced in all treatment schemes. In Scenario 3, although energy production is higher than in the conventional system, the operational costs increase due to the consumption of chemicals.

More than 50% of the total operational costs in a WWTP are associated with sludge management and energy consumption (Lorenzo-Toja et al., 2016b). Therefore, reducing both aspects is key to improving the economic profile of a WWTP. In this sense, the economic importance of integrating the AD unit in the WWTP has been demonstrated by several authors (Appels et al., 2008). In addition, the possibility of including sludge pretreatments based on physico-chemical or biological processes has been investigated (Neumann et al., 2016). The main objective of these pre-treatments is to improve the hydrolysis stage of the AD process since it has been identified as the main limiting stage (Braguglia et al., 2015). Pre-treatments such as chemical pre-treatment or thermal process showed an improvement in the operating costs about 18.6 €/kW·ton in a chemical pre-treatment or about 15.5 €/kW·ton in the thermal process (Ma et al., 2011). In addition, there are other methods, such as ultrasound or microwaves, which show better results in terms of biogas production. However, these methods are still at a pilot or laboratory level (Houtmeyers et al., 2014).

The factor of electricity production should be combined with energy savings. In this sense, the CAS unit consumes more energy than the partial nitrification-Anammox. In addition, there are authors that demonstrate that the effluent can be better in terms of nutrients discharge (Yang et al., 2017). A better quality in the effluent also can increase or decrease the operating costs, if the discharge area is a sensitive zone, operating costs may increase by 76% in comparison with the conventional technologies (Rodriguez-Garcia et al., 2011).

4.4.4. Sensitivity analysis of the functional unit (FU)

In LCA methodology, a crucial step is the definition of the FU, since this decision influences the inventory data and the results. In this case, maximising electricity production is a key factor in our system. However, the main function of WWTPs is to treat wastewater. In this sense, it is important to evaluate the influence of the selection of the FU on the outcomes of the analysis. Therefore, two FU were studied and compared (1 kWh of energy produced and 1 m^3 of treated wastewater). The category studied was the CC category, because this category is the most influenced by possible changes in energy consumption or production (Zouboulis and Tolkou, 2015).

Figure 4.7 shows the results of the different scenarios for both FU. These results are very similar and range from 5% in Scenario 3 to 1% in the other scenarios; therefore, the difference is not significant. Thus, the choice of another FU does not change the results and Scenario 1 would be the best from an environmental perspective.

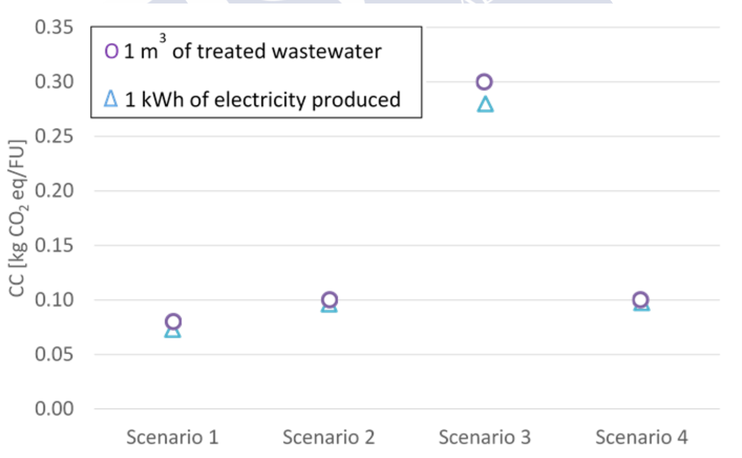


Figure 4.7. Comparison between two different functional units (1 m^3 of treated wastewater and 1 kWh of energy produced) for the climate change category. Symbols: o represents 1 m^3 of treated wastewater; Δ represents 1 kWh of energy produced)

4.5. CONCLUSIONS

In this study, a new treatment strategy focused on OM recovery was evaluated from an environmental and economic perspective. Three schemes based on this strategy: (i) UASB + IFAS; (ii) HRAS + IFAS, and (iii) RBF + CEPT + IFAS) were compared with a conventional treatment scenario (PC + CAS). The UASB and HRAS followed by an IFAS unit had a better environmental profile than the conventional technology. Moreover, the energy consumption in aeration can decrease by 13% when IFAS is integrated. However, not all schemes based on this strategy showed a better environmental and economic profile. Technologies that require chemical achieved worse results than the conventional system in the ecotoxicity and human health categories. In addition, costs can increase by 51% compared to the conventional plant. When a technology is implemented, validation is needed not only form a technology point of view but also from an environmental and economic perspective. In this way, these elements that are considered end-of-pipe systems for waste treatment can be adapted to the circular economy and become more sustainable.

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CHAPTER 5: **Mapping the environmental and economic impacts of innovative technologies for enhancement of biogas production and sludge management in wastewater systems**

SUMMARY

In recent years, new wastewater treatment plans have been proposed to tackle more complex challenges. To address these new configurations, it is necessary to use tools to model, optimise and select the most appropriate plant layout for each scenario. It is not possible to embark on the construction of new facilities unless the previous technical, economic and environmental feasibility studies have been rigorously considered.

It is well known that the elements that penalise the wastewater treatment are: i) energy consumption and ii) sludge management. Based on these premises, the main objective of Chapter 5 is to evaluate which treatment configuration ensures the efficient water-energy nexus and the reduction of the operational costs linked to the wastewater scheme. For this purpose, the treatment configuration of two real plants of different size was modified to include some novel concepts such as physical-chemical and biological processes for the recovery of organic matter OM in the primary treatment, as well as the implementation of a partial nitrification-anammox process in the secondary treatment. According to the modelling results that integrate the environmental and economic indicators using the LCA methodology, the schemes based on HRAS or RBF + chemical addition followed a partial nitrification-Anammox led to the best environmental and economic results. These results are attributed to increased biogas production and reduced electricity demand from the grid. Furthermore, these schemes proved to be cost-effective and environmental-friendly for both plant sizes and configurations.

Andrea Arias[*a], Chitta Rajan Behera[b], Gumersindo Feijoo[a], Gürkan Sin[b], María Teresa Moreira[a]. "Unravelling the environmental and economic impacts of innovative technologies for the enhancement of biogas production and sludge management in wastewater systems". Journal of Environmental Management. 2020, Vol. 227, 110965. https://doi.org/10.1016/j.jenvman.2020.110965. [a]CRETUS institute. Department of Chemical Engineering. Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain. [b] Process and Systems Engineering Center (PROSYS), Department of Chemical and Biochemical Engineering, Technical University of Denmark, Building 229, 2800 Kgs. Lyngby, Denmark

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5.1. INTRODUCTION

The main objective of WWTPs was to discharge an effluent into the environment and to avoid problems related to eutrophication or ecotoxicity (Mo and Zhang, 2013). Conventional nitrification-denitrification processes have been applied to remove COD and nutrients. However, environmental regulations are becoming increasingly strict and include other aspects such as gas emissions or efficient sludge management, which are generally considered to be an environmental and economic burden. In conventional processes, electricity demand can vary between 0.3 kWh/m^3 and 0.6 kWh/m^3 (Wan et al., 2016). Moreover, in some cases, it is necessary to add an external source of OM to successfully complete the denitrification process, which increases operational costs (Morales et al., 2015b). In this context, since the discovery of the autotrophic nitrogen removal process (Anammox), its incorporation in WWTPs has been sought in order to develop more efficient wastewater treatment technologies. This new system does not depend on an external source of OM and can work at considerably lower temperatures, between 10-20 °C (Tao et al., 2014). Moreover, as presented in Chapter 4, several full-scale Anammox process facilities have been developed to treat the digestate of the AD process which is characterised by a low carbon-to-nitrogen ratio (Lackner et al., 2014).

However, the incorporation of the Anammox process has limitations and cannot operate with a high percentage of solids or COD (Lackner et al., 2014). It is for this reason that the highest OM should be removed in the primary treatment. In this sense, several technologies have been developed in order to recover OM in the first stage of treatment, such as RBF or HRAS (Jimenez et al., 2015; Ruiken et al., 2013) and other more conventional have implemented as UASB (Kujawa-Roeleveld et al., 2006). However, the information on their integration in real plants is limited and so far, based on the information available on performance indicators, it is not possible to evaluate an integrated treatment scheme from a techno-economic and environmental point of view.

In this context, a plant-wide modelling and simulation study of the different innovative configurations may provide additional insight on the compatibility of the above discussed technologies. Several plant-wide studies have been conducted to evaluate treatment schemes, technology retrofitting or control strategies (Flores-Alsina et al., 2008; Gernaey et al., 2014). In addition, innovative plant modelling studies have been conducted (Behera et al., 2018). Many of these studies focused only on the techno-economic feasibility lacking environmental aspects of such technologies, which is the primary topic of this chapter.

Today, thanks to advances in computation software, plant-wide simulation can be performed. To evaluate the environmental profile of these new configurations, LCA proves to be a good alternative whose effectiveness has been demonstrated in several wastewater configurations and technologies (Foley et al., 2010; Schaubroeck et al., 2015). Therefore, the main objective of Chapter 5 is to combine the approach of OM recovery to maximise biogas production and a partial nitrification-Anammox to remove nitrogen in the treated effluent as the scenario to be implemented in two real WWTPs of medium and large sizes in different European countries (Spain and Denmark). With the outcomes of the modelling stage, an environmental and economic analysis was conducted to assess whether the wastewater treatment schemes based on this perspective are better than conventional wastewater treatment strategies.

5.2. MATERIALS AND METHODS

5.2.1. Methodology on simulation and environmental assessment

The IWA task group has developed new models and tools for the evaluation of WWTPs such as Benchmark Simulation Model No.2 (BSM2), which is being widely used as a framework for plant-wide analysis (Gernaey et al., 2014). This study addresses several models developed from BSM2 and its interfaces. Table 5.1 summarises the modelling approach used for both conventional and emerging technologies. As part of the simulation strategy, the plant-wide model is initiated using a sequential approach to avoid model convergence problems (Behera et al., 2018). A closed-loop steady-state simulation is then performed using a rigid differential solver such as the *ode15s* in MATLAB-Simulink software

(2016a). In addition, the different plantwide layouts are represented in Figures 5.1. to 5.3. Finally, for the calculation of environmental impacts, the LCA methodology was applied according to the standardised method defined by ISO 14040 (ISO 14040, 2006).

Table 5.1. Modelling approach used for conventional and emerging technologies

Technology Name	Mechanism	Modelling approach reference
Primary clarifier (PC)	Gravitational settling	(Gernaey et al., 2014; Otterpohl et al., 1994)
Enhanced rotating belt filter (ERBF)	Coagulation, flocculation, sieving and cake filtration	(Behera et al., 2018; Boiocchi et al., 2019)
High rate activated sludge (HRAS)	Bio-sorption	(Smitshuijzen et al., 2016)
Modified Ludzack-Ettinger (MLE)	COD oxidation, nitrification, and pre-denitrification	(Guo and Vanrolleghem, 2014; Henze et al., 2000)
Secondary clarifier (SC)	Gravitational settling	(Takács et al., 1991)
Integrated fixed film activated sludge (IFAS)	Partial nitrification/anammox, COD oxidation, conventional denitrification	(Behera et al., 2019; Vangsgaard et al., 2013)
Thickener and Dewatering	Gravitational settling	(Gernaey et al., 2014)
Anaerobic digester (AD)	Hydrolysis, acidogenesis, acetogenesis, methanogenesis	(Batstone et al., 2002)

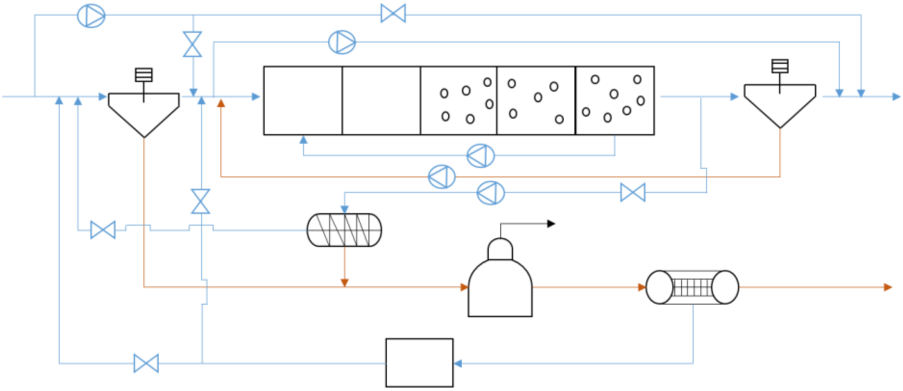

Figure 5.1. Conventional configuration for modelling simulation

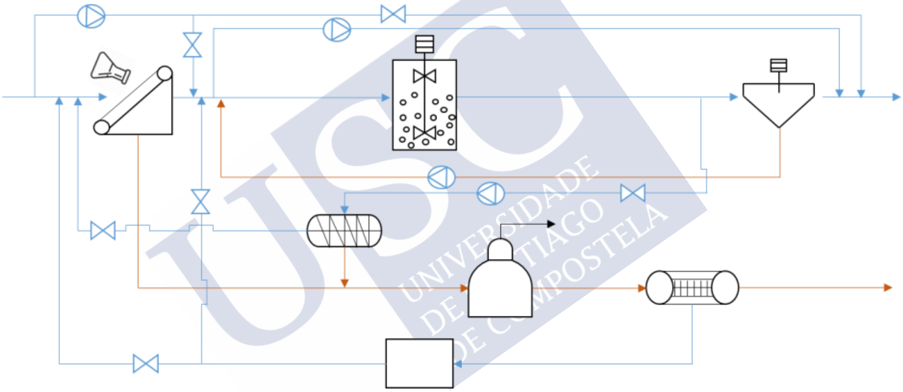

Figure 5.2. Enhanced rotating belt filter (ERBF) configuration for modelling simulation

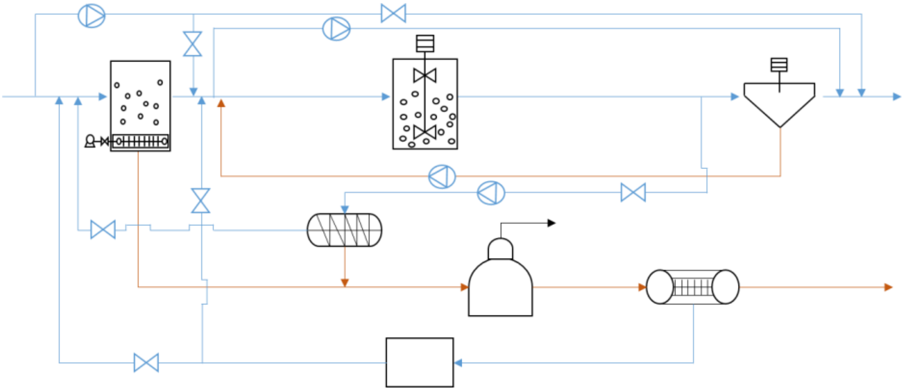

Figure 5.3. HRAS configuration for modelling simulation

5.2.2. Goal and scope of the two wastewater schemes considered

In this Chapter, two real WWTPs (one medium and one large) were considered as the basic configuration for applying the technologies proposed above. It is important to know how to improve the biogas production and the efficiency of the wastewater schemes depending on the technology considered. One wastewater treatment plant is located in Denmark (Avedøre) and is designed for 265,000 equivalent inhabitants with a flow of about 72,000 m^3/d. The second plant is located in Valladolid (Spain). The flow is 213,000 m^3/d with a population of 1,000,000 equivalent inhabitants. All the flows of energy and materials, as well as the emissions associated with the operation of the WWTPs, were considered and quantified in detail.

In this case, the most common FU will be 1 m^3 of treated wastewater (Schaubroeck et al., 2015). However, the main objective of implementing these innovative technologies is the electricity production. In this context, 1 kWh of energy produced in a CHP was defined as FU.

5.2.3. System boundaries for the wastewater treatment configurations

A gate-to-gate perspective was chosen as system boundaries and are presented in Figure 5.4. Only the operational phase of each wastewater scheme was selected. The average reason for choosing only the operational phase is that in a WWTP the main environmental impacts occur in this phase (Lundin et al., 2000).

The base scenario (Scenario 0) is the conventional scheme of a WWTP. Both WWTPs consist of a pre-treatment followed by a PC and an AS process with nitrogen removal in the water line. The sludge line consists of a thickener, an AD unit and a dewatering system. In addition, biogas is transformed into electricity in a CHP unit. The main difference between the two plants is the sludge disposal. In the case of the Valladolid plant, the sludge is applied on agricultural land, while at Avedøre plant, the sludge is incinerated. Therefore, these alternatives are considered for the environmental profile. These conventional technologies (PC and AS units) are replaced by innovative technologies. Thus, two scenarios were studied and compared with the base case:

Scenario 1 consists of the combination of ERBF and IFAS unit in the water line. Scenario 2 is based on HRAS followed by IFAS unit in the water line. The sludge line is the same for all scenarios and does not change. As mentioned above, the only change is the final disposal of the sludge (incineration or land application).

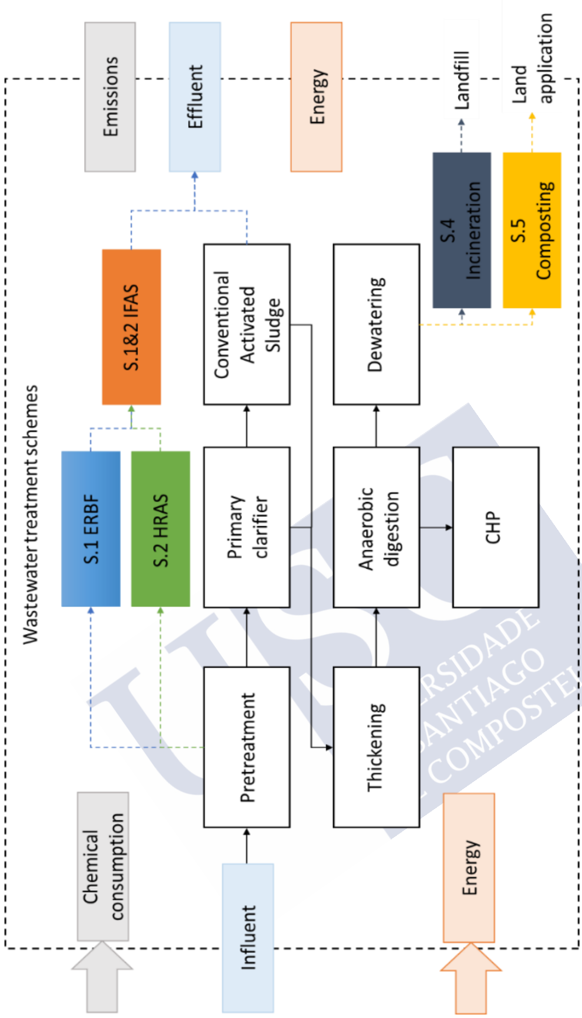

Figure 5.4. System boundaries of the different wastewater schemes considered

5.2.4. Inventory data acquisition through the simulation process

Data inventory collection is the most time-consuming stage and is linked to the quality of the results obtained in the environmental analysis (Finnveden, 2000). In this case, the data associated with COD, nitrogen, phosphorus or heavy metals were obtained from the available information reported by the managers of Valladolid and Avedøre facilities (Table 5.2). These influent parameters were implemented in the model to obtain data related to methane production, energy consumption or effluent parameters. In addition, the characteristics of each electricity country mix were completed with the Ecoinvent v3.5 database (Weidema et al., 2013). Moreover, several simplifications were considered, especially those of foreground processes, as detailed below:

Two different electricity country mix were selected due to the different location of the WWTPs. Spanish and Denmark country mixes were updated and the medium-voltage electricity used in WWTPs was modelled, including the losses in transport (Dones et al., 2007). Euro 4 trucks with a capacity between 12 and 32 t were selected to transport chemicals and sludge. In addition, an average distance of 50 km was considered for the transport of chemicals and sludge. For the application of sludge to the soil, emissions to air (N_2O and NO_3) and water (NO_3^- and PO_4^{-3}) were calculated and taken into account in the final environmental profile (Bruun et al., 2006). The inventories are shown in Tables 5.3 and 5.4 per FU considered (1 kWh of energy produced).

Table 5.2. Main inputs parameters for both plant sizes considered in this study

Influent	Large plant	Medium plant	Units	Reference
COD	362.78	220.89	g/m^3	(Aguas de Valladolid, 2017; BIOFOS, 2017)
TSS	207.33	183.94	g/m^3	(Aguas de Valladolid, 2017; BIOFOS, 2017)
TN	31.76	16.99	g/m^3	(Aguas de Valladolid, 2017; BIOFOS, 2017)
TP	5.44	2.48	g/m^3	(Aguas de Valladolid, 2017; BIOFOS, 2017)
Cr	163.73	189.08	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Ni	319.94	288.12	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Cu	12616.51	18676.44	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Zn	5248.64	696.89	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
As	177.90	50.16	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Cd	12.63	7.71	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Hg	8.54	3.85	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)
Pb	294.84	356.29	mg/m^3	(Lorenzo-Toja et al., 2016; BIOFOS, 2017)

Table 5.3a. Main inputs of the different wastewater schemes considered in the Avedøre plant (FU:1 kWh of energy produced)

	Scenario 0	Scenario 1	Scenario 2
Inputs from the technosphere			
Materials and fuel			
Influent			
COD (kg)	2.93	2.50	1.83
TN (kg)	0.30	0.25	0.18
TP (kg)	$3.28 \cdot 10^{-3}$	$2.80 \cdot 10^{-3}$	$2.05 \cdot 10^{-3}$
Electricity consumption			
Pretreatment (kWh)	0.09	0.08	0.06
PC (kWh)	0.03	–	–
RBF (kWh)	–	0.05	–
HRAS (kWh)	–	–	0.01
Activated sludge (kWh)	0.94	–	–
IFAS (kWh)	–	0.88	0.02
Thickening (kWh)	0.01	$9.82 \cdot 10^{-3}$	$7.24 \cdot 10^{-3}$
AD (kWh)	0.09	0.05	0.03
Dewatering (kWh)	0.17	0.14	0.10
Incineration (kWh)	0.07	0.06	0.05
Chemical consumption			
Primary treatment			
Polyelectrolyte (kg)	–	0.02	–
Dewatering			
Polyelectrolyte (kg)	0.09	0.07	0.05
Transport			
Polyelectrolyte (kg·km)	2.18	2.34	1.35
Ashes (kg·km)	17.09	14.30	10.55
Amount of ashes (kg)	0.68	0.57	0.42

Table 5.3b. Main outputs of the different wastewater schemes considered in the Avedøre plant (FU:1 kWh of energy produced)

	Scenario 0	**Scenario 1**	**Scenario 2**
Outputs to the environment			
Emissions to air			
AD			
CH_4 (kg)	$1.63·10^{-2}$	$1.02·10^{-2}$	$1.02·10^{-2}$
CO_2 (kg)	$3.28·10^{-2}$	$2.05·10^{-2}$	$2.05·10^{-2}$
H_2S (kg)	$5.64·10^{-4}$	$3.52·10^{-4}$	$3.52·10^{-4}$
Emissions to water			
Effluent			
COD (kg)	0.27	0.24	0.15
TN (kg)	0.04	0.03	0.03
TP (g)	3.69	3.69	3.69
Pb (mg)	3.28	2.74	2.02
Cd (mg)	0.23	0.20	0.14
Cu (mg)	9.60	8.03	5.92
Cr (mg)	9.12	7.64	5.63
Hg (mg)	0.47	0.39	0.29
As (mg)	3.75	3.13	2.31
Ni (mg)	32.77	27.41	30.21
Zn (mg)	139.03	116.32	85.79

Table 5.4a. Main inputs to the different wastewater schemes considered in the Valladolid plant (FU:1 kWh of energy produced)

