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Obtaining new-generation biomass fuels and bio-based fertilisers from sludges through biodrying and advanced composting technologies to enhance the overall resource recovery of the water sector

PhD Thesis by

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Iragana dut nik oinarria,
Etorkizuna nire aberria
Besteak lagundu eta maitatzeko
Aurkitu behar dut nire esentzia

Ta ez dut nahi nire gain eraman beste inoren zama,
Itoarazten nauten kate hauetatik askatu
Naturaren zati izanik sakon hartuz arnasa
Itoarazten nauten kate hauetatik askatu

Naturaren zati izanik sakon hartuz arnasa
Barrura so egin eta nahiko nuke
Nor naizen jakin,
Zer naizen jakin

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ABSTRACT

The current European circular economy action plan opens a window towards resource recovery from organic wastes, including sewage sludge. In line with the Green Deals related to climate action and farm to fork strategy, sewage sludge presents potential as nutrient and energy source. The technical, environmental and economic dimensions of different technological solutions have been described in the literature identifying technological complexity and rather limited economic feasibility as the main weaknesses of such technologies. High quality and sustainable biomass fuels and bio-based fertilisers can be obtained by applying biodrying and advanced composting technologies to sludges with potentially high nutrient and energy content, respectively.

With this objective in mind, a pilot plant was built able to monitor relevant process parameters and two innovative control algorithms were implemented for biodrying and advanced composting based on bulk mixture temperature and oxygen uptake rate, respectively.

The technical feasibility of biodrying technology was evaluated for different types of sludge finding cellulosic sludge and primary sludge to be the most promising ones in terms of process efficiency and final product quality. The use of fat-rich complementary materials can be used to effectively enhance the biodrying process of sludges with not satisfactory performance achieving up to 70% removal of initial moisture content. Finally, three different aeration strategies were implemented for biodrying of cellulosic sludge and then compared in terms of biodrying performance and quality of end-products. These strategies were based on different combinations of convective drying with biogenic heat produced achieving in all of the combinations implemented satisfactory drying efficiencies. Two innovative biodrying performance indicators (Energetic Biodrying Index and Biodrying Performance Index) were proposed to better assess the initial and operational conditions that favour the maximum energy process efficiency and the highest end-product quality. The products obtained with all the three strategies consistently achieved satisfactory quality parameters. Finally, the environmental and economic dimensions of the biodrying technology applied to sludges were assessed through the monitoring of relevant polluting gaseous emissions (direct and indirect GHG, NH_3, and tVOCs) and the techno-economic analysis based on LCC methodology.

The production of safe and high-quality bio-based fertilisers from phosphorus rich sludge was possible by using advanced composting. The process was demonstrated to be efficient, robust and replicable.

The quality of the product obtained has been proved to be high enough to be categorised as a solid organic fertiliser rich in nutrients, in particular phosphorus, with an agronomic performance in pot tests equivalent to phosphorus-based mineral fertilisers. Moreover, the deep assessment of organic and inorganic pollutants in the bio-based fertiliser, demonstrated the safety of the product obtained. Polluting gaseous emissions and particularly emissions of ammonia were confirmed to be the main weakness of the advanced composting process of phosphorus rich sludge. This weakness was addressed by using spent mesolite which mitigated ammonia emissions by up to 85%. Finally, techno-economic analysis performed by applying the LCC methodology established a positive economic scenario for plants with big or very big capacity.

As a conclusion, the solutions proposed contribute effectively to the European circular economy by suggesting cheap and efficient alternatives to valorise sewage sludge into energy rich and nutrient rich products with high added value. The solutions boost the end of waste status of sludge making in turn, firm steps towards the independency of non-renewable energy and nutrient sources. Nevertheless, specific regulation and standardisation of the recovered products are critical together with overcoming the legal barriers that this kind of product is facing to reach the European market. That would substantially help to gain the consumers' confidence, increasing also the competitiveness of the products upon conventional products that are already in the market.

RESUM

El pla d'acció europeu d'economia circular actual obre una finestra a la recuperació de recursos a partir de residus orgànics, inclosos els fangs de depuradora. D'acord amb els Acords Verds relacionats amb l'acció climàtica i l'estratègia "de la granja a la taula", els fangs de depuradora presenten un potencial com a font de nutrients i energia. A la literatura s'han descrit les dimensions tècniques, mediambientals i econòmiques de diferents solucions tecnològiques, identificant la complexitat tecnològica i una viabilitat econòmica força limitada com els principals punts febles d'aquestes tecnologies. Es poden obtenir combustibles de biomassa i fertilitzants biològics sostenibles i d'alta qualitat aplicant tecnologies de bioassecatge i compostatge avançat de llots amb un contingut potencialment alt de nutrients i energia, respectivament.

Amb aquest objectiu, es va construir una planta pilot capaç de monitoritzar els paràmetres rellevants del procés i es van implementar dos algoritmes de control innovadors per al bioassecatge i el compostatge avançat basats en la temperatura de la mescla i la taxa de consum d'oxigen, respectivament.