	Scenario 0	Scenario 1	Scenario 2
Inputs from the technosphere			
Materials and fuel			
Influent			
COD (kg)	10.37	7.78	3.88
TN (kg)	0.64	0.41	0.22
TP (kg)	0.10	0.06	0.03
Electricity consumption			
Pretreatment (kWh)	0.31	0.20	0.11
PC (kWh)	0.09	–	–
RBF (kWh)	–	0.13	–
HRAS (kWh)	–	–	0.01
Activated sludge (kWh)	3.52	–	–
IFAS (KWh)	–	2.03	1.02
Thickening (kWh)	0.04	0.02	0.01
AD (kWh)	0.29	0.09	0.04
Dewatering (kWh)	0.55	0.03	0.18
Composting (kWh)	0.20	0.13	0.07
Chemical consumption			
Primary treatment			
Polyelectrolyte (kg)	–	0.05	–
Dewatering			
Polyelectrolyte (kg)	0.85	0.54	0.28
Transport			
Polyelectrolyte (kg·km)	21.16	14.72	7.11
Sludge (kg·km)	72.27	50.77	24.26
Land application			
Agricultural machinery (kg)	2.89	2.03	0.97

Table 5.4b. Main outputs to the different wastewater schemes considered in the Valladolid plant (FU: 1 kWh of energy produced)

	Scenario 0	Scenario 1	Scenario 2
Outputs to the environment			
Emissions to air			
AD			
CH_4 (kg)	$1.67 \cdot 10^{-2}$	$1.67 \cdot 10^{-2}$	$1.67 \cdot 10^{-2}$
CO_2 (kg)	$3.30 \cdot 10^{-2}$	$3.30 \cdot 10^{-2}$	$3.30 \cdot 10^{-2}$
H_2S (kg)	$5.83 \cdot 10^{-4}$	$5.83 \cdot 10^{-4}$	$5.83 \cdot 10^{-4}$
Composting			
CH_4 (kg)	$1.84 \cdot 10^{-2}$	$1.29 \cdot 10^{-2}$	$6.17 \cdot 10^{-3}$
CO_2 (kg)	1.69	2.75	1.31
N_2O (kg)	$9.82 \cdot 10^{-2}$	$1.85 \cdot 10^{-4}$	$9.73 \cdot 10^{-5}$
NH_3 (kg)	$5.06 \cdot 10^{-2}$	$5.61 \cdot 10^{-2}$	$2.95 \cdot 10^{-2}$
Land application			
N_2O (kg)	$1.47 \cdot 10^{-3}$	$1.09 \cdot 10^{-3}$	$5.74 \cdot 10^{-4}$
NH_3 (kg)	$1.21 \cdot 10^{-3}$	$9.00 \cdot 10^{-4}$	$4.73 \cdot 10^{-4}$
Emissions to water			
Effluent			
COD (kg)	0.74	0.45	0.26
TN (kg)	0.19	0.03	0.02
TP (g)	39.70	39.70	39.70
Pb (mg)	11.69	7.42	3.92
Cd (mg)	1.30	0.83	0.44
Cu (mg)	127.26	80.82	42.73
Cr (mg)	6.65	4.22	2.23
Hg (mg)	8.36	5.31	2.80
As (mg)	63.98	40.63	21.50
Ni (mg)	39.29	24.96	13.20
Zn (mg)	381.98	242.58	128.29
Land application			
NO_3^- (kg)	$9.82 \cdot 10^{-2}$	$1.85 \cdot 10^{-4}$	$5.74 \cdot 10^{-4}$
PO_4^{-3} (kg)	$5.06 \cdot 10^{-2}$	$5.61 \cdot 10^{-2}$	$4.73 \cdot 10^{-4}$

Table 5.4b.(cont.) Main outputs to the different wastewater schemes considered in the Valladolid plant (FU:1 kWh of energy produced)

	Scenario 0	Scenario 1	Scenario 2
Outputs to the environment			
Emissions to soil			
TP (g)	1.65	1.05	0.55
Pb (mg)	122.14	77.57	41.01
Cd (mg)	1.27	0.80	0.43
Cu (mg)	382.96	242.20	128.61
Cr (mg)	61.36	38.97	20.61
Hg (mg)	1.20	0.76	0.40
As (mg)	25.58	16.24	8.59
Ni (mg)	46.17	29.32	15.50
Zn (mg)	826.70	524.99	277.63

5.2.5. Environmental and economic indicators selected for the case studies

Two methods were selected to calculate the most representative impacts of a WWTP. EP was calculated with the CML 2001 method (Guinée, 2002) whereas CC, OD, TA, PMF, HT, MET, TET, FET, WC and FD were calculated with the ReCiPe Midpoint (H) method (Huijbregts et al., 2017). The SimaPro 9.0 software was used to implement the inventories.

As for the economic indicators, only the costs associated to the operational phase was considered. These economic indicators are related to the sludge management, electricity and chemical consumption. Biogas is considered as a benefit; thus, it is important computed for the calculation of revenues (Mills et al., 2014). As in the environmental analysis, construction costs were not considered because they represent a minor contribution to the total costs (Termes-Rifé et al., 2013).

5.3. RESULTS

5.3.1. Life cycle environmental profile for each new wastewater treatment scheme

Firstly, the environmental results are presented according to the size of the plant, observing the contribution that each subsystem makes to the total environmental profile. Figure 5.5 shows the contribution per subsystem for the Avedøre case. In the conventional case (Scenario 0), the main contributor to the impact in all environmental categories except EP, TET and HT is the CAS with nitrogen removal unit followed by dewatering. The negative effect is associated with high electricity consumption and direct emissions from the treated effluent as it presents residual concentrations of nitrogen, phosphorus and heavy metals. The discharge of these pollutants in large quantities can cause mortality of aquatic species. In dewatering unit, emissions are associated with the consumption of electricity and polyelectrolyte that is used as an additive to improve the dewatering of the sludge. Depending on the impact category, the negative effect related to polyelectrolyte can vary from 83% in the TET category to 52% in the WC category.

The incineration unit is the unit causing the major impact in the EP and HT categories due to the disposal of ashes that may contain hazardous contaminants such as heavy metals. Other units such as PC or thickener have a negligible impact (Figure 5.5a). In Scenario 1 (including ERBF technology), the main impact is distributed as in Scenario 0 and the reasons for the negative contribution are the same. However, the incorporation of this type of treatment cannot be considered irrelevant and has a contribution of between 2% in the FET or MET categories and 11% in the TET category. This negative effect is related to indirect emissions from chemical production (Figure 5.5b).

Finally, for Scenario 2 (integration of HRAS technology), the environmental profile changes. In this case, the dewatering unit is the main responsible of the impacts in all categories except FET and MET. In these categories, the IFAS unit is the contributor to the impact due to direct emissions associated with the impact of nutrients present in the effluent discharged to the environment. Finally, as in Scenario 0, the impact caused by HRAS can be considered negligible in all categories.

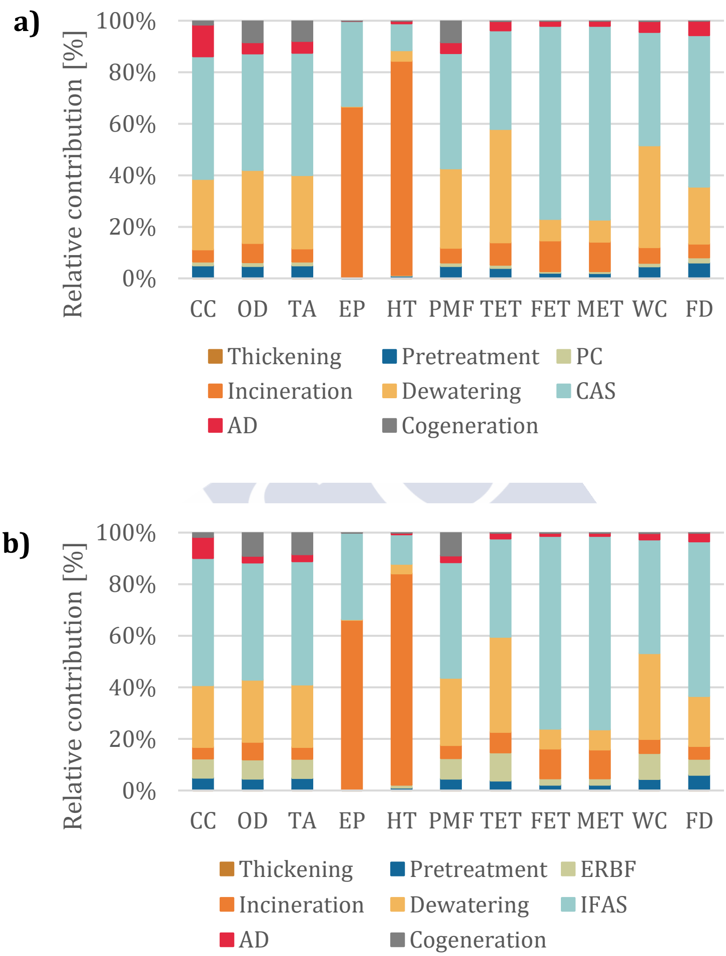

Figure 5.5. Environmental impacts for each scenario considered in Avedøre WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS)

c)

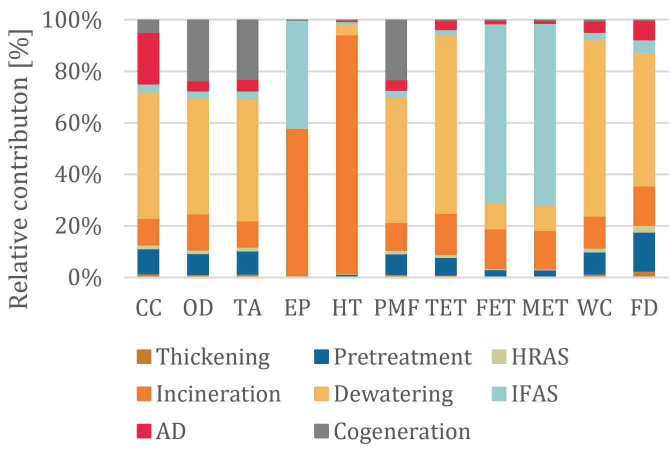

Figure 5.5 (cont.). Environmental impacts for each scenario considered in Avedøre WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS)

Figure 5.6 shows the environmental profile of the large plant for each scenario considered. In this case, the main difference is the incorporation of a composting unit followed by land application. This final disposal has a negative effect on the PMF and TA categories due to air emissions (Table 5.3a). In addition, as in the medium plant, the main factor contributing to the impact is the CAS unit, as electricity consumption in energy-dependent categories such as CC, WC, OD, HT and FD. In categories that do not depend on energy consumption (MET, FET, EP and TET), the negative effect is caused by the discharge of the effluent which contains hazardous substances such as heavy metals that may be harmful to aquatic or terrestrial species. In other units such as cogeneration, PC or thickening, the impacts are very small (Figure 5.6a). For Scenario 1 (ERBF), the negative effects are similar to Scenario 0 (conventional scenario) (Figure 5.6b). The main difference is that the impact on this new unit cannot be considered minor and ranges from 3% in the FET or MET categories to 11% in FD. Finally, for Scenario 3 (HRAS), as in a medium plant, the impact of HRAS can be considered negligible. Therefore, in this case, IFAS consumes less energy due to the recovery of OM. This implies that the impact on energy dependence decreases, while in categories that do not depend on energy consumption, the negative effect is the same as in Scenario 1. In this Scenario, the dewatering unit becomes more important in energy-dependent categories due to the electricity and polyelectrolyte consumption (Figure 5.6c).

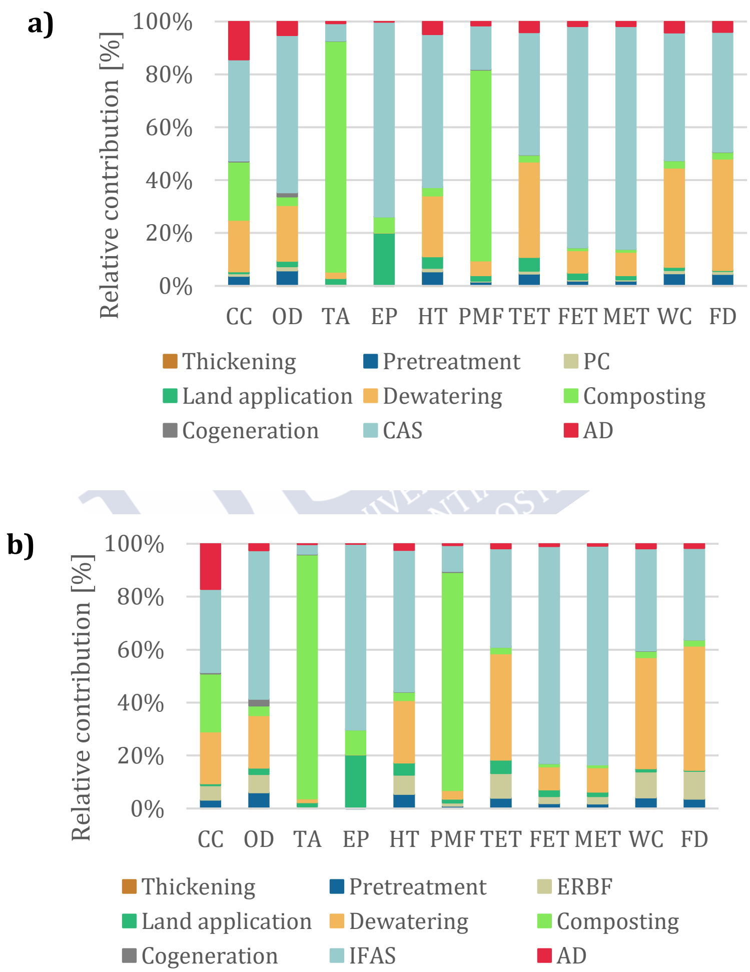

Figure 5.6. Environmental impacts for each scenario considered in Valladolid WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS).

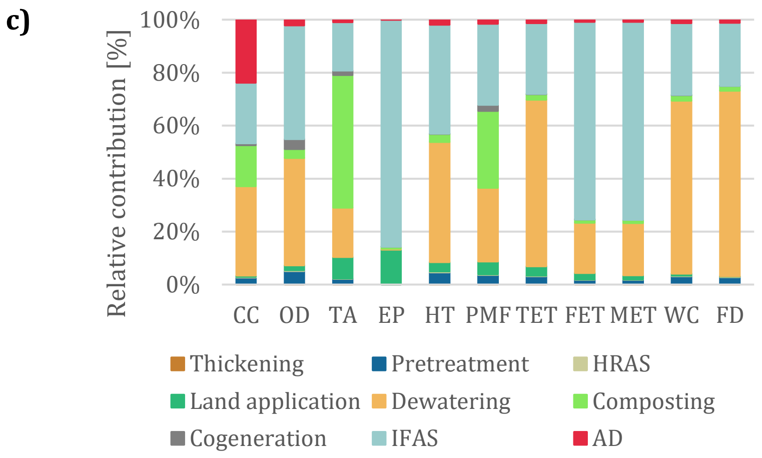

Figure 5.6 (cont.). Environmental impacts for each scenario considered in Valladolid WWTP (FU: 1 kWh of energy produced). (a) Scenario 0 (conventional case) (b) Scenario 1 (ERBF) (c) Scenario 2 (HRAS).

The first environmental analysis (Figures 5.5 and 5.6) provides an insight into the impact that new technologies have on the profile of the WWTP. A priori, the worst environmental profile would correspond to the combination of ERBF, while the impact associated with the HRAS unit can be considered not significant in the impact categories evaluated. An interesting step forward to make a conclusive decision on the selection of the most suitable configuration is to compare the different configurations with each other.

5.3.2. Environmental comparison for different scenarios in both WWTP analysed

In this analysis, only the categories of CC and EP were evaluated due to their special relevance in the environmental profile of WWTPs (Rodriguez-Garcia et al., 2011). CC is related to the energy production and consumption, while EP is related to the quality of the effluent discharged into water courses.

When comparing both plants in terms of these categories (Figures 5.7 and 5.8), the environmental profile decreases when the HRAS unit is integrated followed by the IFAS reactor. The reduction for the CC category, if the values are compared with the conventional scenario, is approximately 68% for the medium plant (Figure 5.7) and 51% for the large plant (Figure 5.8). The main reason for the reduction in the CC category is the increase in biogas production and the reduction in energy consumption.

In addition, for the EP category, the impacts can also decrease by incorporating Scenario 2 (HRAS + IFAS unit): 48% for the small plant (Figure 5.7) and 30% for the large plant (Figure 5.8). The main difference between the environmental profiles for the different scales is that plants have different energy consumption, production or consumption of chemicals. Moreover, the wastewater composition (COD, nutrients or heavy metals) that are treated in each WWTP are different. However, despite the variability found for both plants, the new schemes appear to have a better environmental profile regardless the size of the plant.

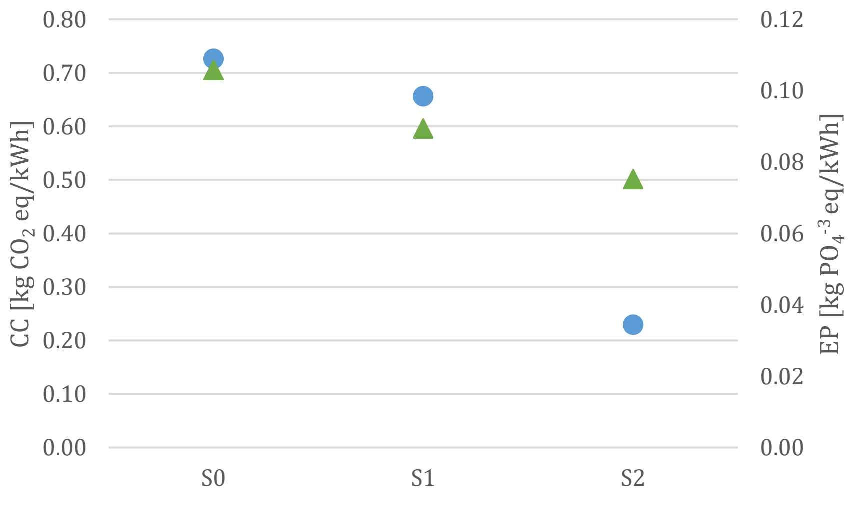

Figure 5.7. Comparison between the different scenarios considered for Avedøre plant. Symbols: (Δ) eutrophication category; (O) climate change category

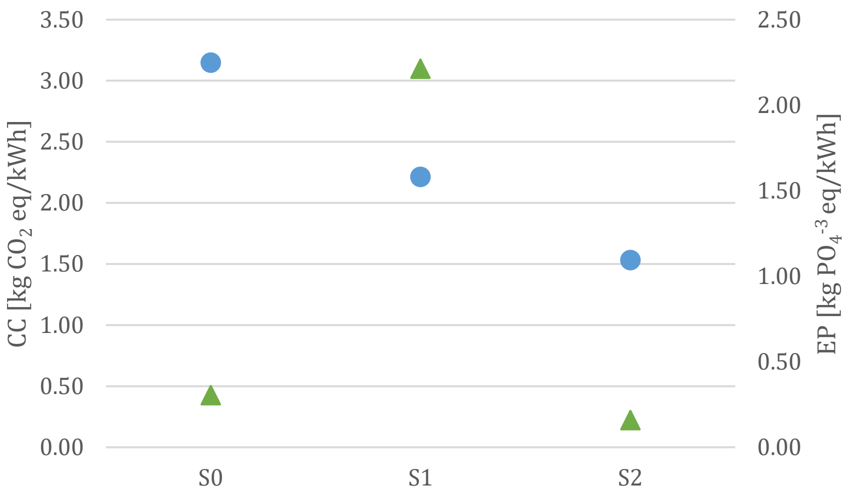

Figure 5.8. Comparison between the different scenarios considered for the Valladolid plant. Symbols: (Δ) eutrophication category; (O) climate change category

5.3.3. Influence of the plant size on environmental impacts

Finally, the incorporation of these new schemes for both plant sizes was compared. As seen above, the best impacts are presented for Scenario 2 in both plant sizes. Although the reduction of impacts is more noticeable in the large plant than in the medium scheme, if the categories are analysed, for the CC category, Scenario 2 has more reduction in the medium plant than in the large plant. On the contrary, in the large plant, the incorporation of the ERBF scheme has better results than in the medium plant (Figure 5.9a). For the EP category, good effluent quality is achieved at the large plant with the incorporation of these technologies. However, although in the medium plant, the reduction is not as great as in large plants, better effluent quality is also obtained (Figure 5.9b). Therefore, although the plants are located in different countries, Scenario 2 (HRAS + IFAS unit) showed a reduction in the CC and EP categories (Figure 5.9). Therefore, the scale or location of the plant does not influence the incorporation of these new technologies in a WWTP, since these technologies could considerably reduce the environmental effects.

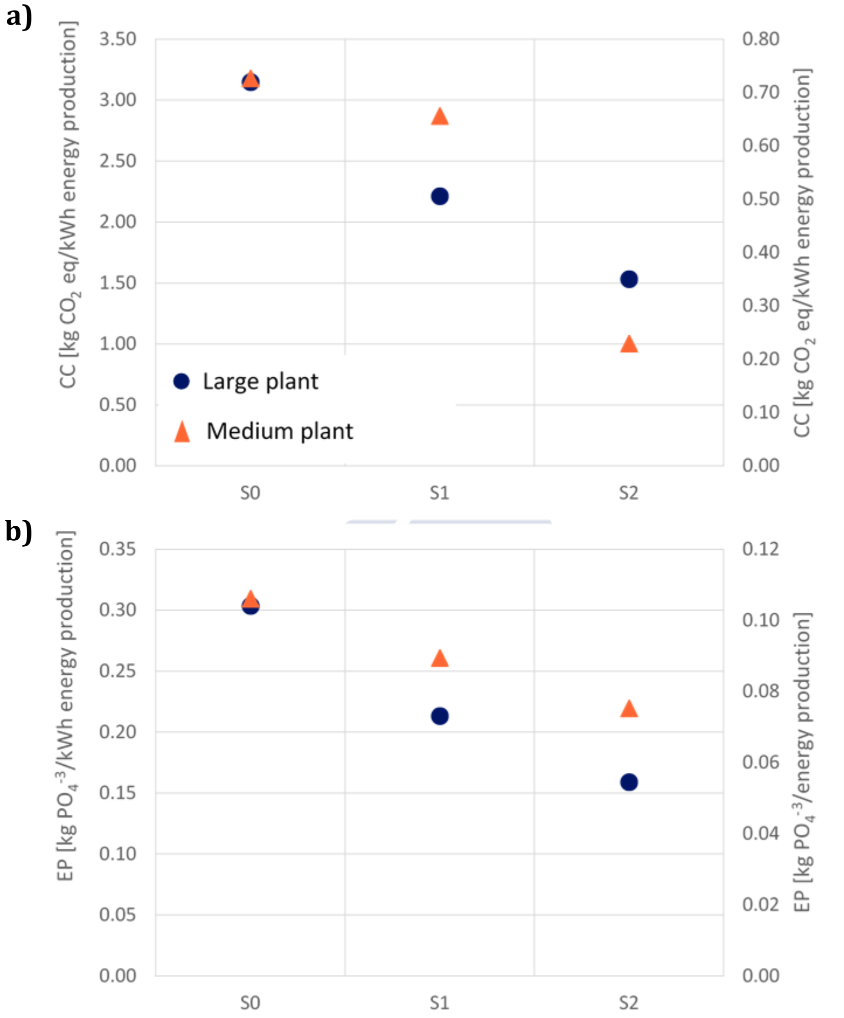

Figure 5.9. Comparison between the different plant sizes (FU: 1 kWh of energy produced). a) Climate change (CC) category; b) eutrophication potential (EP) category. Symbols: (○) large plant; (Δ) medium plant.

5.3.4. Economic results

Table 5.5 shows the economic results for each scenario considered. As in the environmental outcomes, Scenario 2 (HRAS followed by IFAS technology) shows the best economic results for both plant sizes. The implementation of this configuration can reduce the cost between 70% for large plants and 45% for medium plants. Biogas production increases in this scheme and the incorporation of IFAS technology can reduce aeration requirements by 38%. In addition, in Scenario 1 (ERBF followed by IFAS configuration), costs also decrease compared to Scenario 0 (conventional case) by about 19% for the large plant and 23% for the medium plant. In this case, the costs associated with the consumption of chemicals are higher than in the other scenarios. However, aeration electricity may decrease due to the incorporation of the IFAS unit.

As for the final disposal of sludge, incineration has more costs related to electricity consumption than the composting unit (Kelessidis and Stasinakis, 2012). However, the amount of sludge in medium and large plants is not the same; therefore, the costs related to sludge management are higher in the large plant than in the medium plant. But when these technologies are incorporated, the costs associated with sludge disposal can be reduced by 15% for Scenario 1 to 32% for Scenario 2. In general, the incorporation of Scenario 2 can lower all operational costs and Scenario 1 can reduce the cost associated with final sludge disposal and electricity consumption. Although chemical costs will increase, this increase is not reflected in total operating costs.

Table 5.5. Economic results for the different plant sizes and scenarios considered (FU: 1 kWh of energy produced)

	Electricity consumption (€/kWh)	Chemical consumption (€/kg)	Sludge management (€/kg)	Total (€/kWh)
Avedøre				
Case 0	0.25	0.02	0.23	0.51
Case 1	0.22	0.02	0.05	0.30
Case 2	0.04	0.01	0.04	0.10
Valladolid				
Case 0	0.59	0.19	0.26	1.04
Case 1	0.31	0.13	0.18	0.62
Case 2	0.17	0.06	0.09	0.32

5.4. DISCUSSION

5.4.1. Evaluation of the efficiency of different schemes

The main objective of incorporating these innovative technologies is to improve electricity production because biogas is considered a green energy and emissions to the atmosphere are lower than in non-renewable energies (Nair et al., 2014). In this regard, the EROI (defined in the Chapter 2) was calculated (Colosi et al., 2015). Therefore, it must be ensured that the incorporation of innovative technologies is more efficient than conventional systems.

Large plants perform worse in terms of electricity production than medium-sized plants. In large plants, Scenario 0 (conventional scenario) has an efficiency of around 0.33, with the incorporation of innovative schemes, the EROI can be increased by 0.38 for Scenario 1 (ERBF + IFAS) and by 0.69 for Scenario 2 (HRAS + IFAS). The EROI values for the medium plant are better even becoming self-sufficient in Scenario 2. The values are 0.72 for Scenario 0, 0.79 for Scenario 1 and 3.75 for Scenario 2. This is to say, if the medium plant introduces HRAS followed by the IFAS configuration, it could not depend on the grid electricity. In addition, in large plants, this scheme only needs about 31% of the energy from the grid; thus, fossil CO_2 emissions can be reduced.

Energy reduction associated with the Anammox process or enhanced biogas production has been reported at laboratory scale (Cao et al., 2020; Sancho et al., 2019). A similar configuration of the RBF + PC+ denitrification process was evaluated by Gikas (2017), who reported a reduction in electricity consumption of about 85%. This value is close to the scheme of the ERBF + IFAS reactors in the medium plant. The fact that these new schemes to reduce energy consumption and enhance biogas production are still being implemented requires time and background to assess their performance at full scale.

Finally, it is important to note that this energy benefit can reduce indirect energy-related emissions. However, this does not mean that the impacts on the CC category will be zero. In this category, direct air emissions from other units, such as IFAS or AD units, should be considered.

5.4.2. Trade-off analysis of eutrophication impact category

As was mentioned throughout this thesis, eutrophication is one of the most representative impact categories in WWTPs due to the toxicity problems and even mortality of different aquatic species (Zang et al., 2015). It is estimated that the implementation of the conventional nitrification-denitrification process decreases potential eutrophication by 54-58% (Larsen et al., 2007). To evaluate the IFAS technology, as in Chapter 4, the NEB indicator was calculated.

When the PN-Anammox process is included in the WWTPs, the results of nitrogen removal increase in comparison with the conventional case. These removal percentages range from 70% for large plants to 86% for medium plants, which leads to an improvement of between 10 and 20%.

It is important to note that several technologies have been developed to apply the partial nitrification-anammox process for the treatment of domestic wastewater. Ji et al. (2020) reported a nitrogen removal of about 89% using a novel simultaneous nitrogen and phosphorus process consisting of an anammox, endogenous partial-denitrification and denitrifying phosphate removal in an SBR. Gu et al. (2018) studied the feasibility of incorporating an Anammox process in a conventional

WWTP and reported a nitrate removal of 87%. For the treatment of the effluent from the AD unit, this process showed better results and slow growth of biomass, so the amount of sludge can be considered not significant (Morales et al., 2015b). Therefore, the partial nitrification-Anammox can replace the conventional nitrification-denitrification according to the efficiency of nitrogen removal and energy consumption.

5.4.3. Mapping the environmental impact of electricity from WWTPs

When analysing the issue of the water-energy nexus in a WWTP under the LCA approach, it can be observed that the energy produced in cogeneration unit is used in the plant itself. This energy can replace electricity from the grid, and it is considered as green energy. The use of fossil energy implies an unsustainable source of electricity and heat for wastewater treatment. Combining the fact that WWTPs may not be energy self-sufficient with the importance of energy source in terms of energy footprint, the most natural step would be to assess how the electricity mix affects sustainability when assessed through the LCA method (Barragán-Escandón et al., 2017).

In this study, the WWTPs are located in different European countries, so it is interesting to observe how the environmental profile of 1 kWh of energy produced in Spain or Denmark changes. Only, the energy-dependent categories (CC, OD, PMF, TA, HT and FD) were evaluated in this case (Figure 5.10). Denmark has better environmental profile in terms of energy production than Spain. This is because in Denmark about 73% of energy comes from renewable wind and biomass (Danish Energy Agency, 2018). However, in Spain, renewable energy is only 44% (REE, 2018), which means that emissions related to fossil CO_2 are higher in Spain than in Denmark. Thus, wastewater treatment in Spain pollutes more than in Denmark. For this reason, it is very important to have new wastewater treatment systems that consume less energy from the grid and produces more green energy.

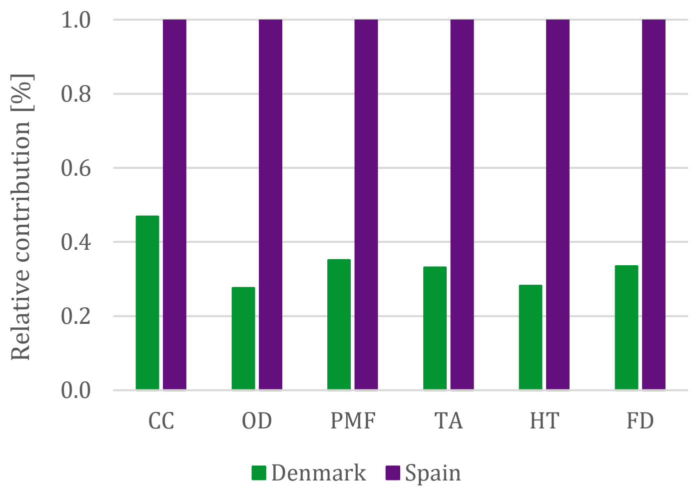

Figure 5.10. Comparison between Spanish and Danish electricity country mix production

5.4.4. Sludge management alternatives

In this study, the final disposal of sludge varies according to the country selected. In Spain, the most common method is land application, while in Denmark, the most common disposal is incineration technology. It is therefore important to know how to change the environmental and economic impacts if one or the other alternative is selected. Incineration is a more expensive alternative to land application due to electricity consumption (Tomei et al., 2016).

However, incineration is not considered environmentally friendly due to the fossil CO_2 emissions in the energy-dependent categories, while composting followed by land application is considered the worst option in the categories that depend on soil emissions associated with heavy metals (Yoshida et al., 2018). But, as mentioned before, in Denmark these emissions are lower than in Spain. In addition, the composting process have air emissions considered GHG emissions such as N_2O or CH_4 (Table 5.2b and 5.3b). Some studies show that direct N_2O emissions can be even more harmful than indirect fossil CO_2 emissions (Rodriguez-Caballero et al., 2014).

To make this comparison reliable, as there are different plant sizes, incineration was considered in the large plant and composting followed by land application was included in the medium plant. Only the CC category was evaluated because it is the category most affected by GHG emissions and electricity consumption. In Denmark, incineration is the best option because land application can increase GHG emissions by 65% associated with N_2O, CH_4 and CO_2 emissions. However, at the plant in Spain, the situation is the opposite. This does not mean that incineration and land application are the best alternatives for treating sludge. Beyond these options, ongoing research is devoted to improve the final management of the sludge such as hydrothermal-pyrolysis (Lishan et al., 2018) or the addition of biopolymers for sludge dewatering (Guo and Wen, 2020).

5.5. CONCLUSIONS

The retrofitting of WWTPs should be addressed under sustainability criteria. It is well known that there are two elements that most penalise wastewater treatment: (i) energy requirements and (ii) sludge management. New technologies should reduce both drawbacks to address technical efficiency, carbon neutrality and reduced economic costs. In this study, several technologies were modelled, two based on OM recovery (HRAS and ERBF) to improve biogas production and another aiming at nitrogen removal (IFAS). Economic and environmental indicators of different plant sizes (one medium and one large) were evaluated and these new schemes: (i) ERBF + IFAS and (ii) HRAS + IFAS, were compared with a conventional scheme (PC + CAS with nitrogen removal).

These schemes based on OM recovery followed by partial nitrification-Anammox showed better environmental and economic results than conventional schemes due to higher biogas production and lower energy consumption. Furthermore, the incorporation of the IFAS unit improved the quality of the effluent in terms of nutrient removal. Although these technologies are more complex than conventional ones, they also showed a better economic profile despite the size of the plant. These positive results are only possible considering the production of energy through biogas valorisation according to the waste-to-energy scheme.

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CHAPTER 6: **The bottom-up approach in the assessment of environmental impacts and costs of the ELAN® process for nitrogen removal**

SUMMARY

In recent decades, the wastewater treatment sector has undergone a shift to adapt to more restrictive discharge limits. When addressing the evaluation of innovative technologies, it is necessary to determine the scale at which reliable and representative values of environmental impacts and costs can be obtained, ensuring that the system under assessment follows the direction of eco-efficiency.

Chapter 6 has evaluated the environmental and economic indicators of an ELAN® system from laboratory conception (1.5 L) to full scale (2 units of 115 m^3) using the LCA methodology. Indirect emissions related to electricity consumption are the main contributor in all impact categories except EP category. The electricity consumption referred to the FU (1 m^3 of treated wastewater) decreases as the scale increases. The rationale behind this can be explained, among other reasons, by the low energy efficiency of small-scale equipment (pumps and aerators). As a result, a value of approximately 25 kg CO_{2eq}/m^3 of treated wastewater is determined at laboratory scale, compared to only 5 kg CO_{2eq}/m^3 at full-scale. When it comes to assessing the reliability of data, a pilot scale system of 0.2 m^3 allowed to perform a trustworthy estimation of environmental indicators, which were validated at full-scale. In terms of operational costs, the scale of approximately 1 m^3 provided a more accurate estimate of the costs associated with energy consumption.

Andrea Arias[a], Iana Salim[a], Alba Pedrouso[a], Nicolás Morales[b], Anuska Mosquera-Corral[a], José Ramón Vázquez-Padín[b], Frank Rogalla[b], Gumersindo Feijoo[a] and María Teresa Moreira[a]. “Bottom-up approach in the assessment of environmental impacts and costs of an innovative anammox-based process for nitrogen removal”. Journal of Environmental Management. 2018, Vol 225, pp 112-119. https://doi.org/10.1016/j.jenvman.2018.07.070. [a]CRETUS institute. Department of Chemical Engineering. Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain. [b] Aqualia, Guillarei WWTP, Camino de la Veiga s/n, E-36720, Tui, Spain

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6.1. INTRODUCTION

In the design of new processes and products, there is a growing demand to label them as sustainable from the early stages or their conception and development. Traditionally, the evolution of an innovative technology, from its conception to its implementation in the market, consists of overcoming a series of successive stages of development, in which performance and operational conditions vary according to the scale, making them comparable to conventional technologies. When introducing environmental and economic perspectives, it is necessary to evaluate the scale level that allows reliable and representative values of environmental impacts and costs to be obtained, ensuring that the emerging technology moves in the direction of eco-efficiency. This stage is critical, as it will mean the “abandonment” or “scale-up” of R&D activities to large-scale installation.

In this context of wastewater treatment, the reduction of the nitrogen load in treated effluents is one of the main objectives to avoid eutrophication and toxicity, which negatively affect aquatic fauna and flora (Li and Brett, 2012). According to the Water Framework Directive (2000/60/EC), a nitrogen limit of 10-15 mg N/L is applied for European WWTPs in sensitive areas, provided that 70-80% of the total nitrogen in the influent is removed. This increased restriction in the legislation leads to the development of novel treatment technologies that need to be validated from environmental and economic points of view (Machado et al., 2009; Wang et al., 2012). Several authors highlighted the balance between nitrogen removal and energy demand, which may lead to an increase in indirect GHG emissions depending on the complexity of the treatment scheme (Lederer and Rechberger, 2010; Vidal et al., 2002).

As mentioned in Chapter 1, conventional nitrogen removal processes require a high energy consumption for aeration and, sometimes, the addition on external carbon sources, which can increase the operational costs (Renzi et al., 2015). For this reason, interesting alternatives to conventional nitrification-denitrification processes such as OLAND (Oxygen Limited Autotrophic Nitrification-Denitrification) (Kuai and Verstraete, 1998) or ELAN® (Vázquez-Padín et al., 2014) have been developed in recent years. These technologies are applied for the treatment of the supernatant of the anaerobic sludge digesters, which are nutrient-rich side streams in WWTP and can reduce oxygen requirements, with no consumption of OM and with an extraordinarily low biogas yield (Vázquez-Padín et al., 2010b).

With the main objective of assessing the sustainability of water treatment technologies, LCA methodology emerges as a good alternative. However, the tendency to use LCA to "prove" the superiority of one product over another has discredited the concept in some areas (Heijungs et al., 2010). One of these weaknesses is attributed to the collection and validity of data required for the LCI. This stage is critical as it will compute the consumption of raw materials, chemicals, water and energy for each stage of the process, as well as emissions to air, water and soil (Finnveden, 2000; Tillman, 2000). When the inventory data are executed on reliable data, it is possible to obtain accurate environmental impacts. This includes the need to make judgements based on the figures collected to assess the likely significance of the various impacts (Reap et al., 2008). However, uncertainty arises regarding the scale of development, this drawback is even more important. Therefore, the definition of the necessary developmental scale, which provides reliable data for LCA, is relevant to ensure the successful implementation of a bottom-up approach.

The main objective of Chapter 6 is to define the scale at which data collection in the LCA methodology provides a reliable assessment of a technology under development. In particular, the assessment of an innovative wastewater treatment technology for nitrogen removal (ELAN®) was conducted from laboratory conception to full-scale.

6.2. MATERIALS AND METHODS

6.2.1. Description of the ELAN® technology

The ELAN® technology combines partial nitrification and anammox processes in the same unit (Vázquez-Padín et al., 2010b). In the partial nitrification process, ammonium oxidising bacteria (AOB) oxidise ammonium to nitrate, while the oxidation of nitrite to nitrate by oxidising bacteria (NOB) should be avoided (Vázquez-Padín et al., 2009). Anammox bacteria are capable of oxidising ammonium to nitrogen gas using nitrite as electron acceptor, without the need of OM or oxygen (Dapena-Mora et al., 2004). Thus, in the ELAN® technology, nitrogen is autotrophically removed. The ELAN® technology was developed in a SBR with a granular sludge. The establishment of aerobic and anoxic zones within the granule, depending on oxygen depth penetration, allows the operation in a single step (Morales et al., 2015a). The SBR operational cycle comprised the following stages: feeding, aerobic reaction, settling and withdrawal (Figure 6.1).

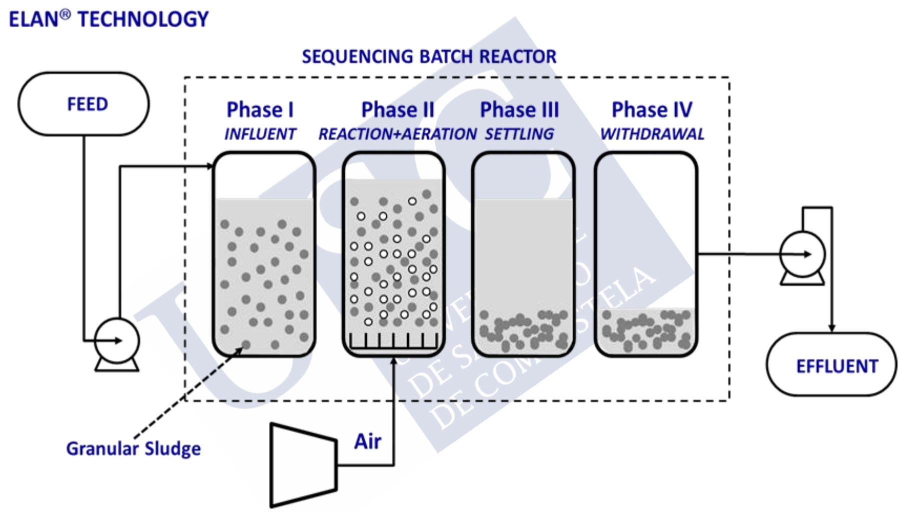

Figure 6.1. Operational cycle of the ELAN® process

Four different reactor sizes (from 1.5 L to 115 m^3) were analysed in this study (Table 6.1): Laboratory Scale (LS), Pilot Plant 1 (PP1), Pilot Plant 2 (PP2) and Full-Scale (FS) (Figure 6.2). The LS reactor, operated under the approach of the ELAN® process, operated at fixed-cycle duration of 3 h for the entire duration of the operating cycles. The volume exchange ratio (VER), or ratio between the volume of effluent discharged and the volume of the reactor, was 25%. The retention time of the pilot and full-scale reactors varied, as this phase stopped when the conductivity values and/or pH reached a certain set point. In addition, the operational strategy was adapted on the basis of hydraulic retention time (HRT) and dissolved oxygen concentration (Vázquez-Padín et al., 2010b) and following the "conductivity versus time slope" as a method for reactor monitoring, as detailed by Vázquez-Padín et al., (2014). For this purpose, the reactor is equipped with a set of probes (conductivity, pH...) connected to a control system. In this study, an average cycle time of 6 hours was considered for reactors PP1 and PP2, and 8 hours for the FS reactor. The VER values of each unit was: 25% for PP1, 21% for PP2 and, finally, 44% for FS.

Table 6.1. Description of the technical characteristics and operational conditions of the different ELAN® reactors evaluated (Morales et al., 2015a; Vázquez-Padín et al., 2009)

	LS*	**PP1**	**PP2**	**FS**
Material	Glass	Stainless Steel	Glass-Fiber	Reinforced concrete
Volume	1.5 L	0.2 m^3	1.2 m^3	115 m^3 (97 m^3)
Power (kW/m^3)	140	16.5	0.90	0.16
T (^{0}C)	18-24	24-30	24-30	24-30
pH	7.7	7.4	7.7	7.5
VER (%)	25	25	21	44
HRT (d)	0.5	1	1.2	0.75
DO (mg O_2/L)	0.5	1.5	0.5	0.2-0.5
NLR (kg N/m^3·d)	0.25	0.77	0.45	0.46

*LS: Laboratory Scale, PP: Pilot Plant; FS: Full Scale

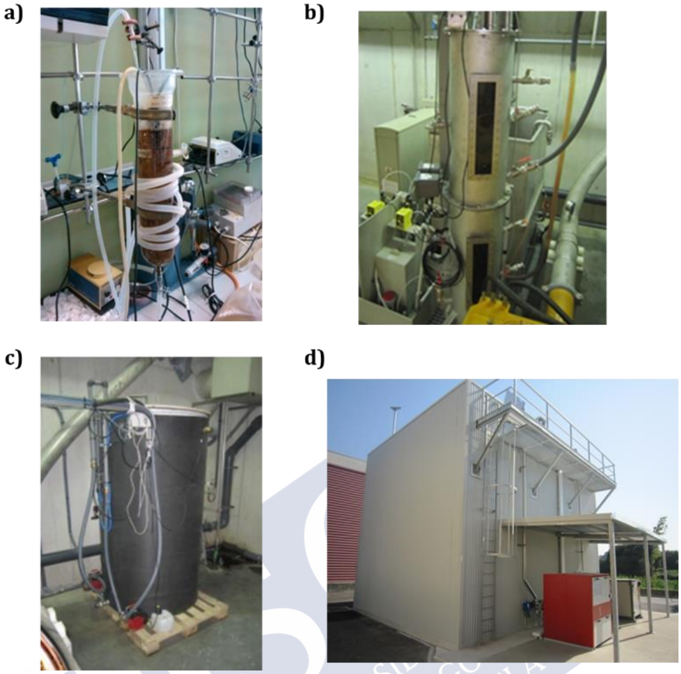

Figure 6.2. Different scales of the ELAN® process: a) Laboratory scale: 1.5L, b) Pilot Plant 1: 200L; c) Pilot Plant 2: 1.2m^3; d) Full-scale: 115m^3

6.2.2. Approach for data collection in LCA for different ELAN® sizes

A gate-to-gate approach was applied in this study and only impacts occurring in the operational phase were considered (Lundie et al., 2004). The construction phase was not taken into account because the infrastructure of each reactor is made up of different materials depending on the scale, availability and cost, which determines that emissions from this phase between the small and full scales are not comparable (Table 6.1). Therefore, only the environmental impacts associated with the operational phase of each reactor were assessed in this study.

As mentioned in Chapter 1, the most common FU used in WWTP LCA studies are the following: population equivalent (Gallego et al., 2008; Machado et al., 2007), kg TN removed (Hauck et al., 2016; Rodriguez-Garcia et al., 2011) or m^3 of treated wastewater (Pasqualino et al., 2011).

Depending on the approach of the different scales, population equivalent is not applied in the LS, PP1 or PP2 scenarios. Consequently, 1 m^3 of treated wastewater was selected as FU, which can be a straightforward solution when comparing different scales of operation. In addition, a sensitivity analysis was performed considering a FU of kg TN removed for the benchmarking of the environmental outcomes

The LCI has been developed with primary data from the laboratory scale, two pilot plants reactors and full-scale reactor, obtaining during the different stages of development of the ELAN® process (Tables 6.2, 6.3, 6.4 and 6.5 respectively). The laboratory scale reactor was operated in the University of Santiago de Compostela (Spain). Pilot and full-scale ELAN® reactors were operated in Guillarei WWTP (Northwest of Spain) by Aqualia company, since 2012 and 2105, respectively.

Emissions to air (NO, N_2O and CO_2) were calculated according to Kampschreur et al., (2008) and Morales et al., (2015b). The energy consumption of the reactors has been calculated according to the operating time and power of the pumps used. The Ecoinvent v3.5 database for the Spanish electricity production and import/export mix process was updated for 2018 with data from the annual report of Red Eléctrica Española 2018 (REE, 2018). In Spain, WWTPs use medium-voltage electricity (Lorenzo-Toja et al., 2016b); thus, the high voltage electricity was converted to medium voltage, considering air emissions and losses in transport (Dones et al., 2007).