Es va avaluar la viabilitat tècnica de la tecnologia de bioassecatge per a diferents tipus de llots, concloent que els llots cel·lulòsics i els llots primaris són els més prometedors en termes d'eficiència del procés i qualitat del producte final. L'ús de materials complementaris rics en greixos pot millorar eficaçment el procés de bioassecatge dels fangs amb un rendiment no satisfactori, aconseguint una eliminació de fins al 70% del contingut d'humitat inicial. Finalment, es van aplicar tres estratègies d'aireació diferents per al bioassecatge de llots cel·lulòsics i es van comparar en termes de rendiment de bioassecatge i qualitat dels productes finals. Aquestes estratègies es van basar en diferents combinacions d'assecatge per convecció i per calor biogènica produïda aconseguint en totes les combinacions implementades eficiències d'assecatge satisfactòries. Es van proposar dos indicadors innovadors de rendiment del bioassecatge (Índex Energètic de Bioassecatge i Índex de Rendiment de Bioassecatge) per avaluar millor les condicions inicials i operatives que afavoreixen la màxima eficiència energètica del procés i la major qualitat del producte final. Els productes obtinguts amb les tres estratègies van assolir sistemàticament paràmetres de qualitat satisfactoris. Finalment, es van avaluar les dimensions ambiental i econòmica de la tecnologia de bioassecatge aplicada als fangs

mitjançant el seguiment de les emissions gasoses contaminants rellevants (GEI directes i indirectes, NH3 i tVOCs) i l'anàlisi tecno-econòmic basat en la metodologia de l'ACC.

La producció de fertilitzants biològics segurs i d'alta qualitat a partir de llots rics en fòsfor va ser possible mitjançant el compostatge avançat. Es va demostrar que el procés és eficient, robust i reproduïble. S'ha demostrat que la qualitat del producte obtingut és prou alta per ser catalogat com un fertilitzant orgànic sòlid ric en nutrients, en particular fòsfor, amb un rendiment agronòmic en bioassaigs en testos equivalent als fertilitzants minerals basats en fòsfor. A més, l'avaluació en profunditat dels contaminants orgànics i inorgànics al fertilitzant de base biològica va demostrar la seguretat del producte obtingut. Es va confirmar que les emissions gasoses contaminants, i en particular les d'amoníac, eren el principal punt feble del procés de compostatge avançat de fangs rics en fòsfor. Aquest punt feble es va solucionar mitjançant l'ús de mesolita esgotada, que va mitigar les emissions d'amoníac fins a un 85%. Finalment, l'anàlisi tecno-econòmic realitzat aplicant la metodologia LCC va establir un escenari econòmic positiu per a les plantes de capacitat de tractament gran o molt gran.

Com a conclusió, les solucions proposades contribueixen eficaçment a l'economia circular europea suggerint alternatives barates i eficients per valorar els fangs de depuradora en productes rics en energia i nutrients amb un alt valor afegit. Les solucions impulsen la fi de la condició de residu dels fangs fent, alhora, passos ferms cap a la independència de les fonts d'energia no renovables i nutrients. No obstant això, és fonamental la regulació i la normalització específica dels productes recuperats, així com la superació de les barreres legals a què s'enfronten aquest tipus de productes per arribar al mercat europeu. Això ajudaria substancialment a guanyar la confiança dels consumidors, augmentant també la competitivitat dels productes enfront dels productes convencionals que ja són al mercat.

RESUMEN

El actual plan de acción europeo de economía circular abre una ventana a la recuperación de recursos a partir de residuos orgánicos, incluidos los lodos de depuradora. En consonancia con los Acuerdos Verdes relacionados con la acción climática y la estrategia "de la granja a la mesa", los lodos de depuradora presentan un potencial como fuente de nutrientes y energía. En la literatura se han descrito las dimensiones técnicas, medioambientales y económicas de diferentes soluciones tecnológicas, identificando la complejidad tecnológica y una viabilidad económica bastante limitada como los principales puntos débiles de dichas tecnologías. Pueden obtenerse combustibles de biomasa y fertilizantes biológicos sostenibles y de alta calidad aplicando tecnologías de biosecado y compostaje avanzado de lodos con un contenido potencialmente alto de nutrientes y energía, respectivamente.

Con este objetivo, se construyó una planta piloto capaz de monitorizar los parámetros relevantes del proceso y se implementaron dos algoritmos de control innovadores para el biosecado y el compostaje avanzado basados en la temperatura de la mezcla y la tasa de consumo de oxígeno, respectivamente.