Table 6.2. Life cycle inventory of LS (1.5 L) per 1 m^3 of treated wastewater. Adapted from Vázquez-Padín et al., (2009)

INPUTS		OUTPUTS	
From the technosphere		**To the environment**	
Materials and fuel		***Emissions to water***	
Water Influent		COD (g)	95.1 ± 54.1
COD (g)	278.5 ± 155.6	TN (g)	51.8 ± 32.4
TN (g)	233.4 ± 27.9	NO_2^- -N (g)	0.6 ± 0.3
NH_4^+-N (g)	233.4 ± 27.9	NO_3^--N (g)	28.5 ± 4.6
TP (g)	47 ± 16.1	NH_4^+-N (g)	25.7 ± 15.2
Electricity		TP (g)	33 ± 12.3
Aeration (kWh)	60	***Emissions to air***	
Feeding (kWh)	4.8	NO (mg)	0.001
Emptying (kWh)	1	N_2O (mg)	0.01
		To the technosphere	
		Products	
		Net Sludge production (g TSS)	0

Table 6.3. Life cycle inventory of PP1 (0.2 m^3) per 1 m^3 of treated wastewater (data supplied by Aqualia)

INPUTS		OUTPUTS	
From the technosphere		**To the environment**	
Materials and fuel		***Emissions to water***	
Influent		TSS (g)	0.26 ± 0.19
TSS (g)	0.52 ± 0.44	VSS (g)	0.23 ± 0.16
VSS (g)	0.40 ± 0.26	COD (g)	214 ± 29.2
COD (g)	405 ± 95.3	TN (g)	202.9 ± 69.9
TN (g)	1122 ± 272	NO_2^- -N (g)	1.86 ± 1.0
NO_2^- -N (g)	0	NO_3^--N (g)	53 ± 25
NO_3^-- N (g)	0	NH_4^+-N (g)	148 ± 43.9
NH_4^+-N (g)	1122 ± 272	TP (g)	36.5 ± 12.3
TP (g)	48 ± 16.1	***Emissions to air***	
Electricity		CO_2 (mg)	3.79
Aeration (kWh)	7.37	NO (mg)	0.002
Feeding (kWh)	1.25	N_2O (mg)	0.02
Emptying (kWh)	1.25	**To the technosphere**	
		Products	
		Net Sludge production (g TSS)	0

Table 6.4. Life cycle inventory of PP2 (1.2 m^3) per 1 m^3 of treated wastewater (data supplied by Aqualia company)

INPUTS		OUTPUTS	
From the technosphere		**To the environment**	
Materials and fuel		***Emissions to water***	
Influent		TSS (g)	0.24 ± 0.3
TSS (g)	0.42 ± 0.5	VSS (g)	0.18 ± 0.2
VSS (g)	0.20 ± 0.1	COD (g)	152 ± 104
COD (g)	229 ± 141	TN (g)	216.4 ± 84
TN (g)	808 ± 162.8	NO_2^- -N (g)	2.40 ± 3.6
NO_2^- -N (g)	0.00	NO_3^--N (g)	75 ± 38.5
NO_3^--N (g)	0.00	NH_4^+-N (g)	139 ± 83.7
NH_4^+-N (g)	808 ± 162.8	TP (g)	33.6 ± 4.5
TP (g)	47 ± 3.71	***Emissions to air***	
Electricity		CO_2 (mg)	5.89
Aeration (kWh)	5.98	NO (mg)	0.001
Feeding (kWh)	0.26	N_2O (mg)	0.01
Emptying (kWh)	0.26	**To the technosphere**	
		Products	
		Net Sludge production (g TSS)	0

Table 6.5. Life cycle inventory of FS (97 m^3) per 1 m^3 of treated wastewater (data supplied by Aqualia company)

INPUTS		**OUTPUTS**	
From the technosphere		**To the environment**	
Materials and fuel		***Emissions to water***	
Water Influent		TSS (g)	0.3±0.2
TSS (g)	0.4±0.4	VSS (g)	0.2±0.1
VSS (g)	0.2±0.4	COD (g)	171.3±31
COD (g)	284.1±55.2	TN (g)	228.8±55.8
TN (g)	797.7±102.8	NO_2^- - N (g)	5.9±6.1
NO_2^- - N (g)	0.00	NO_3^-- N (g)	93.1±18.3
NO_3^-- N (g)	0.00	NH_4^+- N (g)	109.7±23.2
NH_4^+- N (g)	569.1±20.4	TP (g)	44.8±17.6
TP (g)	61.2±34.9	***Emissions to air***	
Electricity		CO_2 (mg)	2.3
Aeration (kWh)	0.7	NO (mg)	0.001
Feeding (kWh)	0.1	N_2O (mg)	0.01
Emptying (kWh)	0.01	**To the technosphere**	
		Products	
		Net Sludge production (g TSS)	0

6.2.3. Environmental assessment-Life Cycle Assessment

The SimaPro 9.0 software was used for the impact assessment. Two different assessment methods were used to provide the most characteristic environmental impacts of WWTPs (Rodriguez-Garcia et al., 2011). EP was calculated using CML 2001 method (Guinée, 2002). CC, OD, TA, POF, PMF, HT, TET, FET, MET, WC and FD were calculated with the ReCiPe Midpoint (H) v1.1 (Huijbregts et al., 2017).

6.2.4. Economic sustainability indicator

The operational costs related to electricity consumption were selected as economic indicator. The amount of sludge generation in the ELAN® process is considered negligible (Vázquez-Padín et al., 2014), so the cost of sludge is not taken into account in this study. Furthermore, since there is no addition of chemicals for the operation of the reactors, the costs associated with the consumption of chemicals are not considered (Vázquez-Padín et al., 2014).

6.2.5. Uncertainty analysis methodology

The management of WWTPs faces variable operating conditions, flows and composition to be treated, which can strongly influence the results of the LCA studies (Yoshida et al., 2014). The most probable uncertainty factors are: (i) the uncertainty of the parameters such as the calibration of the measurement equipment, human errors or mismatches between different measurements of the same parameter and (ii) the uncertainty associated with the background processes included in the databases, such as electricity consumption (Hauschild et al., 2011). In this study, the Monte Carlo uncertainty method included in the SimaPro 9.0 software was applied. In this method, four types of probability can be considered: uniform, triangular, normal and lognormal (Fantin et al., 2015). For the background parameters (Ecoinvent v3.5 databse), the lognormal is the default selected probability distribution, while for the water characterisation parameters the normal distribution was selected. According to other studies (Guo and Murphy, 2012; Longo et al., 2017), the Monte Carlo analysis was performed with 1,000 iterations at a 95% significance level.

6.3. RESULTS

6.3.1. Environmental and economic profiles for ELAN® reactors with different sizes

The process that most contributes to the impact of the different environmental categories is energy consumption, mainly associated with the aeration process (Tables 6.2 to 6.5), which has a drastic effect when considering the scale of the reactor, since at the small scale (corresponding to the first stages of technology development), the equipment used (pumps and aerators) is over-dimensioned, which translates into a large consumption of electricity, and therefore, greater impacts (Figure 6.3).

Table 6.6. Environmental results of the different reactors, resulting in the ELAN® process, for the impact categories under assessment. FU: 1 m^3 of treated wastewater. LS: 1.5 L, PP1: 0.2 m^3, PP2: 1.2 m^3 and FS: 97 m^3.

Impact Categories	LS	PP1	PP2	FS
CC (kg CO_2 eq)	24.39	9.46	6.24	4.62
OD (kg CFC-11 eq)	$3.02 \cdot 10^{-6}$	$4.51 \cdot 10^{-7}$	$2.97 \cdot 10^{-7}$	$1.02 \cdot 10^{-7}$
TA (kg SO_2 eq)	0.12	0.02	0.01	$4.17 \cdot 10^{-3}$
EP (kg PO_4^{3-} eq)	0.14	0.26	0.25	0.28
HT (kg 1,4-DCB eq)	5.13	0.77	0.50	0.17
POF (kg NMVOC)	0.06	0.01	0.01	$2.11 \cdot 10^{-3}$
PMF (kg PM_{10} eq)	0.04	0.01	$4.30 \cdot 10^{-3}$	$1.48 \cdot 10^{-3}$
TET (kg 1,4-DCB eq)	$5.33 \cdot 10^{-4}$	$7.97 \cdot 10^{-5}$	$5.25 \cdot 10^{-5}$	$1.81 \cdot 10^{-5}$
FET (kg 1,4-DCB eq)	0.38	0.06	0.04	0.01
MET (kg 1,4-DCB eq)	0.34	0.06	0.04	0.01
WC (m^3)	0.19	0.03	0.02	0.01
FD (kg oil eq)	5.39	0.80	0.53	0.18

As the scale increases, energy consumption is reduced. The reduction from PP1 to FS is not very high, approximately 9%. This reduction is more important when the scale is increased from LS to FS (75%), which is attributed to the oversizing of pumps and aerators used at small scale. This reduction of energy translates into a lower impact in the different impact categories that are energy dependent (Table 6.6). The impact reduction is the same for all categories (about 75% from LS to FS) except for the CC category.

In the CC category, the impact is caused by the non-biogenic CO_2 emitted from the combustion of fuel fossils. Emissions are reduced as the scale increases from 55% in LS to 10% for FS (Figure 6.3a). In PP1, PP2 and FS, the emissions values are very similar, with impact reductions of 10 to 20% (Figure 6.3a). Considering that the final objective of a WWTP is to reduce the organic load and eutrophication impact, one of the environmental categories classified as essential is EP. This category does not depend on energy consumption, and compared to the other impact categories, the values show an opposite trend and change significantly between one configuration and the others (Figure 6.3b). The LS has a lower eutrophication potential (15%) due to the composition of the wastewater fed into the reactor with a lower concentration of nitrogen, about 77% in comparison to the FS (Vázquez-Padín et al., 2009). For this reason, the impact on the EP category for LS is not sufficiently realistic to be compared with that of the other pilot or full-scale reactors.

For the PP1, PP2 and FS systems, the impact is very similar, approximately 30%. These reactors treated the reject water from the sludge digester in Guillarei municipal WWTP and the removal of compounds such as COD, TN (organic and inorganic) or phosphorus that generate impact in this category was considered for the calculation (Table 6.2 to Table 6.5). Thus, the comparison in the EP category is only feasible between the pilot and full-scale reactors. Since the ELAN® process accomplishes nitrogen removal, it would be interesting to benchmark the eutrophication that it "reduces" in comparison with a conventional system operated for the same purpose, or just the effect, on the secondary treatment of the WWTP where the reject water from the sludge anaerobic digester is recycled; nevertheless, it is beyond the scope of this study.

The effect on the HT category is associated with the indirect emissions from the electricity production. In Figure 6.3c, it can be seen that LS has the largest impact and for the PP1, PP2 and FS, this impact decreases with the size. The reduction from LS to PP1 is 66% whereas the HT impact is further decreased to 75% in FS.

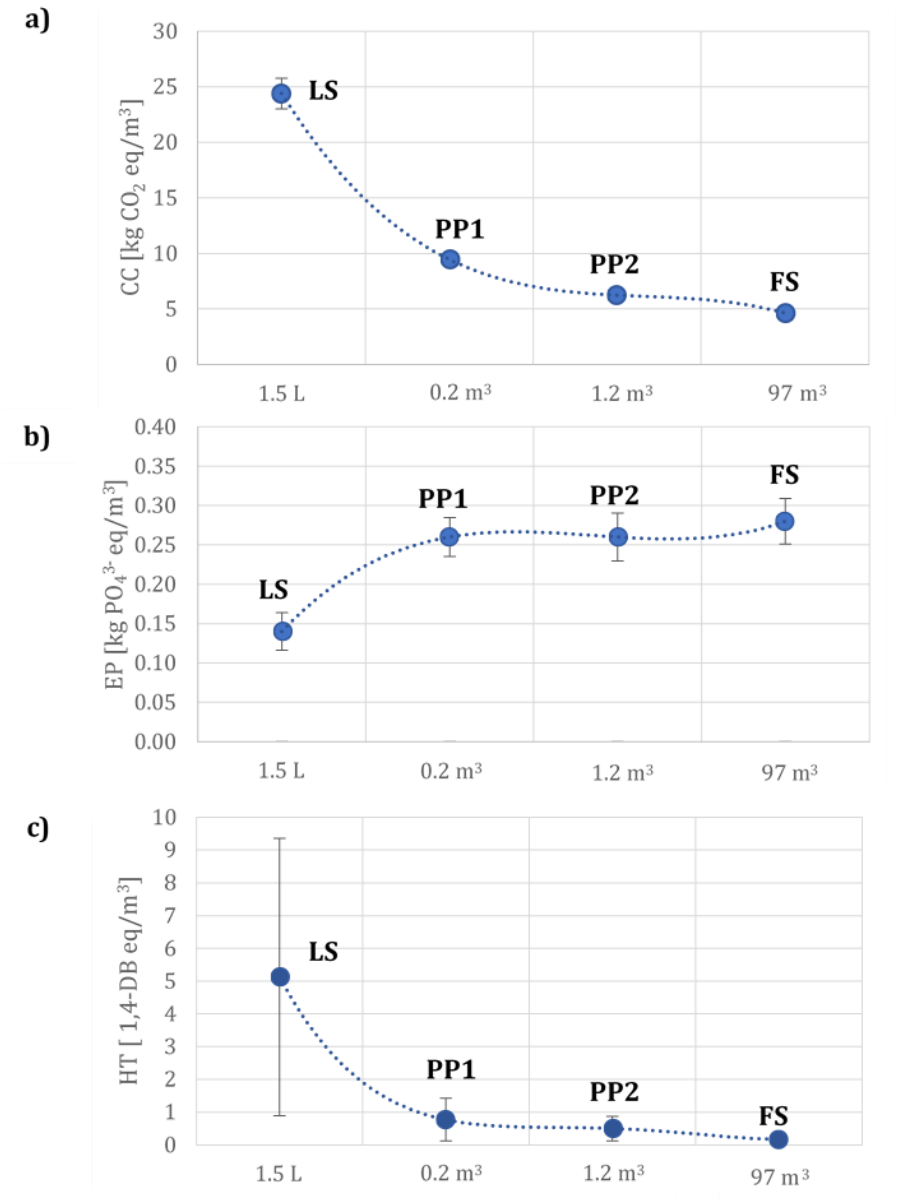

Figure 6.3. Comparison of environmental impacts obtained from the different reactor sizes: a) CC b) EP c) HT impacts

Since there is no consumption of chemicals and the amount of sludge produced can be considered minor, only the operational costs related to the electricity consumption in the reactors evaluated for the development of the ELAN® process were analysed for the economic assessment. The electricity costs are represented in Figure 6.4 per 1 m^3 of treated wastewater (€/m^3), ranging from 8 €/m^3 (LS) to 0.3 €/m^3 (FS). These values are related to the CC impacts of each reactor.

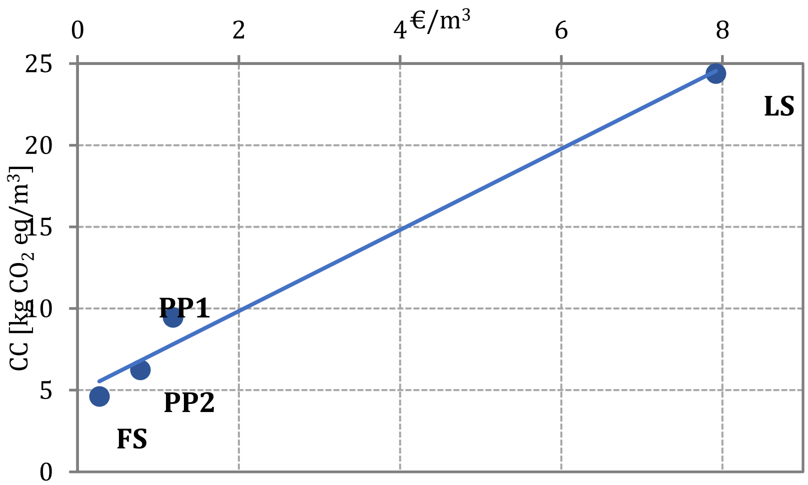

Figure 6.4. CC impact and costs per 1 m^3 of treated wastewater. LS: 1.5 L, PP1: 0.2 m^3; PP2: 1.2 m^3 and FS: 97 m^3

6.3.2. Uncertainty analysis at different ELAN® scales

The uncertainty for the different environmental categories can be represented in terms of the coefficient of variation defined as the ratio of the variability of the data to the standard deviation (Figure 6.5). The uncertainty is independent of the scale of the facility since the same behaviour was found for all categories. Furthermore, uncertainty was less than 30% for all categories with the exception of "Human Toxicity" category. The value of the environmental impact derived from this category depends to a large extent on the electricity production process considered and, more specifically, on the effect of the heavy metals associated with the process. The electricity consumption of the different treatment systems was primary data, but the profile and processes of electricity generation are secondary data (obtained from Ecoinvent v3.5 database). The Ecoinvent processes tend to have a high uncertainty that affects the results and for this reason the uncertainty is higher in this category from 74% in PP2 to 85% in LS. Consequently, the data obtained for the environmental impact study of the ELAN® technology according to the scale of the reactor can be considered representative.

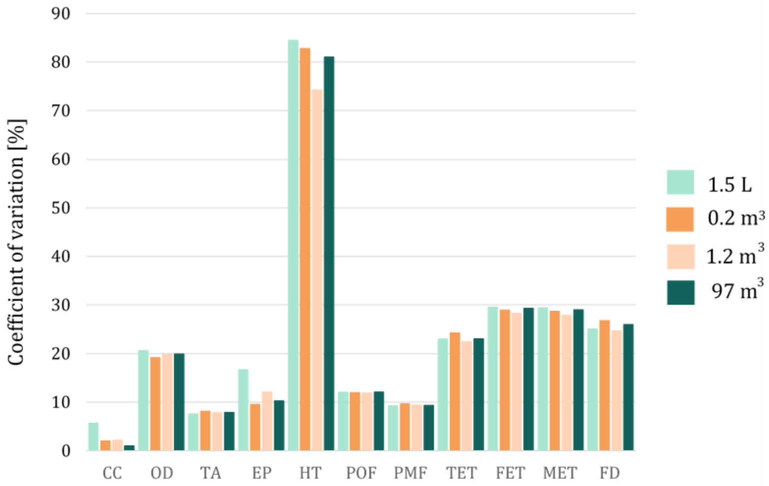

Figure 6.5. Coefficient of variation for each reactor

6.4. DISCUSSION

Currently, extrapolation of laboratory scale emissions to industrial facilities can only be estimated, not measured. However, estimation by bottom-up techniques (i.e. using scale factors) can produce overestimated impacts. By selecting an appropriate scale of development, we can produce inventories that they are neither over nor underestimated to the extent possible, and where uncertainties are reduced. When LCA is used to support decision making, confidence in the LCA data must be ensured. Ideally, inventory data are validated, and uncertainty can be quantified. Obtaining reliable inventory data that are clearly described and precisely reported is not easy and can seriously hamper the implementation of the LCA methodology. The use of published inventory databases may be useful only for background processes, but not especially when it is an innovative technology in its early stages of development. This will help to understand the magnitude of environmental impacts and is a key element in reporting progress and monitoring changes associated with improvement measures towards targets.

6.4.1. Impact categories dependent on electricity consumption in the ELAN® units

In this study, the indirect emissions caused by energy consumption are presented in all categories except EP. It should be noted that electricity emissions depend on the electricity mix of each country. In Spain, electricity production is represented by 59.2% of non-renewable energy and 40.8% of renewable energy (REE, 2018).

As indicated in Section 6.2.1, the ELAN® technology includes a number of energy consuming operational stages (feeding, aeration and withdrawal) (Figure 6.1). Energy consumption should be optimised, as it is a parameter that directly affects the CC category and the major contribution of the different environmental categories. The electricity consumption decreases as the scale increases (FS<PP2<PP1<LS) (Figure 6.3). Consequently, the impacts should be reduced as the scale increases. In the LS or PP1, the installed pumps and aerators were oversized. Accordingly, for the analysis of the LS and PP1 reactors, equipment with reduced energy consumption was not considered. The reduction of LS to PP1 is significant of 56% while the reduction of PP1 to FS represents only 9%. This means that the environmental study would be adequate from a reactor volume of 0.2 m^3 if the process is optimised in terms of installed power (pumps and aerators).

Direct emissions from the partial nitrification-Anammox process come from nitrogen compounds (NO and N_2O). The estimated direct emissions in the ELAN® reactors, in the absence of primary data, do not change significantly for LS and FS. These emissions increase by almost 16% (FS), estimated from the amount of nitrogen removed and validated with the ratios reported for partial nitrification-anammox reactors (Kampschreur et al., 2008). However, this scale is not relevant for comparison with the indirect emissions, which show an increase of approximately 55% from LS to FS reactors.

The conventional nitrification/denitrification processes have a higher electricity consumption than the ELAN® technology, which is mainly attributed to the energy use in the aeration process. The indirect emissions associated with the CC category in conventional reactors are 10.37 kg CO_2 eq/m^3 of treated effluent while in the ELAN® full-scale reactor, the emissions responsible for the CC amount to 4.62 kg CO_2 eq/m^3. This suggest that the use of an ELAN® system instead of a conventional nitrification/denitrification process in the sidestream could reduce emissions by approximately 57%. Even for innovative alternatives such as the SHARON-Anammox technology (partial nitrification-Anammox processes in separate units), the estimated emissions are comparatively higher (up to 13% for NO and N_2O) than in the ELAN® technology where partial nitrification-Anammox takes place in a single unit (Kampschreur et al., 2008; Van Dongen et al., 2001). The fact that low CC impact is produced indicated that the treatment costs will be presumably lower in the case of the ELAN® as well.

6.4.2. Sensitivity analysis of the functional unit (FU)

The functional unit is a relevant decision in the LCA methodology. The selection of two different functional units (1 m^3 of treated wastewater and 1 kg TN removed) for the EP and CC categories (Figure 6.5) were investigated.

The CC category was considered because it depends largely on indirect GHG emissions from electricity consumption, especially during secondary treatment (Lorenzo-Toja et al., 2016b). The consideration of eutrophication finds its interest in the operation of nutrient removal systems for wastewater treatment. It has been reported that the application of a nitrification/denitrification process implies a 54-58% reduction of EP in the mainstream of WWTPs (Larsen et al., 2007). However, the ELAN® reactors upon being a sidestream (reactors in the sludge line) such as other reactors located in the same place, do not lead to the discharge of the treated effluent directly into water bodies, but it is treated in a subsequent phosphorus recovery unit (struvite precipitation) or it is returned to the headwaters of the WWTP (Morales et al., 2015b), causing no increase of the nitrogen load of the mainstream and improving as a consequence the quality of the effluent from the WWTP. Moreover, the only impact category that is not fundamentally dependent on electricity consumption is EP category. Figures 6.6a and 6.6b show that the values of the two functional units are very similar. Therefore, the choice of another FU would not change the results of this study and the appropriate size for an environmental study would remain the same (0.2 m^3 reactor).

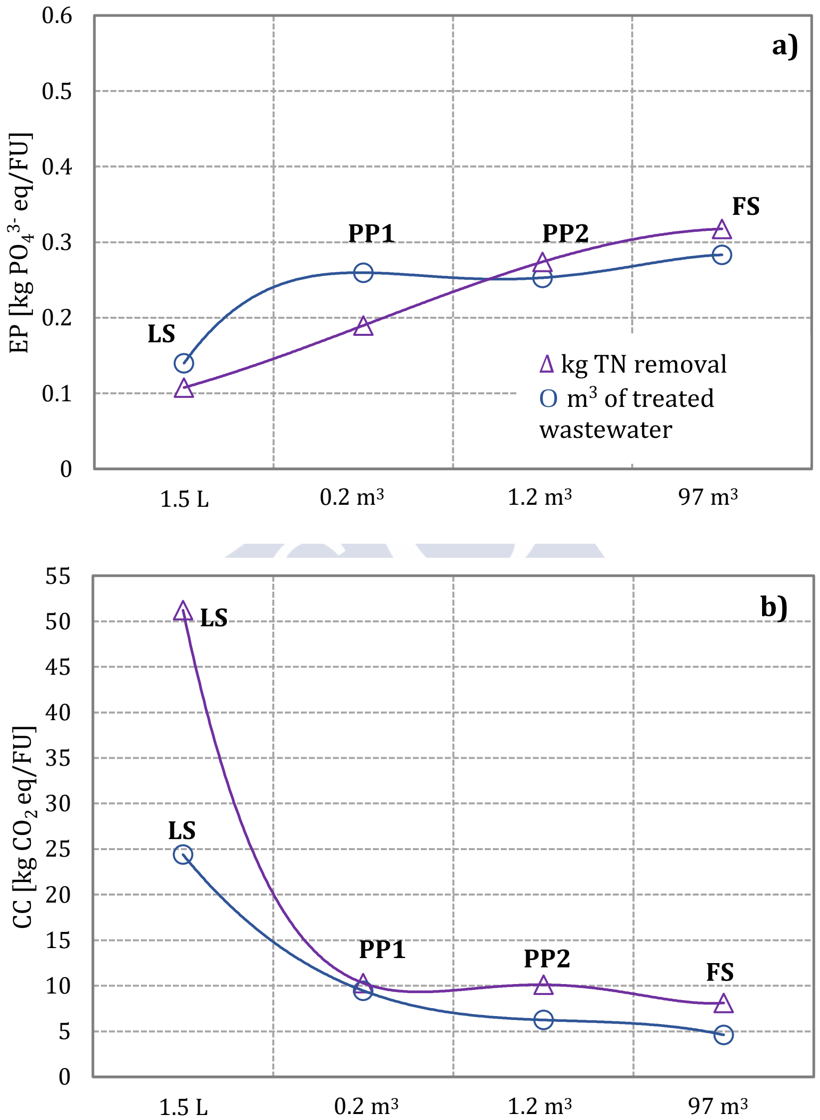

Figure 6.6. a) Comparison between two different functional units (1 m³ of treated wastewater and 1 kg TN removal) for EP category. b) Comparison between two different functional units (1 m³ of treated wastewater and 1 kg TN removal) for the CC category. LS: 1.5 L; PP1: 0.2 m³; PP2: 1.2 m³ and FS: 97 m³

6.4.3. Validation of the data used for the different ELAN® sizes

As indicated above, the composition of the wastewater presents a significant degree of variability, which may condition the results of the LCA study. Therefore, it is important to validate the data, but sometimes this is difficult because a large number of measurements are required and the aggregation of the data into impact categories may mean the loss of a precise focus (Balkema et al., 2002). Figure 6.3 shows the impact assessment profile for the CC, EP and HT categories per FU (1 m^3 of wastewater) in relation to the standard error of the mean, i.e. the standard deviation of all possible data in relation to the number of iterations of the Monte Carlo analysis. For energy-dependent categories such as CC and HT, the most significative deviations occur at LS (Figure 6.2), this is due to the electricity consumption at this stage, which is higher than in the other reactors. The uncertainty is reduced from approximately 78% in LS to 2% in FS. This is in line with the results of the study presented in the results section. Finally, in the EP category the variation between the different reactors is similar, which is attributed to their greater dependence on the effluent and influent conditions (COD, NT or TP). These parameters are actual measurements and, in this study, show less deviation and more consistency than the electrical process (background process).

There are abundant literature reports on large-scale environmental assessment of WWTP, but little information is available on the environmental and economic analysis of innovative technology under development. This study allows validating the bottom-up techniques strategy in LCA studies, specifically for the analysis of innovative technologies in the field of wastewater treatment and management. Therefore, it is important to know at what point in the development of a technology it makes sense to conduct LCA analyses in order to assess whether the technology is economically and environmentally friendly. In addition, the hotspots of the final environmental impact can be precisely known in the early stages of the technology development, so that operational strategies or design modifications can be introduced at later scales to minimise the final impact.

In short, this chapter sets up the turning point at the scale level from which the decision is made as to whether a technological innovation can be feasible or not and, therefore, continue the bottom-up strategy.

6.4.4. Economic aspects compared to other wastewater treatment technologies

To compare the magnitude of the cost presented by the ELAN® technology, the SCENA system (as an example of innovative technology applied at sidestream conditions) and the CAS process have been considered. For SCENA, the corresponding cost of electricity is 0.52 €/m^3 and it is double for CAS (1.09 €/m^3) (Renzi et al., 2015). However, the cost associated with ELAN® is lower (0.27 €/m^3) than those from SCENA and CAS (Renzi et al., 2015). The SCENA system is more complex than the ELAN® technology as it comprises a fermentation unit, a screw press filter and, finally, a batch sequencing reactor (Frison et al., 2014). In this case, as the sequencing batch reactor is the unit where partial nitrification/denitrification takes place, this reactor was considered in the estimation of costs related energy consumption. An important question is to determine the level of technological development required for the estimation of accurate costs. In this case, the economic data shown in Figure 6.4 are similar for PP2 and FS. The PP1 value remains high compared to PP2 and FS, as it represents approximately 12% of the energy consumption cost. Therefore, an appropriate reactor volume to obtain an economic evaluation in terms of operational costs is approximately 1 m^3.

When it comes to evaluate the economic aspects for only one technology, it makes sense to use electricity-related operating costs for comparison. However, for the different technologies, the implementation costs of one or the other technology are likely to be very different. One of the advantages that ELAN® process stands out from other technologies on the market is that cheaper robust probes are used and the reactor configuration is simpler than other options (Morales et al., 2015b).

6.5. CONCLUSIONS

After applying the LCA methodology to explore the minimal reactor volume which provides a reliable result to evaluate impacts from a technology under development, a volume of 0.2 m^3 was preferred. An environmental assessment can be made when the energy consumption (pumps and aerators) is optimised for the reactor size. This is because in EP, which is the category that does not depend on energy consumption, the impact is practically the same for PP1, PP2 and FS. Therefore, it is possible to make and environmental assessment of the PP1 level. Regarding to the operational cost the appropriate volume for an economic assessment is approximately 1 m^3.

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CHANGING THE PARADIGM OF WASTEWATER TREATMENT

CHAPTER 7: **Water footprint of a decentralised wastewater treatment strategy based on membrane technology**

SUMMARY

The growing pressure on water resources has led to the search for alternatives to the conventional wastewater systems. Within this framework, decentralised systems arise as a good alternative, which comprises collection, treatment and final disposal and/or reuse of water in an area close to the point of origin. Turkey is a country affected by water scarcity; thus, it is essential to improve water recovery through efficient technologies that allow for nutrient recovery and have the potential for water reuse for irrigation to counteract the consumption of drinking water.

In Chapter 7, a decentralised MBR plant in Turkey was evaluated under the most relevant environmental indicators according to the LCA methodology approach: CC and EP categories. In addition, the water scarcity footprint indicator was estimated using the available remaining water method (AWARE). Once the impacts of the plant under study were determined, and, two sensitivity analyses were carried out: i) considering the different solid retention times (SRTs) in the MBR operation, and ii) how the total impact of construction affects the environmental profile in decentralised systems. The sub-processes with the greatest environmental impacts are electricity consumption associated with the operation phase and infrastructure in the construction phase. These impacts are significantly reduced when water is reused in the irrigation of the green areas, approximately 23% in CC, 5% in EP and about 100% in the AWARE category. No significant influence of the SRT variable was observed on the environmental impacts for the range examined, since it only affected the EP category, determining an optimum SRT value of 50 days for the MBR operation.

Andrea Arias[a], Isabel Vallina[a], Yago Lorenzo[a], Okan Tarik Komesli[b], Evina Katsou[c], Gumersindo Feijoo[a] and María Teresa Moreira[a]. 2019. "Water Footprint of a Decentralised Wastewater Treatment Strategy Based on Membrane Technology". Chapter 4. In: Muthu S. (eds) Environmental Water Footprints. Environmental Footprints and Eco-design of Products and Processes. Springer, Singapore. ISBN: 978-981-13-2508-3* [a] Department of Chemical Engineering, University of Santiago de Compostela, Santiago de Compostela, 15782, Spain [b] Department of Environmental Engineering, Ataturk University, Erzurum, Turkey [c] Department of Civil and Environmental Engineering, Brunel University London, Uxbridge, UK

TABLE OF CONTENTS-CHAPTER 7

7.1. INTRODUCTION

Population growth is demanding more resources such as food, water or energy. In contrast, declining rainfall combined with high evaporation contributes to water scarcity. Therefore, integrated water resources management is one of the main aspects from the perspective of the urban, agricultural and industrial water cycle. In this framework, water reuse has emerged as the most viable alternative since reclaimed water reduces the demand for fresh water (Hochstrat et al., 2007).

As mentioned in Chapter 1, the centralised strategy is the most developed alternative for treating wastewater. However, the discharge of the effluent far from the point of origin makes it difficult to reuse it as process or irrigation water (Hophmayer-Tokich, 2006). In addition, high investment costs and the construction of an extensive collection sewerage network can be a problem in poorer areas that do not have the economic resources to meet those costs (Remy and Jekel, 2008). For this reason, in areas characterised by water scarcity or socio-economic instability, decentralised systems emerge as alternatives for recovering water, nutrients or energy. As these systems are more compact, they allow the installation of more advanced technologies, such as MBR that adapt the quality of reclaimed water to local needs (Prieto et al., 2013).

In this context, Mediterranean countries have experienced increasing water scarcity in recent decades. In particular, the annual per capita water availability in Turkey is about one fifth of that of water-rich countries, which is lower than the world average (Yuksel, 2015). Therefore, Turkey is urged to improve water availability considering the foreseen estimations for 2023, when the amount of water will be less than 1,000 m^3/(inhab·year) (Arslan-Alaton et al., 2010). Moreover, the water requirement in Turkey is steadily increasing in the agricultural sector, with a demand representing 74% of the total water consumption (Cakmak et al., 2007). Thus, if there is not a solution to this problem, it is estimated that by 2020, water reserves will not meet the demand of this sector.

Reclaimed water from wastewater may be an option for these water-scarce countries. However, wastewater reclamation can be complex, expensive and resource-intensive due to the need to implement advanced treatment processes to achieve the required effluent quality. In this sense, MBR technology has been widely applied to wastewater treatment because of the advantages it offers, such as its compact and simple design, adequate biomass control, high hydraulic efficiency, as well as the high quality of the effluent that can be used directly in irrigation (Brepols et al., 2008).

In this framework, LCA methodology can help to identify the hot spots of decentralised wastewater treatment schemes not only in terms of operation but also from a construction perspective. Thus, the main objective of Chapter 7 is to study the environmental assessment of a decentralised scheme that evaluates a MBR for the treatment of urban wastewater generated at the METU University Campus in Ankara (Turkey), in order to reuse reclaimed water for irrigation of green areas. Within this framework, special attention was paid to water for irrigation and to the environmental impacts avoided. The environmental impacts of reclaimed water were evaluated using the AWARE method, which will be explained in the materials and methods section.

7.2. MATERIALS AND METHODS

7.2.1. Case of study description and operation of MBR facility

The system includes a submerged rotatory membrane vacuum bioreactor (VRM) designed for a capacity of 2,000 habitant equivalents (Komesli and Gökçay, 2014). The plant consists of two tanks and peripheral equipment (Figure 7.1). The two tanks are separated by a wall, which is turn connects the two tanks through five holes at the bottom of the wall that separate them. The volume of the first tank is 85 m^3. The second anoxic tank (23 m^3) is used to house the membrane unit. The wastewater from the residence halls and the university buildings is collected in a 10 m^3 storage tank and pumped to the treatment plant. At the inlet of the aeration tanks, a screw-type sieve separates wastewater from materials larger than 3 mm (Komesli et al., 2007).

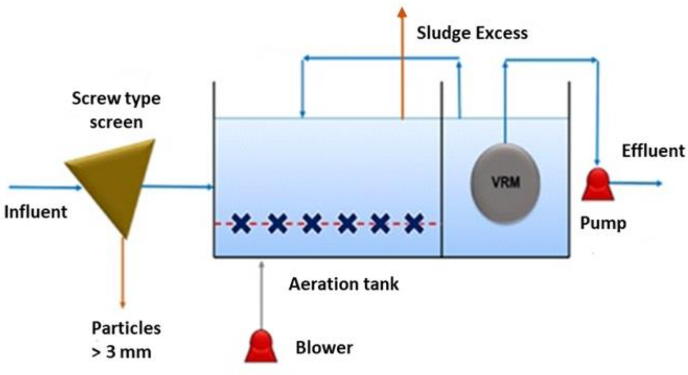

Figure 7.1. Flow diagram of the wastewater treatment plant with a membrane unit

The rotation speed of VRM system is about 2.5 rpm. The treated wastewater is pumped to the membrane modules by means of six radial hoses (Komesli et al., 2015). The membrane unit operates in intermittent vacuum cycles. The aeration tank is equipped with several membrane-types diffusers located at the bottom of the tank. The sludge is partially recirculated from the membrane tanks at a variable frequency to control the concentration of mixed liquor suspended solids (MLSS) (Komesli et al., 2007). The polyethersulfone membrane (PES) modules are flat type with a pore size of 0.038 μm and a total surface area of 540 m^2. There is a membrane support driven by an electric motor that provides the rotation speed to the filter support unit, creating a cross-flow on the surface of the membrane modules along with a coarse aeration from the bottom of the modules (Komesli et al., 2007).

The pumping system operated in periods of 10 min (8 min vacuum and 2 min relaxation). The pump operating regime was changed because the MBR worked equally well in lower and more frequent vacuum periods; therefore, the cycle was changed to 5-min cycles (4 minutes of vacuum and 1 minute of relaxation). During the relaxation time, the vacuum pump stopped but aeration and rotation continued (Komesli et al., 2015). Table 7.1 shows the operating conditions of the reactor during 10 years of operation.

Table 7.1. Operational parameters of the MBR

Operating time (years)	10
Membrane	HUBER, A.G-Flat type
Flow (m^3/d)	108-172.8
Permeate flow ($L/h{\cdot}m^2$)	8.3-13
HRT (h)	15-24
SRT (d)	10-150
TMP (mbar)	-80-(-300)
Organic load ($kg/m^3{\cdot}d$)	0.31-1.53
Nitrogen charge ($kg/m^3{\cdot}d$)	35-70

The maintenance of the membrane unit is carried out with chemical cleaning when the intermembrane spaces are clogged, resulting in decreased permeate flow and increased transmembrane pressure (TMP), especially when the TMP is kept constant and below -300 mbar. The membrane modules are chemically cleaned with 0.5% NaClO for 5-10 h, twice a year. During the 10 years of operation, the membrane was not changed or replaced. However, in the 5th year of operation, the damaged membrane surface was about 50 m^2, but the flow rate increased to reach the same flow rate as in the first years of operation (Komesli et al., 2015). Moreover, a sensitivity analysis with the different SRTs was conducted in this study, which as a parameter that was modified during the operation phase. It is therefore necessary to know whether or not this parameter affects the environmental profile.

7.2.2. Definition of functional unit and system boundaries of the MBR facility

In this study, the FU was defined as 1 m^3 of reclaimed water (Pasqualino et al., 2011) because the water will be used to the irrigation of the green areas in the METE University Campus of Ankara (Turkey). Moreover, on the system boundaries the construction phase of the plant, that it can be an important factor in decentralised systems was included (Lorenzo-Toja et al., 2016b). Thus, in this case, the construction, operation and maintenance phases of MBR, as well as the treatment of sludge and the water reuse for irrigation are included within in system boundaries. Figure 7.2 also shows that during the construction phase, the manufacture of blowers, pumps and pipes is outside the system boundaries due to the lack of data. If considered, this would cause a great deal of uncertainty for the inventory at this stage. Also, within the operational phase of the plant, wastewater collection and screening systems for solids removal were excluded.

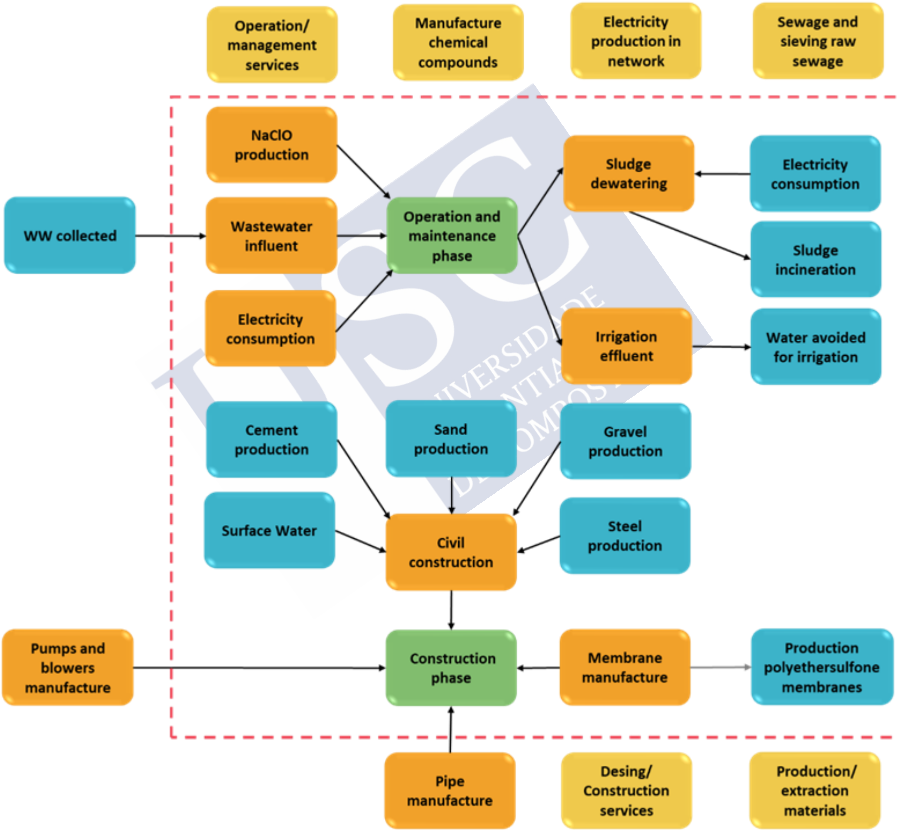

Figure 7.2. System boundaries of the MBR plant

7.2.3. Life cycle inventory and simplifications for the decentralised plant

The primary and secondary data used for this analysis are presented in Table 7.2. Primary data corresponds to the operation of the facility for 10 years. Secondary data were obtained from the Ecoinvent v3.5 database (Wernet et al., 2016). The simplifications considered in this work are presented below.

Trucks: Euro 4 trucks (3.5-7.5 t) were selected as transport vehicles for chemicals and sludge, due to the smaller amount generated in the decentralised systems, in contrast of conventional wastewater treatment plants, between 16 to 32 t (Lorenzo-Toja et al., 2016b).

Chemical consumption: NaClO was obtained from the Ecoinvent v3.5 database (Weidema et al., 2013). The polyelectrolyte doses required for sludge dewatering were those of typical ranges: 5-8 kg polymer/1000 kg dry matter (Tchobanoglous et al., 1998). The amount of sodium hypochlorite was calculated from Komesli et al., (2015). There were two membrane cleaning per year.

Electricity production: the production in Turkey was adapted for 2016. High voltage electricity was converted to medium voltage considering atmospheric and losses in transport (Dones et al., 2007).

Table 7.2. Life cycle inventory of MBR plant per 1 m^3 of treated wastewater

INPUTS		**OUTPUTS**	
From the technosphere		**To the environment**	
Materials and fuel		***Emissions to water***	
Influent		COD (g)	15.23
COD (g)	445.23	NH_4 (g)	18.47
NH_4 (g)	54.59	NO_3 (g)	3.04
NO_3 (g)	0.17	PO_4^{3-} (g)	4.68
PO_4^{3-} (g)	8.24		
Electricity consumption		**To the technosphere**	
Aerator (kWh)	1.40	***Products***	
MBR (kWh)	1.07	Sludge to incineration (g)	840.49
Dehydrator (kWh)	$2.08 \cdot 10^{-3}$		
Chemical consumption			
NaClO (15%) (g)	3.69		
Polyelectrolyte (g)	0.3		
Transport			
NaClO (15%) (kg·km)	$9.23 \cdot 10^{-2}$		
Sludge [b] (kg·km)	45.93		
Construction			
Infrastructure	$7.96 \cdot 10^{-7}$		
PES membrane (g)	0.32		

Moreover, reports and construction projects were retrieved to complete missing data. Data on the construction of the facility and the membrane are presented in Tables 7.3 and 7.4, respectively. The data obtained for the construction of the membrane are all primary data except the thickness (Judd, 2011) that was estimated according to the information provided by the manufacturer (HUBBER). The density of the polymer was taken from the wolfram-alpha database.

Table 7.3. Characteristics of membrane unit

Material	Polyethersulfone (PES)
Surface area (m^2)	540
Pore size (μm)	0.038
Thickness (μm)	300
Material density (kg/m^3)	1470
Membrane weight (kg)	238.14

Table 7.4. Inventory data for wastewater treatment plant construction for 2,000 hab-eq

Material/Construction process	Units	Value
Excavation by hydraulic digger	m^3	5,512
Transport by lorry	t·km	78,070
Transport by train	t·km	92,341
Electricity consumption by construction	kW·h	63
Concrete	m^3	1,584
Reinforcing Steel	kg	122,879
Tap consumption	kg	193,204
Aluminium	kg	1,378
Limestone	kg	33,975
Stainless Steel	kg	9,868
Fiberglass	kg	3,104
Copper	kg	1,457
Synthetic rubber (EPDM)	kg	1,394
Rock wool (insulation material)	kg	1,378
Organic Chemical Compounds	kg	6,415
Bitumen	kg	792
Inorganic Chemical Compounds	kg	792
Low Density Polyethylene (LDPE)	kg	32
High Density Polyethylene (HDPE)	kg	3,865
Polyethylene Terephthalate (PET)	kg	3,896

7.2.4. Assessment methodology and impact categories

The methodologies used for the assessment of life-cycle impacts were the following: Availability Water Remaining (AWARE) for the estimation of the impact of water used for irrigation and water scarcity footprint (Bayart and Ekambi, 2016), CML for calculating the EP category (Guinée, 2002), and finally the IPCC methodology for evaluating the CC category (Stocker et al., 2013), respectively. The SimaPro v9.0 software was used for the impact assessment.

The AWARE method calculates the mid-point indicator of water use, which indicates the available remaining water in a watershed relative to the world average, after meeting the demands of humans and aquatic ecosystems (Frischknecht et al., 2016). To assess potential water deprivation, it is assumed that the less water available per area, the more likely it is that another user will be deprived of this resource (Kounina et al., 2013). The method is based on the different between the availability and demand (1/AMD). When the demand is equal to or great than the availability (negative value of AMD), CF_{AWARE} is set as maximum (Puerto, 2013). The AWARE category is represented by Equations [1] and [2].

$$\mathrm{AMD_i} = \frac{(\mathrm{Availability} - \mathrm{HWC} - \mathrm{EWR})}{\mathrm{Area}} \quad [1]$$

$$\mathrm{CF_{AWARE}} = \frac{\mathrm{AMD_{global\_average}}}{\mathrm{AMD}}, \mathrm{for\ Demand} < Availability \quad [2]$$

The HWC refers to the sum of human water consumption and the EWR refers to environmental water requirements. AMD is the availability minus demand. This methodology is used to determine the water scarcity footprint and to evaluate the water deprivation potential of other users when they consume water in a given geographical area. The first step is to calculate the area, expressed in $m^3/(m^2 \cdot month)$. Secondly, the value is normalised with the world average results (AMD= 0.0136 $m^3/m^2 \cdot month$) and inverted; this value represents the relative value compared to the annual average of water consumption (in m^3). The world average is calculated as a weighted average of consumption. This indicator ranges from 0.1 to 100. Value 1 corresponds to an area with the same amount of available water. In contrast, the value of 10 represents an area where there is 10 times less water (Frischknecht et al., 2016). The inverse (1/AMD) represents the surface-time equivalent for generating one cubic meter of unused water in the region under study (Frischknecht et al., 2016).

Water availability represents renewable water; the values are taken by the WaterGap model that is an average model for a period of 50 years. This model includes human consumption estimated for different sectors such as domestic, agricultural, livestock, among others. Ecosystem demand is evaluated using the variable flow method (VFM). This method classifies the flow regimen as high, medium or low. The annual variability is taken into account to preserve aquatic ecosystems (Schenker et al., 2015).