Se evaluó la viabilidad técnica de la tecnología de biosecado para diferentes tipos de lodos, concluyendo que los lodos celulósicos y los lodos primarios son los más prometedores en términos de eficiencia del proceso y calidad del producto final. El uso de materiales complementarios ricos en grasas puede conllevar a mejorar eficazmente el proceso de biosecado de los lodos con un rendimiento no satisfactorio, logrando una eliminación de hasta el 70% del contenido de humedad inicial. Por último, se aplicaron tres estrategias de aireación diferentes para el biosecado de lodos celulósicos y se compararon en términos de rendimiento de biosecado y calidad de los productos finales. Estas estrategias se basaron en diferentes combinaciones de secado por convección y por calor biogénico producido logrando en todas las combinaciones implementadas eficiencias de secado satisfactorias. Se propusieron dos innovadores indicadores de rendimiento del biosecado (Índice Energético de Biosecado e Índice de Rendimiento de Biosecado) para evaluar mejor las condiciones iniciales y operativas que favorecen la máxima eficiencia energética del proceso y la mayor calidad del producto final. Los productos obtenidos con las tres estrategias alcanzaron sistemáticamente parámetros de calidad satisfactorios. Por último, se evaluaron las dimensiones ambiental y económica de la tecnología de biosecado aplicada a los lodos mediante el seguimiento de las emisiones gaseosas

contaminantes relevantes (GEI directos e indirectos, NH3 y tVOCs) y el análisis tecno-económico basado en la metodología del ACC.

La producción de fertilizantes biológicos seguros y de alta calidad a partir de lodos ricos en fósforo fue posible mediante el compostaje avanzado. Se demostró que el proceso es eficiente, robusto y reproducible. Se ha demostrado que la calidad del producto obtenido es lo suficientemente alta como para ser catalogado como un fertilizante orgánico sólido rico en nutrientes, en particular fósforo, con un rendimiento agronómico en bioensayos en maceta equivalente a los fertilizantes minerales basados en fósforo. Además, la evaluación en profundidad de los contaminantes orgánicos e inorgánicos en el fertilizante de base biológica, demostró la seguridad del producto obtenido. Se confirmó que las emisiones gaseosas contaminantes, y en particular las de amoníaco, eran el principal punto débil del proceso de compostaje avanzado de lodos ricos en fósforo. Este punto débil se solucionó mediante el uso de mesolita agotada, que mitigó las emisiones de amoníaco hasta en un 85%. Por último, el análisis tecno-económico realizado aplicando la metodología LCC estableció un escenario económico positivo para las plantas de capacidad de tratamiento grande o muy grande.

Como conclusión, las soluciones propuestas contribuyen eficazmente a la economía circular europea al sugerir alternativas baratas y eficientes para valorizar los lodos de depuradora en productos ricos en energía y nutrientes con alto valor añadido. Las soluciones impulsan el fin de la condición de residuo de los lodos dando, a su vez, pasos firmes hacia la independencia de las fuentes de energía y nutrientes no renovables. No obstante, es fundamental la regulación y normalización específica de los productos recuperados, así como la superación de las barreras legales a las que se enfrentan este tipo de productos para llegar al mercado europeo. Esto ayudaría sustancialmente a ganar la confianza de los consumidores, aumentando también la competitividad de los productos frente a los productos convencionales que ya están en el mercado.

LIST OF TABLES

LIST OF FIGURES

LIST OF ACRONYMS

ARGs	Antibiotic resistance genes
AT4	Cumulative oxygen consumption in 4 days
BA	Bulking agent
BBF	Bio based fertiliser
BI	Biodrying index
BMF	Biomass fuel
BPI	Biodrying performance index
CAPEX	Capital expendidures
CG	Coffee grownd
COD	Chemical oxygen demand
CS	Cellulosic sludge
d.b	dry basis
DE	Diatomaceous earth
Dm	Dry matter
DRI	Dynamic respirometric index
EBI	Energetic biodrying index
EBPR	Enhanced biological phosphorus removal
EC	Electrical conductivity
EEA	European environment agency
EP/EC	Energy production/Energy consumption
FAS	Free air space
FI	Fertiliti Indicator
Fn	Airflow in the cycle n
Fn+1	Airflow in the cycle n+1
GHG	Greenhouse gases
GWP	Global warming potential
HHV	Higher heating value
IRR	Internal rate of return
LCA	Life cycle analysis
LCC	Life cycle costing
LFT	Lifetime
LHV	Lower heating value
MBT	Mechanical and biological pre-treatment
MC	Moisture content
MS	Mixed sludge
MSW	Municipal solid waste
NPV	Net present value
OFMSW	Organic fraction of municipal solid waste
OPEX	Operational expendidures
OUR	Oxygen uptake rate
PAH	Polycyclic aromatic hydrocarbons
PC	Plant capacity
PCB	Polychlorinated biphenyls
PE	Population equivalents
PEF	Product environmental footprint
PHAs	Polyhydroxyalkanoates
PP	Payback period
PPCP	Pharmaceuticals and personal care products
PPS	Pulp and paper mill sludge
PRS	Primary sludge
PS	Phosphorus rich sludge
PW	Pruning waste
RIE	Respiration Index Efficiency
RIE-EC	Respiration Index Efficiency - Energy consumption
RTD	Residence time distribution
S1, S2, S3	Strategies 1, 2 and 3, respectivelly assessed for biodrying
SDG	Sustainable Development Goals
SI	Safety index
SPR	Specific production ratio
SRF	Solid recovered fuels
SS	Secondary sludge
St.	Stage
TIP	Temperature Increasing phase
TOC	Total organic carbon
TKN	Total potassium
TKN	Total Kjeldahl nitrogen
TP	Total phosphorus
TS	Total solids
TSP	Triple super phosphate (positive control in composting trials)
VFA	Volatile fatty acids
VOCs	Volatile organic compounds
VS	Volatile solids
w.b	wet basis
WW	Winterization waste
WWTP	Wastewater treatment plant