For the interpretation of the results with respect to the world average, it is important to understand that a characterisation factor of 1 is not equivalent to the factor of average water consumption in the world. That is, the factor that we can use when the location is unknown. This value is calculated as the weighted average of the factor, based on 1/AMD and not on AMD. This implies that consumption has a value of 43 for unknown uses and 20 and 46 for non-agricultural water consumption, respectively (Boulay et al., 2015). The water scarcity footprint can be calculated using AWARE methodology as the product of water consumption and the characterisation factor, as shown in Equation [3].

$$\text{Water scarcity footprint} = \text{water consumption} \cdot \frac{1}{\text{AMD}} = \text{m}^3_{\text{global\_average}} \qquad [3]$$

7.3. Environmental results for the impact categories considered for the decentralised plant

The results are estimated on the basis of the FU: 1 m³ of treated wastewater and are shown in Table 7.5. In the case of CC, the total impact obtained is 1.28 kg CO_2 eq/m³ of treated wastewater, considering that the treated effluent is reused for irrigation of the green areas of the university campus. The results of water reuse lead to an environmental benefit of around 23.8%. However, if the water is not reused for irrigation, the environmental impacts can increase by 2.08 kg CO_2 eq/m^3. Taking into account the sub-processes of the examined system, the electrical consumption of the plant has the greatest environmental impact, with a contribution of 85.6%. The electricity consumption is attributed with the aerators of biological reactor (0.81 kg CO_2 eq/m^3) and the vacuum, aeration and rotation of the membrane bioreactors, which accounted for 0.62 kg CO_2 eq/m^3. These emissions exceed the respective ones associated with the construction phase, and operationally the reuse of water shows a value of -0.39 kg CO_2 eq/m^3 to offset the impact of electricity consumption. The high contribution of energy consumption is related to the fossil CO_2 emissions, with a share of 66.2% in the profile of energy production in Turkey.

The impacts related to the consumption and chemical and transport of chemicals, as well as to the transport of sludge, can be considered non-significant. Chemical consumption (NaClO and polyelectrolyte) represents 0.08% of the total impacts. The transport of sludge and chemicals is also minimal, accounting for 2% of total impacts.

As for the EP category, MBR technology is able to reduce the negative effect on this category by 68% when comparing the effluent between the treated effluent (11.4 g PO_4^{-3}/m^3) or untreated effluent (36.1 g PO_4^{-3}/m^3). The operation (1.76 g PO_4^{-3}/m^3), the construction of the plant (0.33 g PO_4^{-3}/m^3) and sludge incineration (0.21 g PO_4^{-3}/m^3) are the main sub-processes that contribute most to the impact due to the indirect emissions related to energy consumption. In addition, the direct emissions associated to the effluent discharge also contribute to the negative effect. However, if this water is reused, there are environmental credits in this category of about 5% of the total impact. The latter is offset by the impacts generated during the production and transport of chemicals, the manufacture of the membrane and the processes associated with the sludge management, which account for 3% of the total impacts.

Finally, for the AWARE category, as in the other categories studied, the construction of the plant and the consumption of electricity during the operation phase are the stages that imply the highest water consumption and, therefore, those that contribute most to the water scarcity footprint (Table 7.5). The greatest potential for water deprivation for other users derives from the construction process and the energy used to operate the plant. On the other hand, the reuse of water for irrigation resulted in a negative water scarcity footprint. The latter indicates an avoided deprivation of other users of 51.81 m^3 of water per m^3 of treated wastewater. This result is due to the fact that, by reusing water, this amount of water is prevented from being extracted from the water network, thus avoiding the stages of collection, distribution and treatment. If water reuse were not considered, the water scarcity footprint would increase to 7.50 m^3 of water per m^3 of treated wastewater (Table 7.5).

Table 7.5. LCA results for the impact categories under assessment. FU: 1 m^3 of treated wastewater

PROCESS	**IPCC**	**Eutrophication**	**AWARE**
Sub-processes	**kg CO_2 eq/m^3**	**g PO_4^{3-} eq/m^3**	**m^3 world eq**
NaClO (15%)	$5.76 \cdot 10^{-4}$	$1.44 \cdot 10^{-3}$	$5.65 \cdot 10^{-4}$
Polyelectrolyte	$5.01 \cdot 10^{-4}$	$4.56 \cdot 10^{-4}$	$3.07 \cdot 10^{-4}$
NaClO Transport	$4.76 \cdot 10^{-5}$	$4.33 \cdot 10^{-5}$	$5.84 \cdot 10^{-6}$
Sludge Transport	$2.37 \cdot 10^{-2}$	$2.15 \cdot 10^{-2}$	$2.90 \cdot 10^{-3}$
Plant Infrastructure	0.18	0.33	1.56
Membrane construction	$3.01 \cdot 10^{-3}$	$4.67 \cdot 10^{-3}$	$4.09 \cdot 10^{-3}$
Aeration electricity	0.81	1.00	1.36
VRM electricity	0.62	0.76	1.04
Centrifuge electricity	$1.20 \cdot 10^{-3}$	$1.48 \cdot 10^{-3}$	$2.03 \cdot 10^{-3}$
Irrigation	-0.39	-0.69	-55.81
Incineration	$3.35 \cdot 10^{-2}$	0.21	$2.08 \cdot 10^{-2}$
Influent	-	11.41	-
Effluent	-	36.07	-
Reuse	1.28	13.77	-51.81
No reuse	2.08	14.46	7.51

7.4 DISCUSSION

7.4.1. Trade-off analysis of the climate change and eutrophication impact categories

In this section, despite their major relevance in environmental awareness, the CC and EP categories were studied in greater detail. In the MBR unit, the impacts are caused by the energy consumption associated with the operational phase, in agreement with other studies (Hospido et al., 2012; Ioannou-Ttofa et al., 2016), who reported that the energy consumption of the plant is responsible for more than 95% of the impacts. The construction phase accounts for 28.7%, which is dependent on the use of concrete and steel for infrastructure, acquiring an importance that has been traditionally considered non-significant.

Concerning EP category, the NEB indicator (Godin et al., 2012) was calculated to analyse different MBRs unit and also to compare them with conventional systems. This indicator was explained in Chapter 4. The main results are shown in Figure 7.3. The MBR facility studied in this case has an average performance similar to that of a conventional WWTP (Lorenzo-Toja et al., 2016b). In addition, in comparison with other studies on MBR units, the value of the NEB varies according to the configuration (Hospido et al., 2012). The NEB value of the MBR facility under study is similar to that of a UASB reactor followed by a hybrid reactor. The membrane is located in a separate chamber before the effluent is discharged. The impacts associated with the treated effluent from an MBR vary from 13-20 g PO_4^{-3}/m^3. The results obtained in this study are within this range; thus, it can be concluded that MBR efficiency was maintained or even improved in the scale-up of the system.

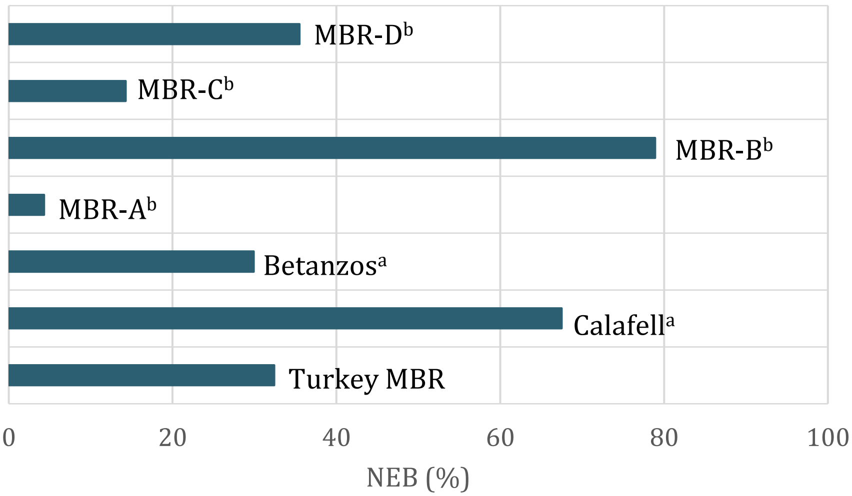

Figure 7.3. Net environmental benefit comparison between different plants. Note [a] Lorenzo-Toja et al., (2016) [b] Hospido et al., (2012)

7.4.2. The importance of water reuse: giving the floor to the AWARE category

This section will assess the water scarcity footprint, i.e., the potential for water deprivation for other users, regardless of the type of user, in a given geographical area: Turkey. To date, no work has been published that applies the AWARE methodology to assess the water scarcity footprint of wastewater treatment systems, only in the field of food production (Bayart and Ekambi, 2016; Schenker et al., 2015). Several methodologies have been applied to determine the water footprint, such as ReCiPE, however, they have limitations. Opher and Friedler, (2016) analysed the impact of the water depletion in decentralised and centralised systems. The results did not follow a general trend. This is because systems that convert seawater into drinking water have an environmental benefit in this category. The seawater is considered an infinite source of water and therefore its consumption has no impact. Centralised systems with water reuse, because they consider the consumption of drinking water to have an impact on the category of water depletion.

Morera et al., (2016) calculated the water footprint of the Garriga WWTP. The methodology used was the Water Footprint Network (WFN) methodology, which consists of classifying water intro three types: blue water footprint related to the water that evaporates during the operational phases of the WWTP, grey water footprint associated with the concentration of the effluent, and finally, the green water footprint is the water evaporated by vegetation. In this case of WWTPs, green water is not considered. This study concludes that the water footprint is reduced with secondary treatment, so there is a decrease in the grey water footprint when treating the wastewater. In this study, this methodology is so inappropriate that water scarcity is not considered. Therefore, this methodology was not considered.

7.4.3. Studying the influence of the SRT and the construction phase on the environmental outcomes

A sensitivity analysis was conducted to identify the effect of SRT (10 to 140 days) on the CC, EP and AWARE categories. Moreover, a comparative analysis was performed including and excluding the construction phase.

Influence of SRT on CC category

As mentioned before, the main contributors to the impact in CC category are the construction phases and the energy consumption of the decentralised plant. Figure 7.4a shows that while the SRT increases, energy consumption decreases, resulting in less impact on this category. The energy consumption has a deviation of ± 0.1 kWh/m^3, so the observed decrease is not relevant, reflecting that this variable does not depend on the SRT as long as the operation of the reactor is feasible and efficient. On the other hand, the higher the SRT, the greater the amount of excess sludge, and as a consequence, the impacts due to the transport of sludge and the incineration process increase. Furthermore, Figure 7.4b compares the overall impacts of all the scenarios examined (with and without reuse of treated effluent) for the different sludge retention times. The impact of not reusing the water is lower at higher SRTs, (2.13 kg CO_2eq/m^3 to 1.87 kg CO_2eq/m^3). This decrease is due to lower energy consumption during the operation. However, the decreases are not relevant due to the standard deviation of energy consumption, which indicates the independence of the SRT.

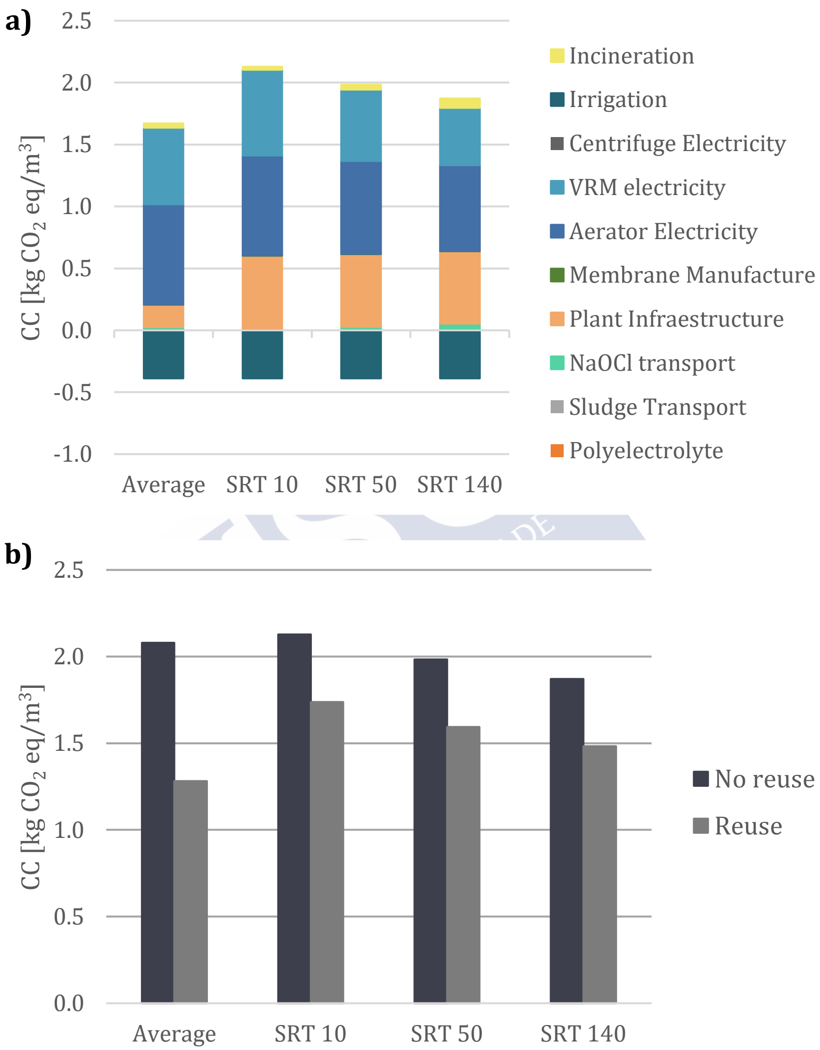

Figure 7.4. Comparison of the impacts associated with the CC category for different operating SRTs: (a) different sub-systems of the plant (b) without or with water reuse

Influence of the SRT on the eutrophication category

The treated effluent is the main contributor to the impacts in the EP category due to the OM, nitrogen and phosphorus content. Figure 7.5a shows the variation of EP impacts as a function of SRT. The MBR can reduce the impact of this category by approximately 52-81%. The change at different SRT is that at low (i.e. 10 days), nitrification is not effective. The optimum reactor performance was achieved at a SRT of 50 days. However, when the SRT is high (140 days), excessive biomass accumulation arises as a major operational problem. Moreover, in terms of effluent impacts, the water effluent is similar to those of SRT of 40 days and 140 days: 33% (Figure 7.5a). The minimum impacts for the EP category are when the SRT is 50 days, the negative effects represent about 12%. Therefore, the optimal selection of SRT is based on the pollutant load of the effluent, but not exclusively on it. There are other important factors such as dissolved oxygen due to an inefficient operation of the aerators or an inadequate equipment sizing.

In the eutrophication is analysed by sub-process (Figure 7.5b), the electricity consumption decreases by applying a SRT of 140 days; however, this reduction is not significant. Therefore, the sub-processes are independent of the SRT.

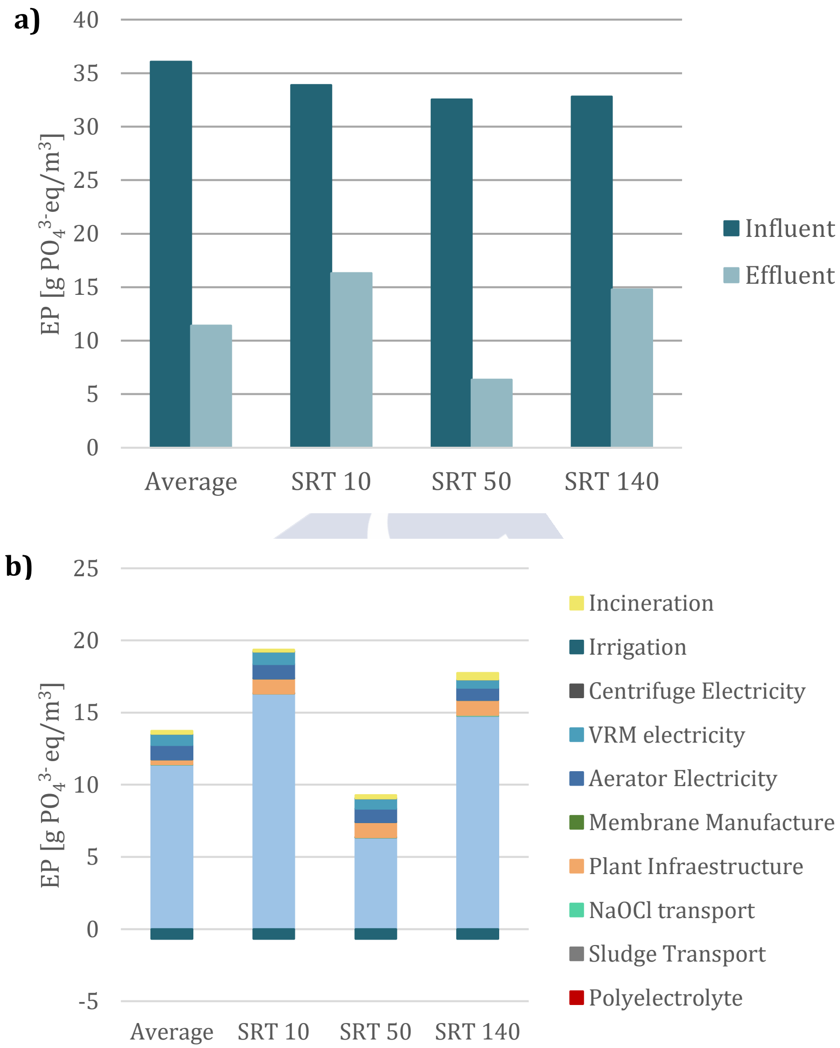

Figure 7.5. Comparison of the impact associated with the eutrophication potential for different operating SRTs: (a) without and with water reuse (b) different sub-systems of the plant

Influence of SRT on the AWARE category

The AWARE category allows the assessment of water scarcity footprint. The reuse of irrigation water remains constant and independent of the solid retention time. Therefore, the reduction in energy consumption affects the category of water use for all SRTs examined (Figure 7.6). In this case, the category is independent of the different SRTs, as observed in the CC category.

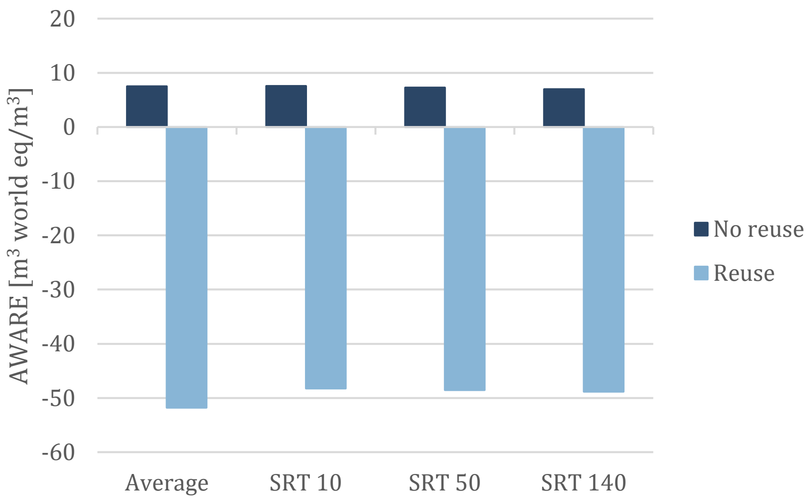

Figure 7.6. Comparison of the AWARE category according to different scenarios of water reuse

Influence of the construction phase on the impact categories

The construction phase has an impact on all categories; therefore, an environmental analysis has been carried out without the construction phase. The results of the different categories are shown in Table 7.6. The impact of the CC category decreases by 14% when the construction phase is not taken into account. Moreover, the water reuse to irrigate green areas offsets the emissions caused by electricity consumption by approximately 28%. The transport of chemicals and sludge has an impact of 4%. The overall reduction in impacts is attributed to the contribution of emissions associated with the construction of tanks for the storage (irrigation water storage) of the treated effluent. In the EP category, the effluent remains the main source of impact (85%). The electricity consumption in the operation phase and sludge incineration are responsible for 15% of the impact. Water reuse offsets the impact in this category by 5%.

The evaluation of the impact associated with water consumption, neglecting the construction phase, led a considerable increase in the benefits of reuse. Conversely, if water is not reused to irrigate green areas, the potential for water deprivation by other global users will increase.

Table 7.6. LCA results without construction phase for the impact categories under assessment

PROCESS	IPCC	EP	AWARE
Sub-processes	**kg CO_2 eq/m^3**	**g PO_4^{3-} eq/m^3**	**m^3 world eq**
NaClO (15%)	$5.76 \cdot 10^{-4}$	$1.44 \cdot 10^{-3}$	$5.65 \cdot 10^{-4}$
Polyelectrolyte	$5.01 \cdot 10^{-4}$	$4.56 \cdot 10^{-4}$	$3.07 \cdot 10^{-4}$
NaClO Transport	$4.76 \cdot 10^{-5}$	$4.33 \cdot 10^{-5}$	$5.84 \cdot 10^{-6}$
Sludge Transport	$2.37 \cdot 10^{-2}$	$2.15 \cdot 10^{-2}$	$2.90 \cdot 10^{-3}$
Membrane Construction	$3.01 \cdot 10^{-3}$	$4.67 \cdot 10^{-3}$	$4.09 \cdot 10^{-3}$
Aeration electricity	0.81	1.00	1.36
VRM electricity	0.62	0.76	1.04
Centrifuge electricity	$1.20 \cdot 10^{-3}$	$1.48 \cdot 10^{-3}$	$2.03 \cdot 10^{-3}$
Irrigation	-0.39	-0.69	-55.81
Incineration	$3.35 \cdot 10^{-2}$	0.21	$2.08 \cdot 10^{-2}$
Influent	-	11.41	-
Effluent	-	36.07	-
Reuse	1.10	12.77	-53.38
No reuse	1.40	13.46	2.43

7.5. CONCLUSIONS

The construction phase of a decentralised plant applying MBR technology plays an important role in terms of the impacts associated with the target categories of CC, EP and AWARE. In terms of the operational phase, the energy consumption in the plant is the main hot spot of the decentralised systems. However, the reuse of treated water significantly improves the environmental profile in all categories. This allows the water potential of a country at risk of water scarcity to improve.

Sensitivity analysis for the operation of the system at different SRTs showed that this parameter is independent in the different sub-systems. The EP category is the only one that depends on the nutrient content of the effluent and is, therefore, affected by the SRT. The optimum SRT of 50 days will imply a reduction of eutrophication impacts.

Finally, decentralised systems can be a good alternative to reduce environmental impacts when resources are recovered because they are more flexible, and it is easier to adapt these systems especially in countries where there are real problems of water scarcity.

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CHAPTER 8: **Environmental analysis of servicing centralised and decentralised wastewater treatment for population living in neighbourhoods**

SUMMARY

The planning and construction of large-scale wastewater infrastructure, such as sewerage networks and WWTPs, is undertaken by the public sector or by publicly regulated monopolies. Within the framework of water cycle management, there is an increasing movement of the population towards the cities where economic activity is concentrated. This scenario is particularly pronounced in certain regions of the world and makes it necessary to rethink whether decentralised treatment offers an alternative for ensuring the servicing of wastewater treatment in new urban developments.

In Chapter 8, four systems were evaluated: two centralised and two decentralised configurations, from an environmental and economic perspective, posing as working hypothesis how different wastewater treatment schemes influence of the carbon footprint of the population living in a neighbourhood. Decentralised systems present a reduction in the carbon footprint of residents of around 20-23% depending on the scheme. Although decentralised systems have higher construction costs, they can be amortised due to lower energy consumption. Considering the problems associated with the changing and replacing existing networks, decentralised wastewater treatment schemes is especially recommended for new dwelling developments, based on its environmental and economic indicators.

Andrea Arias[a], Manuel Rama[a], Sara González-García[a], Gumersindo Feijoo[a], María Teresa Moreira[a].