1. INTRODUCTION

1.INTRODUCTION

Bearing in mind that the resources of our planet are finite and sometimes scarce, the classical linear economic model that assumes unlimited growth and resources has no place. Hence, the world is making efforts for the transition towards a more circular economy, which would allow a resource-efficient economy by lengthening the lifecycle of products while reducing waste production and greenhouse gas emissions. Accordingly, circular economy has been raised as an effective approach that contributes greatly to sustainable development challenge addressed through the United Nations Sustainable Development Goals (SDGs). Specifically, Europe is targeting to lead the transition to the circular economic model to increase the competitiveness of the European economy, create business opportunities and new jobs by implementing innovation in all the productive chains and forms of consumption. To this aim, the European Commission established in its Circular economy action plan (European Commission, 2020b) some key priorities, raised legislative proposals, and launched several strategies being European Green Deals the most noteworthy in this context. Several initiatives for climate action and healthier and more sustainable food systems are expected to boost the change towards the new economic model.

1.1 BIODEGRADABLE WASTES IN THE CIRCULAR ECONOMY FRAMEWORK:

The establishment of waste hierarchy (Waste Framework Directive 2008/98/EC and its modification 2018/851) and Landfilling limitation (Landfill Directive 1999/31/EC) were done as first firm steps towards circularity. In this framework, leaving aside the need for a more efficient production chain allowing the reduction of the volume of wastes generated, the development of effective recycling strategies is fundamental. Herein, some criteria for end-of-waste status were established for all types of wastes, considering them valuable resources aimed to be reintroduced into production processes to be ultimately transformed into new products. Among the different types of wastes generated by human activity, biodegradable waste was defined in the Landfill Directive as "*any waste that is capable of undergoing anaerobic or aerobic decomposition*" (European Commission, 1999). The bio-based product concept is applied to those products obtained from the technological transformation of biodegradable wastes in the framework of circular strategies.

By definition, bio-based products are wholly or partly derived from materials of biological origin. They are new-generation products obtained when they undergo a wide range of industrial processes that can lead also to higher process efficiencies, reduction of energy and/or water consumption, reduction of harmful gaseous emissions, and/or a reduction of wastes produced during the process. All the mentioned contribute to the development of markets and increase the competitiveness of European bioeconomy sectors. At this point and in this specific context, the concept of biorefinery arises as a facility that integrates different conversion processes and technologies to produce fuels, power, and value-added chemicals from biodegradable materials, among others from wastes. Accordingly, European Commission is already promoting its circular economy action plan. For instance, the new regulation on the European fertilising product market (Regulation 2019/1009) includes organic- and waste-based fertilizers in the categories established suitable for the European fertilisers market. New-generation innovative and sustainable bio-based fertilisers (BBF) are aligned with the strategies mentioned and fit perfectly in the new market of European fertilising products. Similarly, in the context of promoting the production of renewable fuels (Directive 2018/2001) towards the mitigation of climate change, European Commission encourages the development and implementation of technological solutions for the production of energy from renewable sources.

Parallelly, under the umbrella of the Water Framework Directive (2000/60/EC), the Directive regarding Urban wastewater treatment (91/271/ECC) prioritised the improvement of wastewater collection systems and their treatment infrastructure.

1.2 INTEGRAL TREATMENT OF WASTEWATER

The development level of the urban wastewater collection network and the integral treatment capacity for wastewater is strongly linked to the development level of a country (Colón et al., 2017; Cieslik et al., 2018). Following the SDG related to clean water and sanitation, increased sanitation is certainly being achieved all over the world, reducing thus, the spreading of diseases and improving the overall public health. Information regarding urban wastewater generation and treatment all over the world is limited and most probably outdated (Sato et al., 2013). According to the report published by UNICEF and the world health organization in 2019, 54% of the world population are able to use safely managed sanitation services (UNICEF & WHO, 2019). In general terms, it can be stated that high-income countries have the most advanced wastewater collection and treatment networks. Moreover and having Europe as a reference, it could be affirmed that almost the totality of wastewater can be collected and appropriately treated (EEA, 2020). Middle income countries, have less-developed wastewater infrastructure and serve a lower percentage of their population, while, low-income countries have no sanitation system or if existing, it is very basic and limited to most urban areas. At this juncture, the differences among countries regarding safely managed sanitation services are clearly shown in Figure 1-1.

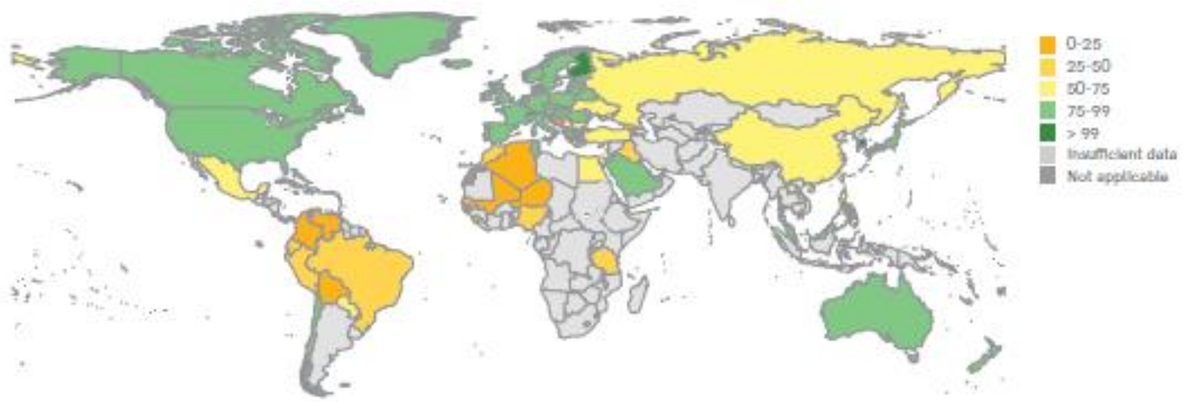


Figure 1-1. Proportion of population using safely managed sanitation services in 2017 (%) (taken from UNICEF and WHO, 2019).

Parallelly, the treatment level that collected urban wastewater undergoes varies notoriously among countries. A summary of the most widespread treatment operations for wastewater consisting of different sequential stages is shown in Figure 1-2.

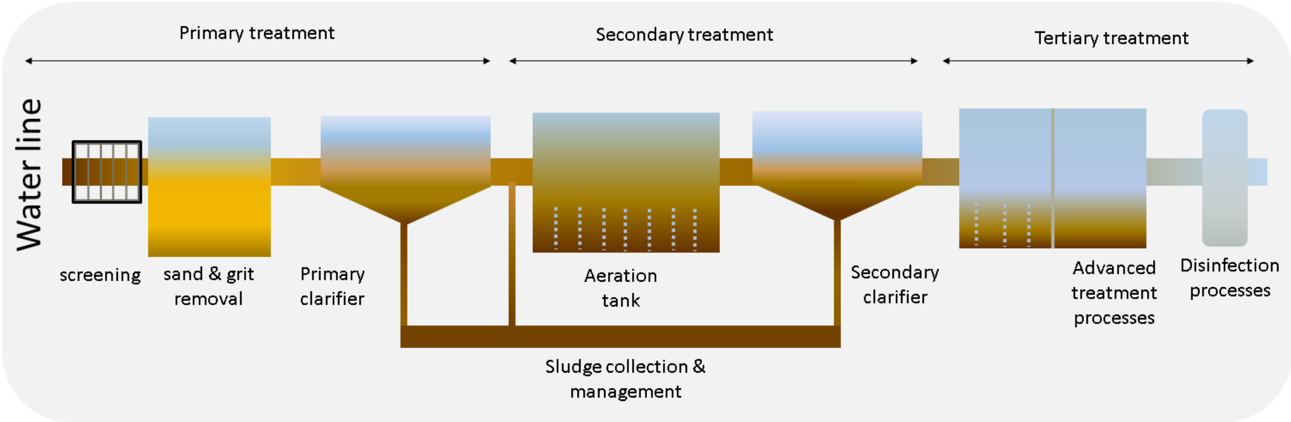


Figure 1-2. Summary of the main wastewater treatment stages in a conventional WWTP.

Primary treatment allows the removal of most coarse materials, sand, and fat that could affect the subsequent secondary treatment, which relies on biological processes to remove dissolved and suspended organic matter using microorganisms (Nunes et al., 2021). Different configurations can be found at this secondary stage, being the most extended one the system based on conventional activated sludge. The wastewater treatment is completed when more advanced treatment stages mostly to remove nutrients are implemented in the plant as tertiary treatment stage. To give some values as a reference, in Europe 88% of wastewater undergoes a minimum secondary treatment while 86% undergoes more stringent treatment (nitrogen and phosphorus removal) (European Commission, 2020). Finally, wastewater treatment produces inevitably excess sludge as a by-product that should be also considered and are discussed in the next sections.

Apart from the urban use of water and its subsequent wastewater treatment, the industry needs to be also considered when evaluating the use of water and the generation of wastewaters. According to FAO, the water withdrawal for industry constitutes 43.5 % of the total withdrawal for human activities in Europe, while the global average was estimated to be around 19 %. (FAO, 2017; EEA, 2018). European Environment Agency relates water use to a range of economic activities, either in manufacturing industry (iron and steel, non-ferrous metals, non-metallic minerals, chemicals, pulp and paper, and wood, food and drink, and other manufacturing activities processing) or energy supply that transforms primary energy source into a ready-to-use energy form. The industrial use of water is roughly equivalent for cooling systems and for industrial manufacturing (EEA, 2018).