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a CRETUS Institute. Department of Chemical Engineering, Universidade de Santiago de Compostela, E-15782, Santiago de Compostela, Galicia, Spain

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8.1. INTRODUCTION

Since 1950 the urban population has grown exponentially (Steffen et al., 2015). Currently, more than 55% of the world population lives in cities, and this figure is expected to increase to 60% by 2030 (United Nations, 2015). This means that, from an economic perspective, cities will concentrate more than 80% of the global GPD (World Bank, 2019), although their occupation is only 3% of the world´s land area. It is estimated that human activities, such as transport, food and energy consumed at the households are responsible for about 70% of GHG emissions (Goldstein et al., 2013; González-García and Dias, 2019; Kennedy et al., 2009). Given that the population is concentrated in cities, it is necessary to assess the strategies followed to reduce the environmental impacts of residents living in these areas (Lahmouri et al., 2019).

In this regard, WWTPs have focused on trying to reduce their impact to become carbon neutral (Shen et al., 2015). Clearly, the centralised and decentralised options for wastewater treatment present large differences in the process scheme, the equipment deployed and chemicals or energy requirements. Beyond these characteristics, another difference between the two alternatives is attributed to the possibility of reusing resources. While in centralised systems, resource recovery is hampered by the transport and distribution of both streams, decentralised treatment offers the possibility of reusing reclaimed water and biofertilisers in nearby green and agricultural areas (Samuel et al., 2016). While there are indicators of environmental benefits associated with the decentralised treatment, it is important to quantify the environmental credits of this approach and compare it to a conventional centralised system.

In this framework, there are several studies that compare the differences in GHG emissions using the LCA methodology (Kobayashi et al., 2020; Lahmouri et al., 2019). Moreover, LCA also used to compare the environmental impacts associated with the wastewater treatment facilities were evaluated, the environmental benefits that one system or another has on the impacts of dwellers living in a neighbourhood.

For this reason, the main objective of this study is to analyse how conventional or decentralised systems can reduce the total impact of a resident who chooses to live in a building or residential area with centralised or decentralised treatment. The first centralised option is the real case in Santiago de Compostela, whereas the other configurations were analysed based on plant-wide simulations. The second option considers a modification of existing centralised scheme, but with the incorporation of AD unit for biogas recovery. The decentralised configurations considered in this study consider the segregation of black and grey water and the use of two types of toilets: conventional or vacuum toilets. Both configurations consist of an UASB for energy recovery and an aerobic membrane for the treatment of black water. The grey water is treated in an SBR. Not only the environmental but also the economic indicators of the different alternatives will be evaluated, which will make it possible to rank the different options under a combined sustainability perspective.

8.2. MATERIALS AND METHODS

8.2.1. Goal and scope

The main objective of this study is to benchmark the environmental and economic profile of a resident living in a neighbourhood with centralised and decentralised wastewater treatment system according to four different schemes (detailed below in Figures 8.1 and 8.2). The centralised scheme (actual scheme in Santiago de Compostela) consists of a WWTP designed for 220,000 equivalent inhabitants with a flow of 75,000 m^3/d. The WWTP consists of a pre-treatment unit followed by a coagulation and flocculation process and AS process. The treated wastewater is discharged into the aquatic environment. The sludge line comprises a thickening and homogenisation units followed by a dewatering unit. The sludge is treated in a composting unit and then applied in agriculture (Scenario 1 in Figure 8.1b). A modification including the AD unit in the sludge line and a CHP unit for bioenergy production corresponds to Scenario 2 of Figure 8.1c.

Two decentralised schemes with the segregation of streams: black and grey water were designed to treat a flow of about 800 m^3/d for 7120 inhabitants (Figure 8.2). The BW, coming from the toilets, is treated in a UASB unit followed to an MBR. The GW from the showers or washing machines is treated in an SBR. Reclaimed BW and GW streams are used for irrigation. Coupled to the UASB unit, a CHP will transform the biogas into energy and heat. This energy will be used in the houses, whereas the heat will be used to maintain the UASB unit at 35 °C, and in case of heat excess, it will be used in the houses. The main difference between the two decentralised systems are the type of toilets. In the first case, toilets are considered as conventional toilets (Scenario 3), while in the second case, vacuum toilets are implemented as an option (Scenario 4). Vacuum toilets consume less water and this implies a higher concentration of organic matter to increase the biogas yield (Kujawa-Roeleveld et al., 2006).

To apply the LCA methodology, a gate-to-gate perspective is selected. The FU was selected as 1 resident that generates 0.125 m^3 of water per day, which corresponds to the amount of wastewater generated by one person in one day (Wan et al., 2016).

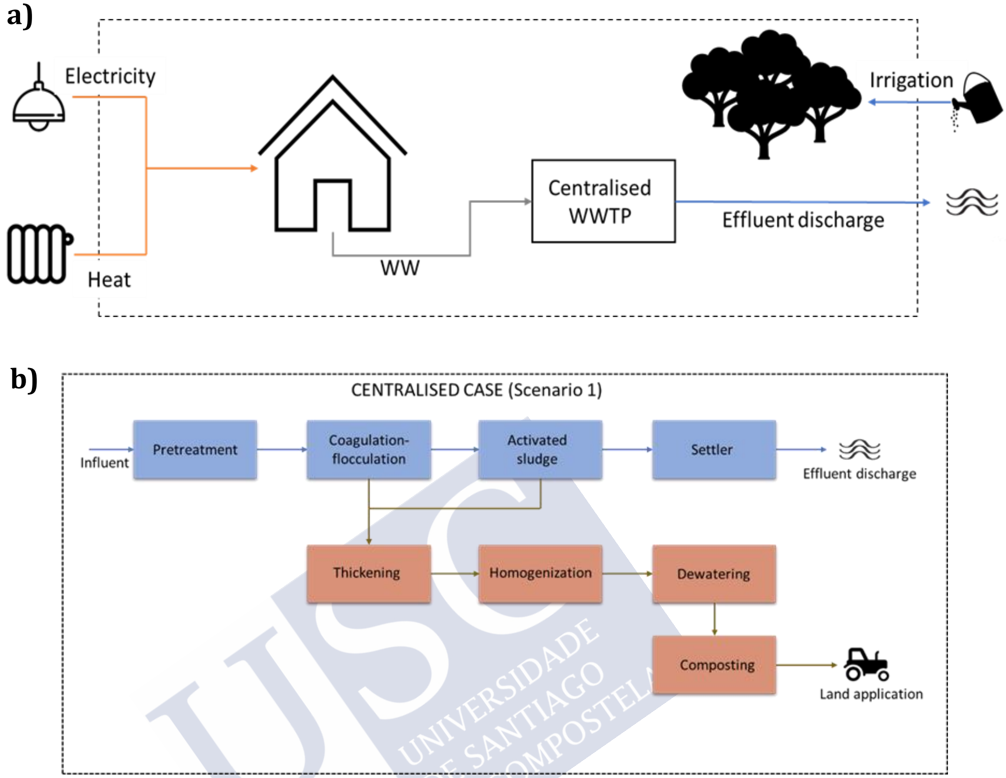


Figure 8.1. Centralised neighbourhood scheme: a) general configuration; b) system boundaries for Scenario 1 (WWTP of Santiago de Compostela); c) System boundaries for Scenario 2 (centralised case with AD unit)

c)

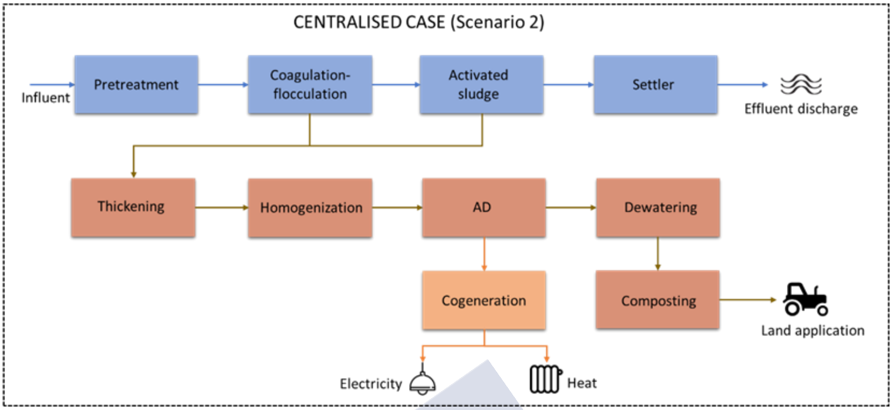


Figure 8.1. Centralised neighbourhood scheme: a) general configuration; b) system boundaries for Scenario 1 (WWTP of Santiago de Compostela); c) System boundaries for Scenario 2 (centralised case with AD unit)

a)

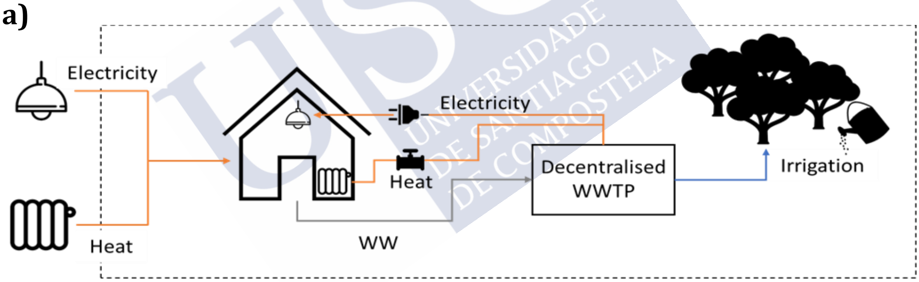
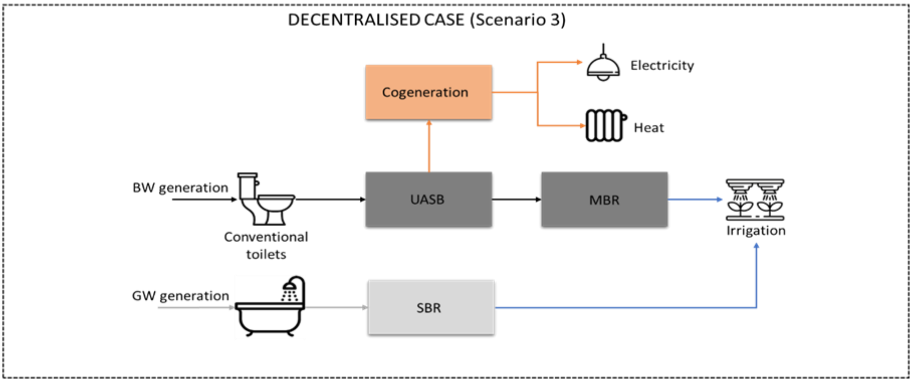


Figure 8.2. Decentralised neighbourhood scheme: a) general configuration; b) System boundaries for Scenario 3 (decentralised system with conventional toilets); c) System boundaries for Scenario 4 (decentralised system with vacuum toilets)

b)



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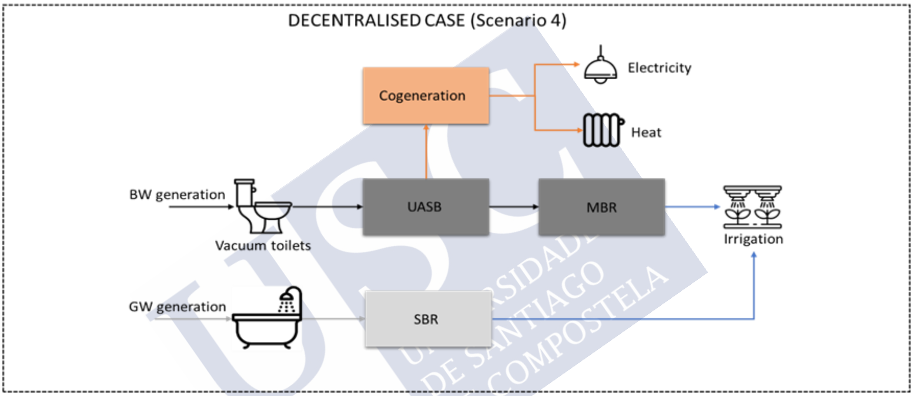

Figure 8.2. Decentralised neighbourhood scheme: a) general configuration; b) System boundaries for Scenario 3 (decentralised system with conventional toilets); c) System boundaries for Scenario 4 (decentralised system with vacuum toilets)

8.2.2. Life cycle inventory (LCI) for the different wastewater treatment configurations

The inventories were made with primary data (real data) and secondary data (calculated or bibliographic data), reported in Tables 8.1 and 8.2. The primary data correspond with the real data which are associated with the centralised case. The characteristics of the wastewater, the amount of sludge generated and the consumption of chemicals were obtained from an internal report (PRTR, 2017). Moreover, electricity consumption and biogas production (Scenario 2) were obtained using estimated data. In the decentralised cases, the data are obtained through bibliographic information and mass balances. BW, GW, biogas transformation or energy consumption are bibliographical data (Komesli et al., 2007; Zang et al., 2015; Zeeman et al., 2008). Therefore, the inventories were completed with the Ecoinvent v3.5 database (Wernet et al., 2016). Finally, several simplifications were made for background data.

Electricity: Spanish electricity country mix was updated for the 2018 year with the data form the annual report (REE, 2018). As regards the consumption of chemical products, polyelectrolyte was implemented as cationic resin taking into account the Ecoinvent v3.5 database (Wernet et al., 2016). Biogas composition was considered such as 75% CH_4, 24% CO_2 and 1% H_2S (Kujawa-Roeleveld et al., 2006). Finally, the emissions of composting to air (CH_4, CO_2, N_2O and NH_3) were estimated through bibliographic data (Boldrin et al., 2009).

Table 8.1. Main inputs to the different scenarios considered in this study. FU: 1 resident. S1: Scenario 1, S2: Scenario 2, S3: Scenario 3, and S4: Scenario 4

		S1	S2	S3	S4
Inputs from the technosphere					
Materials and fuel					
Influent					
COD (g)	BW (g)	42.35	42.35	101.3	101.3
	GW (g)			53.13	53.13
TN (g)	BW (g)	1.31	1.31	175	175
	GW (g)			2.15	2.15
TP (g)	BW (g)	0.51	0.51	21.87	21.87
	GW (g)			0.72	0.72
Electricity consumption					
Pre-treatment (kWh)		$1.57\cdot10^{-3}$	$1.57\cdot10^{-3}$	-	-
Coagulation-flocculation (kWh)		$5.15\cdot10^{-3}$	$5.15\cdot10^{-3}$	-	-
CAS (kWh)		$4.03\cdot10^{-2}$	$4.03\cdot10^{-2}$	-	-
Thickening + homogenization (kWh)		$2.27\cdot10^{-4}$	$2.27\cdot10^{-4}$	-	-
AD (kWh)		-	$2.91\cdot10^{-3}$	-	-
Dewatering (kWh)		$2.27\cdot10^{-3}$	$2.27\cdot10^{-3}$	-	-
Composting (kWh)		$2.40\cdot10^{-4}$	$2.40\cdot10^{-4}$	-	-
Toilets (kWh)		-	-	-	0.06
UASB (kWh)		-	-	$3.76\cdot10^{-3}$	0.01
MBR (kWh)		-	-	0.15	0.15
SBR (kWh)		-	-	0.10	0.10

Table 8.1.(cont.). Main inputs to the different scenarios considered in this study. FU: 1 resident. S1: Scenario 1, S2: Scenario 2, S3: Scenario 3, and S4: Scenario 4

	S1	**S2**	**S3**	**S4**
Inputs from the technosphere				
Chemical consumption		-		
Coagulation-flocculation				
$FeCl_3$ (kg)	$1.93\cdot10^{-3}$	$1.93\cdot10^{-3}$	-	-
Thickening + homogenization				
Polyelectrolyte (kg)	$4.51\cdot10^{-2}$	$4.51\cdot10^{-2}$	-	-
Transport				
$FeCl_3$ (kg·km)	0.05	0.05	-	-
Polyelectrolyte (kg·km)	1.13	1.13	-	-
Sludge (kg·km)	0.96	0.96	-	-
Land application				
Agricultural machinery (kg)	0.04	0.04	-	-

Table 8.2. Main outputs to the different scenarios considered in this study. FU: 1 resident

	S1	S2	S3	S4
Outputs to the environment				
Emissions to air				
AD unit				
CH_4 (kg)	-	$4.67 \cdot 10^{-4}$	$1.06 \cdot 10^{-2}$	$1.26 \cdot 10^{-2}$
CO_2 (kg)	-	$5.48 \cdot 10^{-4}$	$1.05 \cdot 10^{-2}$	$1.25 \cdot 10^{-2}$
H_2S (kg)	-	$9.67 \cdot 10^{-6}$	$2.20 \cdot 10^{-4}$	$2.60 \cdot 10^{-4}$
Composting unit				
CH_4 (mg)	$4.68 \cdot 10^{-4}$	$4.68 \cdot 10^{-4}$	-	-
CO_2 (mg)	0.13	0.13	-	-
N_2O (mg)	$2.25 \cdot 10^{-4}$	$2.25 \cdot 10^{-4}$	-	-
NH_3 (mg)	$1.87 \cdot 10^{-2}$	$1.87 \cdot 10^{-2}$	-	-
Land application				
N_2O (kg)	$1.33 \cdot 10^{-3}$	$1.33 \cdot 10^{-3}$	-	-
NH_3 (kg)	$1.09 \cdot 10^{-3}$	$1.09 \cdot 10^{-3}$	-	-
Emissions to water				
NO^-_3 (kg)	0.02	0.02	-	-
PO_4^{-3} (kg)	$1.72 \cdot 10^{-3}$	$1.72 \cdot 10^{-3}$	-	-
Outputs to the technosphere				
Cogeneration unit				
Electricity production (kWh)	-	0.01	0.28	0.33
Heat production (kWh)	-	0.01	0.25	0.29
Water for irrigation (m^3)	-	-	0.01	0.01

8.2.3. Indicators for evaluating environmental and economic profile

The inventory data were implemented in the SimaPro 9.0 software to obtain the most representative impacts to the different configurations. In this case, the most representative categories are CC due to electricity production and consumption that can affect the reduction or increase of the resident carbon footprint. The other relevant category in this study is WC. As mentioned above, water is used for irrigation. In the centralised case, this water comes from the tap water network, so this tap water has environmental impacts, while in the decentralised case, the water comes from the WWTP. Therefore, it is important to know how this change affects the environmental impacts associated with the residents living in the neighbourhood. These two impact categories were calculated using the ReCiPe Midpoint (H) method (Huijbregts et al., 2017). In addition, for the CC category, the environmental impacts are transformed into environmental prices to obtain how much it would cost to implement these systems from an environmental perspective. The main reason for calculating only the environmental prices in the CC category is because of WC category does not have characterisation factor in this methodology (De Bruyn et al., 2018).

Operational and capital costs were calculated as direct economic indicators, whereas environmental prices were quantified such as indirect economic indicators. Operating costs were associated with sludge management, chemical and energy consumption, while capital costs considered only the construction of the unit and incorporated into the total value of WWTP.

8.3. RESULTS AND DISCUSSION

8.3.1. Carbon footprint for each resident according to the different wastewater scheme

Environmental impacts were only assessed for the CC and WC categories. In this first section, environmental impacts will be studied for the CC category. For this reason, it is important to know how much energy and water is consumed per resident in each house. In Spain, energy and heat that is consumed per inhabitant is about 1.581 kWh and 425 kWh in a year (IGE, 2016). The biogas produced in the anaerobic digestion units can be used to supply energy and heat to the houses. In this way, the reduction of the carbon footprint per resident can be estimated.

The results of the resident´s carbon footprint, depending on the wastewater treatment configuration (centralised or decentralised) are shown in Figure 8.3. The worst values in terms of heat and electricity are presented for Scenario 1 (centralised case) because there is no AD unit; therefore, there is no generation of these products. Furthermore, when the AD unit is incorporated into the centralised system, the carbon footprint can only be reduced by about 4% in terms of heat and energy. Decentralised cases show better results in reducing the carbon footprint because the production of energy and heat is higher than in centralised systems. The best case is when vacuum toilets are incorporated (Scenario 4) and the reduction is 23% for electricity consumption and 66% for heat production. In Scenario 3 (conventional toilets), the increase is also significant, about 20% for energy and 54% for heat. Thus, these decentralised systems help to decrease the carbon footprint of a resident living in a decentralised wastewater treatment system.

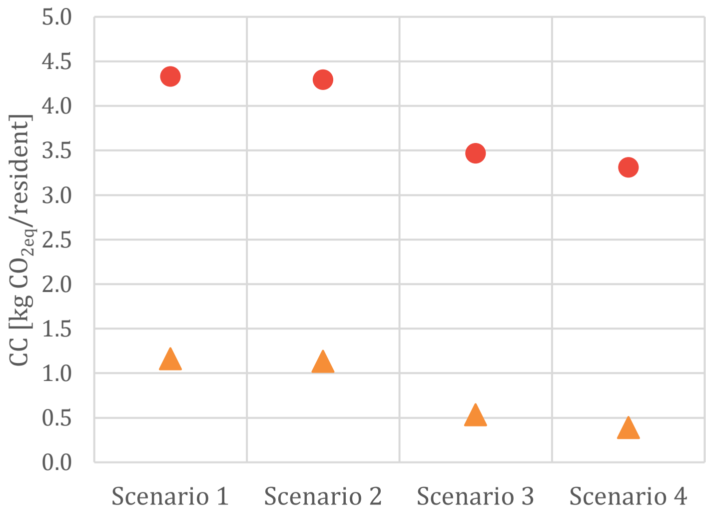

Figure 8.3. Carbon footprint in terms of energy and heat for a resident that lives in a centralised or decentralised wastewater scheme. Scenario 1: conventional system, Scenario 2: conventional system with AD unit; Scenario 3: decentralised system with conventional toilets; Scenario 4: decentralised system with vacuum toilets

In the context of reducing the carbon footprint, it is also important to study the environmental impacts in the CC category for each wastewater treatment scheme. The different wastewater treatment schemes were compared and, in addition, the main impacts for each scheme were analysed. As in the previous analysis, the centralised cases are the worst options because there is no electricity production (Figure 8.4a). In addition, in terms of CC impacts, Scenario 3 (conventional toilets) is the best scenario, even better than Scenario 4 (vacuum toilets). Energy production is higher in Scenario 4 (about 16% than in Scenario 3), however, the energy consumption of the vacuum toilets implies undesirable impacts. Although, the energy consumption is higher than in Scenario 4, the impacts are better than in the conventional systems.

If the subsystems of each system in this category are studied, the main impact is the thickening + homogenisation followed by the CAS unit in centralised systems. In the first unit, the impact is associated with the consumption of polyelectrolyte to ensure good sludge dewatering, while in the CAS unit, the negative effect is related to the consumption of energy for aeration. Moreover, the AD incorporation in Scenario 2 does not represent a significant increase in the impact. In decentralised systems, MBR followed by the SBR represent the worst environmental profile due to the energy consumption associated with these units. In addition, the vacuum toilets also have a negative effect of about 10% of the total impact. However, electricity production minimises the total impact of these systems with environmental credits of around 50% in both systems (Figure 8.4b).