In most cases, the industrial use of water jeopardizes the physic-chemical quality of such water and it should be treated before discharging it back to the water bodies. The pollutant content of industrial wastewaters is highly variable and dependent on the industrial process. Some of them are analogous to urban wastewaters (food and drinks manufacturing), while others present notably high concentrations of nutrients (chemicals and fertilisers industry, and certain food manufacturing), metals (Metal processing and mineral industry), or persistent organics content and emerging substances (Textiles and chemical industry, pharmaceuticals). In this context, there exist broadly three strategies for industrial wastewater management that are shown schematically in Figure 1-3. Industry's direct or indirect releases of pollution to the environment, including discharge to water bodies, is regulated in the Industrial Emissions Directive (2010/75/EU).

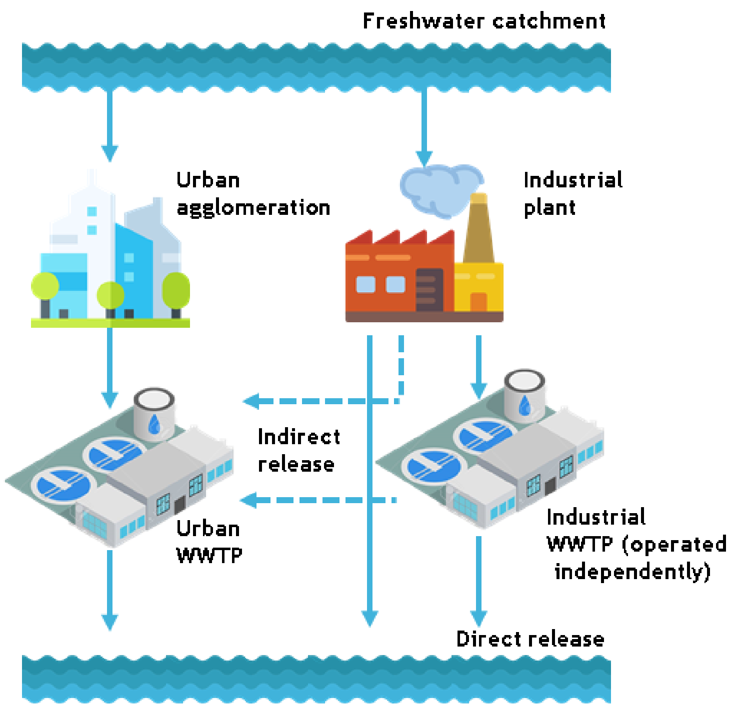


Figure 1-3. Industrial wastewater treatment cycle and discharge alternatives (direct and indirect). (adapted from EEA, 2018).

It must be highlighted, though, that the goal of a WWTP is assumed to be the safety of any downstream receiving container (environment, municipalities, etc), considering them from a one-health perspective. Hence, the most broadly implemented technologies pursue meeting effluent legal quality standards rather than resource recovery, and thus, the application of resource recovery technologies in wastewater is still poor (Raheem et al., 2018; Kehrein et al., 2020). In the framework of circular economy and considering its valuable resources, wastewater presents huge potential. It contains a considerable chemical energy potential that can be transformed into electricity and heat. Apart from that, a wide range of products can be obtained from wastewaters when applying the most suitable resource recovery strategy, including fertilizers, cellulose, volatile fatty acids, extracellular polymeric substances, single-cell protein, and CO_2 among others.

1.3 SEWAGE SLUDGE

The gradual improvement of the wastewater collection network and the rising number of facilities for its treatment has also led to increased sludge production. Roughly talking, 5-25% of the total volume of wastewater treated has been estimated to be recovered as sludge (Colin & Gazbar, 1995). To highlight the challenge that sewage sludge poses, it constitutes 9% of the total organic wastes generated in Europe, behind animal and vegetable wastes, organic fraction of municipal solid waste, and wood wastes (Alibardi et al., 2020). Specifically, the global annual sewage sludge generation was estimated to have reached 45 million dry tonnes (Gao et al., 2020).

Analysing in detail the problem, the global sewage sludge generation map is shown in the Figure 1-4. In this context, countries with the highest number of households connected to the sewer system and the highest populational densities seem to be, the highest sludge producers, thus the USA, China, and Europe are the main sludge producers. However, the values given in the map do not certainly reflect the real faecal sludge problem, given the not fully developed wastewater collection network and treatment in some countries. This might be probably the case for China, India and other Asian and Latin American countries.