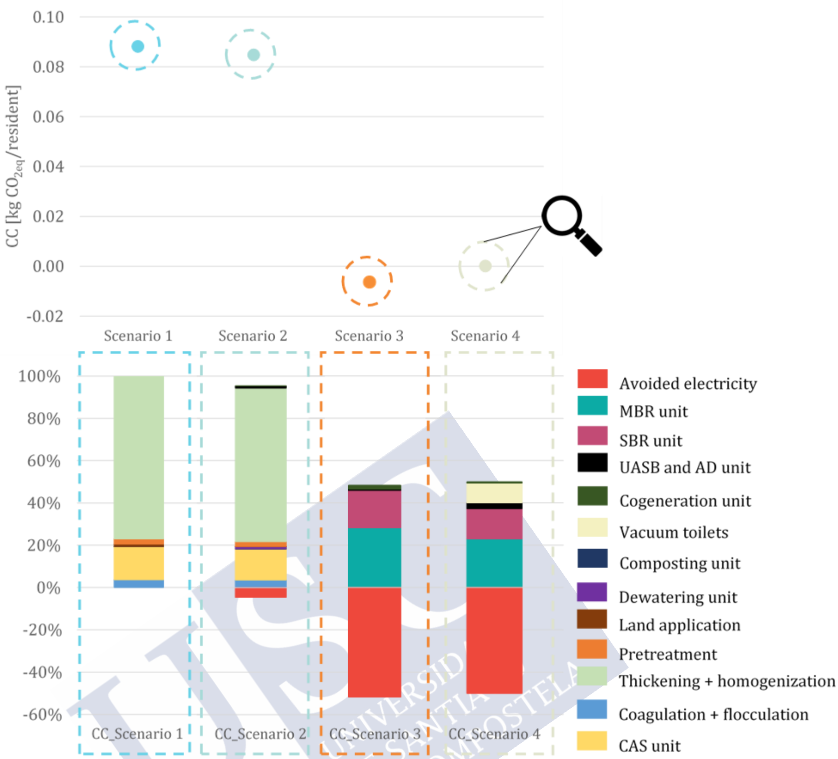

Figure 8.4. Environmental impacts for CC category for each resident and environmental impacts for each sub-system that conforms the different wastewater treatment schemes. Scenario 1: conventional system; Scenario 2: conventional system with AD unit; Scenario 3: decentralised system with conventional toilets; Scenario 4: decentralised system with vacuum toilets

8.3.2. Water consumption and reduction for the different wastewater treatment schemes

In this section, the reduction in the water consumption was evaluated according to the different wastewater treatment configurations. In Santiago de Compostela, the water used for irrigation is 11.10 m^3/inhabitant·year (IGE, 2016). Thus, as in the CC category, the water necessary for irrigation was compared among the different wastewater configurations per inhabitant, in addition, the WC category and the sub-systems affecting to this category were analysed and compared.

The total water used in irrigation in this city is 6663 m^3/d. This number includes the irrigation of parks, green areas and the provision of water for the fire stations. The neighbourhood studied requires 216 m^3/d of irrigation water. In both centralised systems, this number does not decrease because the water in these systems is discharged into the environment. However, in the decentralised cases, water is reused for irrigation. The wastewater generated in Scenario 3 (conventional toilets) is about 788 m^3/d because these toilets consume more water than vacuum toilets (Scenario 4), in which the wastewater flow is about 663 m^3/d. This means 100% savings in both systems. Therefore, the environmental impacts of tap water treatment would be avoided. For the irrigation of green areas only 216 m^3/d of water is required, this means that there is an excess of water of about 572 m^3/d (conventional toilets) and 418 m^3/d (vacuum toilets) that could be used in other situations, for example, cargo trucks for street cleaning, firefighting, among others.

Thus, in the case of decentralised systems, it is not necessary to purify the tap water for irrigation, which means that only the impacts of the irrigation process itself will be taken into account. However, in centralised cases there is no water recovery, so in Scenarios 1 and 2 the impacts of irrigation are associated with the treatment of drinking water.

If the irrigation process is analysed for the different scenarios, the environmental results for the WC category show that in the case of centralised systems the impact values are 0.12 m^3 of water per resident, while for decentralised cases the negative effect is about $3.03 \cdot 10^{-3}$ m^3 of water per resident. These results show an improvement of around 99% in the environmental profile because the production and distribution of tap water that is caused by the centralised configurations involve large environmental impacts.

Figure 8.5 shows the main results for the WC category for each wastewater treatment scheme considered. In addition, the subsystems that contribute to these impacts are studied. As in the CC category, the best environmental results (including environmental credits) correspond to the decentralised schemes due to energy and water recovery. The conventional toilet scenario (Scenario 3) has a better environmental profile because there is less energy consumption and the energy consumption related to hydropower plants indirectly affects this category. For this reason, the conventional case with an AD unit has better environmental profile than the conventional case, although the impacts are very similar (2% reduction compared to Scenario 1). The main impact of the centralised systems occurs in the thickening + homogenisation unit and is caused by the consumption of polyelectrolyte. As in the CC category, in decentralised systems, the negative effect is associated with the MBR unit followed by the SBR unit. This is due to the high energy of these units compared to the others included in the wastewater scheme. The main reason for these impacts is the indirect emissions associated with hydropower production. In Spain, this energy represents around 15% of the total energy country mix (REE, 2018) (Figure 8.5).

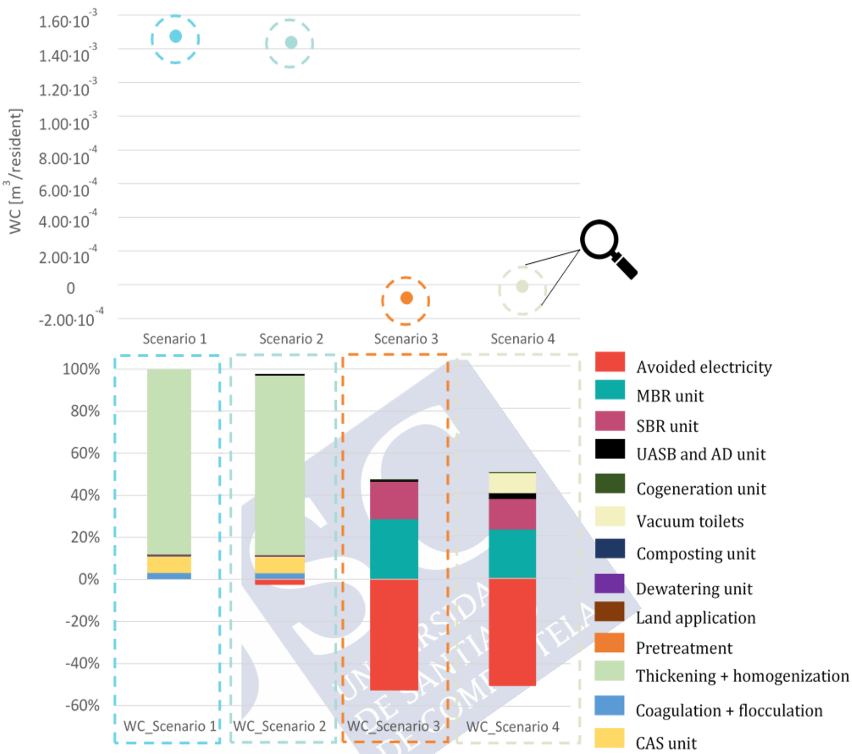

Figure 8.5. Environmental impacts in WC category for each resident and environmental impacts for each sub-system that conforms the different wastewater treatment schemes. Scenario 1: conventional system; Scenario 2: conventional system with AD unit; Scenario 3: decentralised system with conventional toilets; Scenario 4: decentralised system with vacuum toilets

8.3.3. Economic results of the different wastewater treatment schemes

In this section, capital and operating costs were calculated for all the wastewater configurations evaluated. The results are shown in Table 8.3 (FU: 1 resident). In this case, decentralised schemes have higher capital costs than centralised systems because the units are more complex. Capital costs can increase by about 98% in these new configurations. In addition, if the benefits are not considered in wastewater treatment plants, decentralised systems also have more economic costs. The increase is about 77% for conventional toilets and 82% for vacuum toilets. However, when the benefits of decentralised systems are taken into account, the trend changes. The costs in Scenario 3 and 4 can decrease by about 67% of the total operational costs compared to centralised systems. Concerning to the environmental prices, the trend is the same than in the operational costs. The decentralised systems present environmental credits whereas in the centralised schemes, the environmental costs can increase in 10%. Finally, if the operating, environmental and construction costs are linked, decentralised systems are better from an economic point of view. The total costs can be reduced by 27% for Scenario 3 and 21% for Scenario 4.

Finally, if the centralised systems are compared from an economic point of view, operating and construction costs can be reduced by 1%. Thus, a priori, the incorporation of the AD unit can be an advantage, however, the energy production in this unit should be improved. Within this framework, the payback time must be calculated to obtain the different values according to the different wastewater treatment configurations.

The values of the different configurations are presented in Table 8.3. The worst result is shown for the conventional case without AD unit (Scenario 1), the payback time is about 13 years followed by Scenario 2 (conventional case with an AD unit). It is true that if the AD unit is incorporated the period can be reduced by 5 years. Decentralised plants, although they have higher investment costs, electricity production is higher, and the payback time is shorter than in centralised cases. In Scenario 3 (conventional toilets), the time is about 5 years while in Scenario 4 (vacuum toilets) it is 4 years.

Table 8.3. Operational and construction costs for the different wastewater treatment schemes considered (FU: 1 resident)

	S1	S2	S3	S4
Operational costs (€)				
Electricity consumption				
Pre-treatment	$1.88\cdot10^{-4}$	$1.88\cdot10^{-4}$	-	-
Coagulation-flocculation	$6.18\cdot10^{-4}$	$6.18\cdot10^{-4}$	-	-
CAS unit	$4.84\cdot10^{-4}$	$4.84\cdot10^{-4}$	-	-
Thickening + homogenization	$2.73\cdot10^{-5}$	$2.73\cdot10^{-5}$	-	-
AD unit	-	$3.49\cdot10^{-4}$	-	-
Dewatering	$2.72\cdot10^{-4}$	$2.72\cdot10^{-4}$	-	-
Composting	$2.76\cdot10^{-5}$	$2.76\cdot10^{-5}$	-	-
Toilets	-	-	-	$7.39\cdot10^{-3}$
UASB	-	-	$4.51\cdot10^{-4}$	$2.11\cdot10^{-3}$
MBR	-	-	0.02	0.02
SBR	-	-	0.01	0.01
Cogeneration unit				
Lubricant oil	-	$1.98\cdot10^{-4}$	0.10	0.12
Chemical consumption				
$FeCl_3$	$1.23\cdot10^{-7}$	$1.23\cdot10^{-7}$	-	-
Polyelectrolyte	0.08	0.08	-	-
Sludge management				
Sludge	0.03	0.03	-	-
Avoided electricity				
Electricity	-	$1.46\cdot10^{-3}$	0.10	0.12
TOTAL OPEX (€)	0.10	0.09	0.03	0.04

Table 8.3.(cont.). Operational and construction costs for the different wastewater treatment schemes considered (FU: 1 resident)

	S1	**S2**	**S3**	**S4**
Construction costs (€)				
Pre-treatment	$9.18\cdot10^{-5}$	$9.18\cdot10^{-5}$	-	-
Coagulation-flocculation	$9.18\cdot10^{-5}$	$9.18\cdot10^{-5}$	-	-
CAS unit	$8.17\cdot10^{-5}$	$8.17\cdot10^{-5}$	-	-
Thickening + homogenization	$6.80\cdot10^{-5}$	$6.80\cdot10^{-5}$	-	-
AD unit	-	$7.40\cdot10^{-5}$	-	-
Dewatering	$4.81\cdot10^{-5}$	$4.81\cdot10^{-5}$	-	-
Composting	$7.08\cdot10^{-5}$	$7.08\cdot10^{-5}$	-	-
Cogeneration	$7.10\cdot10^{-5}$	$7.10\cdot10^{-5}$	0.01	0.01
UASB	-	-	$2.30\cdot10^{-3}$	$2.30\cdot10^{-3}$
MBR	-	-	0.01	0.01
SBR	-	-	0.01	0.01
TOTAL CAPEX (€)	$4.53\cdot10^{-4}$	$5.98\cdot10^{-4}$	0.04	0.04
ENVIRONMENTAL COST (€)	0.09	0.08	-0.01	$-3.07\cdot10^{-5}$
TOTAL COST (€)	**0.19**	**0.17**	**0.05**	**0.07**
Payback time (y)	13	8	5	4

8.3.4. Sensitivity analysis

In the previous section, the advantages of electricity recovery in decentralised systems were demonstrated. In this framework, it is important to think about what would happen if electricity was not recovered in decentralised systems. For this analysis, two scenarios were considered: (i) a comparison between centralised and decentralised systems with energy recovery and (ii) a comparison between centralised and decentralised systems without energy recovery. The most affected category in terms of energy is CC category (Stocker et al., 2013). For this reason, only the CC category will be analysed.

The results are shown in Figure 8.6. The results for Scenario 1 are the same, because this system does not have energy recovery. However, for the other cases, the environmental profile in this category can increase between 5% for Scenario 2 and 99% for the decentralised cases.

It is interesting to note that if there is no energy recovery, the decentralised cases have very similar impact values to the centralised system, even higher for Scenario 4 (vacuum toilets). Therefore, if there is no recovery (such as energy or water) in decentralised systems, their application will not be appropriate to treat wastewater because the environmental impacts will be greater.

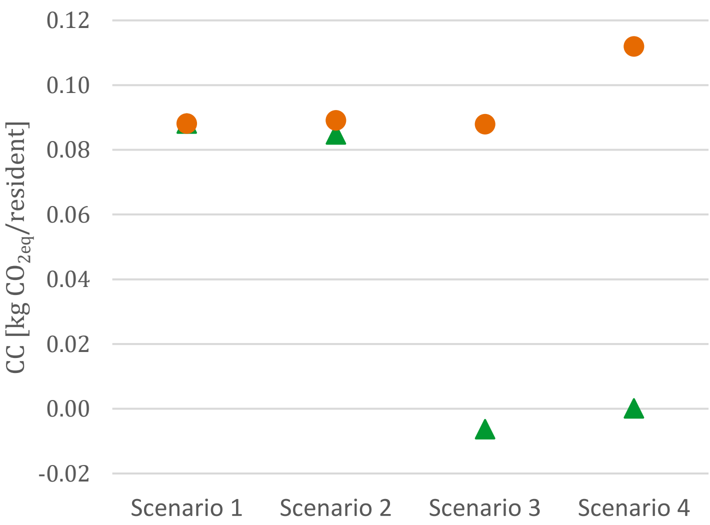

Figure 8.6. Climate change profile for the different wastewater treatment configurations (FU: 1 resident). Symbols: (o) without energy recovery; (Δ) with energy recovery. Scenario 1: conventional system; Scenario 2: conventional system with AD unit; Scenario 3: decentralised system with conventional toilets; Scenario 4: decentralised system with vacuum toilets

8.3.5. Broadening the scope in centralised and decentralised systems to include sewer network

As mentioned above, in terms of investment costs, decentralised systems have disadvantages compared to centralised systems. In general, these systems can be more complex due to the construction of the membrane or the aeration equipment. However, centralised systems are more robust and less adaptable to recovering products such as water or nutrients. In addition, decentralised systems are characterised by a lower sewage network compared to centralised systems (Opher and Friedler, 2016). It is estimated that the sewer network has an have a significant contribution to the overall impact of construction of WWTPs (Petit-Boix et al., 2014). Thus, in this case, it was evaluated how the sewerage network affects the environmental profile.

In the city of Santiago de Compostela, the extension of sewage network is 647 km, which implies an amount of 5 m/inhabitant. In decentralised systems, this figure is estimated to be about 3.7 m/inhabitant (Kjerstadius et al., 2017). If the environmental profiles are compared, as expected, the centralised system (455 kg CO_{2eq}/resident) has 76% higher amount that the decentralised systems (108 kg CO_{2eq}/resident). These impacts are related to the production of concrete for trenching and pipe material but not only these factors are important, there are taken into account the capacity of the sewer network. The sewer network in Santiago de Compostela has a higher capacity because there is no separation network (wastewater and rainwater), therefore, the capacity of the sewerage must be high because in this city the rainfall is high. On the contrary, in the decentralised system, although there are two pipes (one for BW and another for GW), the capacity is reduced. This implies less environmental impacts related to the construction of the pipelines, ditches or even direct emissions related to the construction.

The introduction of a separate network in Santiago is not simple due to the protection of its old town, so changing the sewage network is not a viable option, but decentralised systems of wastewater and sewage can be a good alternative in new neighbourhoods and can improve the environmental profile of these networks not only in the CC category but in all categories, making the resident have less consumption of carbon and water in terms of irrigation than residents who choose another type of neighbourhood.

8.4. CONCLUSIONS

In this Chapter, the carbon footprint and water consumption for irrigation of a resident living in a centralised wastewater district was compared to that of a resident who chooses to live in a decentralised wastewater district. The study was carried out in the city of Santiago de Compostela. In this framework, two centralised configurations: (i) conventional system without AD unit and (ii) a conventional system with the incorporation of AD unit were compared with decentralised options: one with conventional toilets and another with vacuum toilets. The decentralised options show a reduction of the resident carbon footprint by 20-23% due to electricity production. Furthermore, with the reclaimed water, these systems can supply water for irrigation of green areas, so no extra consumption of tap water is required. Although these new systems present more construction costs and are more complex, the recovery time is less than in conventional systems due to the recovery of products such as energy or water. However, the incorporation of these systems is not easier due to the robustness of conventional systems. Thus, the option of decentralised cases can be an optimal solution for new buildings or residential areas.

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GENERAL CONCLUSIONS AND FUTURE PERSPECTIVES

General conclusions and future perspectives

The main objective of this doctoral thesis was to analyse and compare different wastewater treatment configurations from an environmental and economic point of view. This topic is in line with the growing concern to alleviate the effects related to climate change and water scarcity caused by anthropogenic activities and population growth. In this sense, WWTPs should be included in the philosophy of the circular economy and have emerged as a solution to recover products such as energy, nutrients and reclaimed water. In this context, two innovative strategies for wastewater treatment were evaluated: (i) one for centralised systems (Chapter 2 to Chapter 6) and (ii) one based on decentralised wastewater treatment schemes (Chapter 7 and Chapter 8). It was demonstrated that environmental impact methodologies and economic indicators provide useful information to assist the integration of these wastewater treatment strategies. The main findings and conclusions drawn from the different sections that make up this thesis are presented below:

Section I: Improving centralised wastewater treatment systems. The main objective is to create a virtual wastewater treatment plant that encompasses the best technologies from an environmental and economic point of view for centralised systems. The conceptual design of a "virtual plant" will be based on the analysis developed from Chapter 2 to Chapter 6 in the framework of five different studies detailed below.

In Chapter 2, AD technology was analysed at different scales with the main objective of assessing the environmental and economic viability of this technology. This treatment can be a good alternative for treating sludge due to the reuse of biogas as heat or energy as well as the potential of the digestate as a biofertiliser. In addition, if the amount of sludge is not very high, there are alternatives to improve the production yield of biogas such as the co-digestion of sewage sludge with food waste from households, services or even from the agro-food sector. Therefore, this technology will be included in the "virtual plant" as this chapter showed its efficiency. Despite the environmental and economic advantages of AD, sludge treatment can be a slow process. As mentioned in the introduction and in Chapter 3, hydrolysis, an initial stage in the AD process, is a limiting step, so in order to improve this unit and save treatment time, two alternative pre-treatments were proposed: chemical and thermal hydrolysis. In this case, the pre-treatments proved to be a good alternative to accelerate the hydrolysis stage and improve biogas production. It is true that the consumption of electricity and chemicals worsens the environmental profile but, it can be compensated with the increase in biogas production. This means that pre-treatments can also be a good alternative for the "virtual plant". However, these processes are still under development, and more information is needed to incorporate these pre-treatments into a real sludge line.

In Chapter 4 and 5, treatment schemes at different scales were proposed to treat wastewater in a carbon neutral perspective. In Chapter 4, the WWTP scale is 100,000 equivalent inhabitants and in Chapter 5, the WWTPs scales are for 265,000 and 1,000,000 of equivalent inhabitants. As a summary, Figure 10.1 shows the results for different scenarios and different plant sizes. This figure may indicate the trend that plants should follow to have more environmentally friendly and economically viable schemes in centralised WWTPs. Thus, a priori, for large plants, the conventional scheme (PC + CAS) is the worst scheme due to the high energy consumption of the CAS unit, and there is less biogas production than in the other scenarios. The case of RBF + CEPT + IFAS that was incorporated in the smaller plant (100,000 equivalent inhabitants) is interesting. Although, there is a reduction in aeration due to the incorporation of IFAS technology it is not appropriate due to the consumption of chemicals in the primary treatment. This incorporation of the chemical can increase the environmental profile and economic impacts. For this reason, not all schemes are appropriate. The best solutions from an environmental and economic point of view and that can try to make plants carbon neutral are combinations based on UASB and IFAS as well as the HRAS and IFAS sequence. In the first case, the sludge line is not necessary, so this implies a reduction in land occupation and, in the UASB unit, biomass growth is slower than in aerobic units, so this implies a reduction in the amount of sludge. If this scheme is not possible, the HRAS unit allows a high OM recovery with a high methanisation factor while the subsequent IFAS stage provides advantages such as good nitrogen removal and low energy consumption (Figure 9.1).

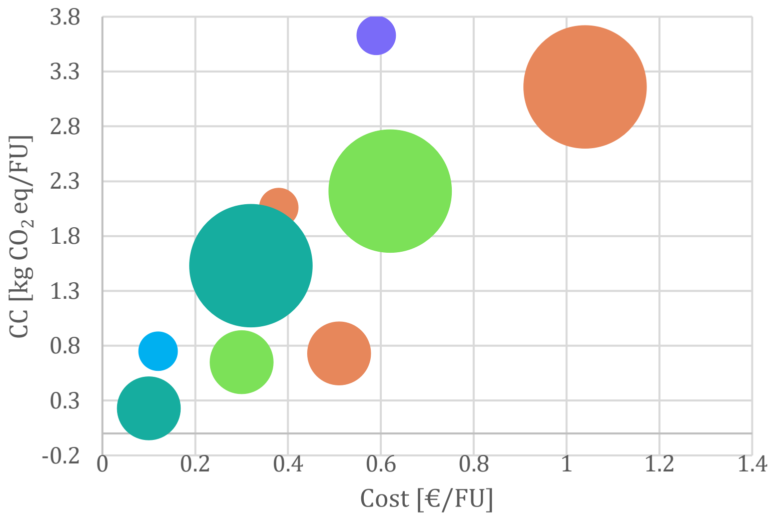

Figure 9.1. Different environmental and economic results for the wastewater treatment schemes studied. Bubbles represent the size of the plant and the colours correspond to different schemes. Orange: PC + CAS technologies, purple: RBF + CEPT+ IFAS, blue: UASB + IFAS, green: ERBF+ IFAS, and finally, turquoise: HRAS + IFAS

Finally, Chapter 6 is related to the scale-up of an emerging technology. This chapter is very important in determining the minimum scale for reliable LCA and economic evaluation. In a context where decentralisation is becoming increasingly important, it is crucial to verify this methodology in the calculation of environmental impacts. This study can help to know whether the LCA approach makes sense in decentralised schemes. After conducting the study, the minimum volume that provides reliable environmental impacts was selected as 0.2 m^3, while for economic indicators, the minimum scale was 1 m^3. This means that when decentralised systems are studied, the volume needed to have consistent data will be 0.2 m^3. Smaller scales may provide an unrealistic profile.

Section II: Changing the paradigm of wastewater treatment. This section consists of two chapters that focus on the possible advantages and disadvantages of different decentralised systems. Thus, as in the previous case, the main objective is to try to give a general approach. However, in this case it is more complicated than in the previous section because, the chapters are based on two different perspectives.

First, a decentralised wastewater treatment plant based on a MBR unit for 2,000 inhabitants and located in Turkey was evaluated from an environmental point of view. In this case, the priory of the system was reused water in green areas because Turkey is a country with water deficit. For this propose, an indicator called AWARE was calculated. In addition, in this analysis, construction and operation phases were studied to quantify the environmental impacts related with the construction of the decentralised systems. With the objective of water reuse in mind, MBR is a technology that achieves satisfactory results on terms of water quality. Moreover, the water reuse had significant environmental impacts in all categories. Finally, for decentralised systems, the construction stage associated with the membrane fabrication can present high environmental impacts.

Finally, Chapter 8 focuses on the point of view of the inhabitant. The study from the point of view of the inhabitant is very important because the citizen is increasingly aware that in a changing and continuously growing world, anthropogenic activities must ensure an exhaustive control of emissions and therefore lower carbon and water footprint values. Thus, this study compared a resident living in a centralised area with one living in a decentralised neighbourhood for wastewater treatment. The carbon footprint of a resident in terms of energy consumption can be reduced by 20-23% in areas that incorporate a decentralised system. Additionally, the water demand for green areas can be covered by reclaimed water. Therefore, there are no impacts related to water treatment and distribution. Finally, it is important to point out that these plants can be more flexible, and it is easy to recover resources, especially in countries where there are pressing problems of water scarcity. Therefore, these systems can also be a good option for new buildings or residential areas.

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