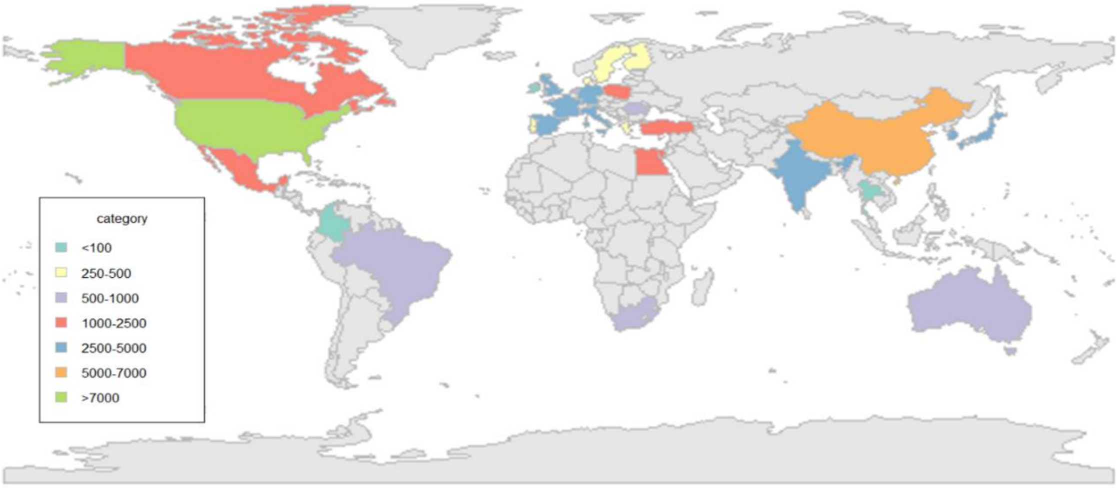


Figure 1-4. Global generation and distribution of sewage sludge (in thousand tonnes of dry sludge per year) (data taken from LeBlanc et al., 2009; Lam et al., 2016; Sharma et al., 2017; Eurostat, 2020).

Parallelly, there are some examples of industrial wastewater treatment plants that potentially can produce biological sludges comparable to municipal sewage sludges. Among the previously mentioned industrial sectors, the wastewater treatment plants in pulp & paper mill facilities, slaughterhouses, tannery facilities, palm oil mill industry, sugar cane mills, and other analogous facilities produce sludges that might be equivalent to urban wastewater sludges. Reliable data on the generation of industrial wastewater sludges is even more scarce than municipal ones. At the European Union level, Eurostat only makes 10 countries available, from a total amount of 35 Countries in the database.

As previously mentioned, until recent years, the main issue of WWTPs was dealing with wastewater to guarantee the quality of the water effluents meeting stringent standards. Consequently, sewage sludge was considered an unavoidable secondary by-product produced in the process. However, its high generation rate, which is expected to gradually increase together with the development of sewer systems, makes sludge management a major concern nowadays. In fact, about 50% of the operational costs of a wastewater treatment plant has been attributed to the sludge handling and disposal, reaching an average cost of 160-240€ per tonne of dry sludge (Winkler et al., 2013; Kacprzak et al., 2017; Gao et al., 2020; Yakamercan et al., 2021).

Sewage sludge management has not been a priority in many countries. For instance, lack of regulations and economic support for appropriate sewage sludge disposal or treatment in South America or Africa has led to practices such as landfilling or direct discharge to the environment being the most common ones (Shaddel et al., 2019). In more developed countries such as USA or Europe, land application of sludge is still a most important practice, together with incineration, although stringent legal restrictions have to be accomplished in that case. All in all, all those widespread alternatives evidence the lack of value of sewage sludge.

1.3.1 RESOURCE RECOVERY POTENTIAL OF SEWAGE SLUDGE

Sewage sludge is considered as a heterogeneous and complex mixture of organic compounds (mainly carbohydrates, proteins, and fats but also organic pollutants), microorganisms, inorganic compounds (metals, silicates, aluminates and calcium and magnesium-containing compounds) and water (Oladejo et al., 2019). Nevertheless, the volume produced of sewage sludge and its characteristics are highly dependent on its origin, wastewater treatment system, environmental requirements, and seasonal variations.

It is worth highlighting that sewage sludge is considered a promising source of nutrients (it contains nitrogen, phosphorus and potassium, and to a lesser extent, calcium, sulphur and magnesium) for plant growth. As a remark, the yearly nitrogen and phosphorus load in the European sewage system was estimated to reach 2.3-3.1 and 1.1 million tonnes of nitrogen and phosphorus (as P2O5), respectively (Kominko et al., 2021), which are majorly recognised to be stored in the sludge after the treatment process. Apart from that, sewage sludge is also a valuable source of organic carbon that applied into soils, is able to improve its physic-chemical properties, contributing also to the restoration of degraded soils due to the intensive agriculture that they have historically withstood (Nunes et al., 2021). Organic matter in conventional sewage sludge constitutes typically at least half of the dry sludge. In fact, nitrogen content in dewatered sewage sludge is typically high, which is primarily found in its organic form, mainly as proteinaceous material, since nitrogen in ammoniacal form is typically located in the liquid phase when it is subjected to phase separation (Raheem et al., 2018). Nitrogen is a highly variable compound and therefore nitrogen and different nitrogen species content varies with the operations and transformations that wastewater undergoes along with its treatment. Apart from that, various authors have affirmed that the highest part of phosphorus (between 60 and 90%) in the wastewater is incorporated in conventional sludges (Chrispim et al., 2019; Kominko et al., 2019; Shaddel et al., 2019). Wastewater has been recognised to be a promising source for recovered phosphorus. According to van Dijk et al., (2016) about 297 thousands of tonnes of phosphorus can be found in the sewage sludge in European countries with a potential yearly recovery of 182 thousand of tonnes of phosphorus (van Dijk et al., 2016) that would be able to meet 15-22% of world phosphorus demand for agriculture (Drangert, 2012; Yuan et al., 2012; Chrispim et al., 2019). The phosphorus demand in form of inorganic fertiliser (P_2O_5) for agricultural use was estimated to reach 1.2 million of tonnes in 2017 by Eurostat (Santos et al., 2021). Phosphorus is in fact a crucial element in European agriculture that was introduced in the list of critical raw materials back in 2014 (European Commission, 2014). Phosphorus is a limited resource that Europe normally imports given the absence of major phosphorus deposits.

Overall, considering its significant nitrogen and phosphorus contents, sewage sludge shows the potential to partially substitute chemical fertilizers in agriculture (Kominko et al., 2021; Santos et al., 2021). Consequently, the maximisation of nutrients recovery in sludge would help not only to achieve a more sustainable European agriculture but also to develop a more robust and self-sustaining food production, boosting ultimately the European bioeconomy.

Conversely, sewage sludge might be also a relevant pollution source for soil and indirectly for air and water when it is not properly managed, being particularly concerning pathogens and heavy metals. The origin of heavy metals in sludge has been suggested to be the mixing of municipal and industrial wastewater (Healy et al., 2016; Sharma et al., 2017), as well as with stormwater runoff. In this regard, agricultural use of sludge has become an issue recently, as the established limits generate controversy, especially regarding the potential content of persistent and toxic elements such as heavy metal and organic pollutants, including pharmaceuticals, personal care products and endocrine-disrupting compounds, which are not still considered in established regulations (Mantovi et al., 2005; Mateo-Sagasta et al., 2015; Mohapatra et al., 2016). Legal constraints and the lack of specific regulation led to uncertainties regarding the safe use of sewage sludge for nutrient recovery and thus, the barriers for the use of sewage sludge in agricultural soil with or without previous treatment are increasing. At this point, it seems clear that the regulatory framework regarding sludge management (Directive 86/278/EEC), which was published 30 years ago, needs to be revised. Many deficiencies were identified in the directive concerning mainly current risks for environmental and human health derived from the agricultural use of sewage sludge. Opposition to agricultural use of sewage sludge is increasing among stakeholders (Mininni et al., 2015; Kominko et al., 2021) and some European countries have already established more stringent rules for this practice, which are difficult to meet. For instance, there seems to exist an increasing trend to ban the land application of sewage sludge in Europe (e.g., Switzerland, Germany, or the Netherlands) (Shaddel et al., 2019; Santos et al., 2021).

Moreover, sewage sludge is not considered a key source for materials recovery in the Waste Framework Directive (2008/98/EC and its modification 2018/851), finding end-of-waste fitting concept marginally in the field of wastewater resource recovery, limited to being a feedstock for biogas production or soil amendment (Kehrein et al., 2020). Accordingly, sewage sludge is excluded as an input material in the new regulation laying down the rules on EU fertilising products on the EU market (EU Regulation 2019/1009). Consequently, there are still important challenges to be addressed, such us demonstrating that certain sludges can be used as safe and valuable source of materials, overcoming the current regulatory barriers

Apart from material resources, sewage sludge was estimated to retain 60% of the initial energy content of wastewater, containing considerable chemical energy (11.1-22.1 MJ kg^{-1}) (Oladejo et al., 2019; Bora et al., 2020). Consequently, waste-to-energy technologies for the energy recovery from sewage sludges have attracted interest in recent years due to the current European strategies towards renewable energies. In fact, within the 2030 EU Energy Strategy, a renewable-energy target of the energy consumption of 32% by 2030 was approved in the revised renewable energy directive 2018/2001/EU. Specifically, the energy roadmap 2050 established by European Commission (European Commission,2011), considers bioenergy a promising energy source able to contribute greatly to achieve the energy system transformation, including a key role of advanced and non-conventional biofuels (Scarlat et al., 2015; IRENA, 2018; Philippidis et al., 2018). In this context, although the major current contributors in the promotion of renewable energies are wind, water, and solar power, biofuels (including renewable wastes) are also relevant and promising renewable energy sources. According to Eurostat, in 2019, 34% of the energy consumed in Europe was from renewable energy sources, of which 8% were from biofuels or bioenergy (Eurostat, 2020). In addition to that, the European Commission's Knowledge Centre for Bioeconomy affirmed biological origin fuels to be the main current renewable energy sources consumed in Europe, of which practically 75% were consumed in the form of biomass, still mainly consisting of agroforestry wastes (Scarlat et al., 2019). Given that, sewage sludge has been suggested to be an attractive source of energy with a potential energy content comparable to other low-rank fuels including lignite and most of the biomass. Considering all together, properly managed, sewage sludge might be transformed into a key biomass source capable of contributing significantly to the current and future European renewable energy targets.

Nevertheless, if bioenergy is going to become an actual renewable energy source in the new European economy, a change in the energy market is needed, removing the remaining barriers to integrate bioenergy in it to develop a truly competitive market. Additionally, the development and adaptation of the infrastructure for the integration of this kind of energy are urgent and critical (Cadillo-Benalcazar et al., 2021).

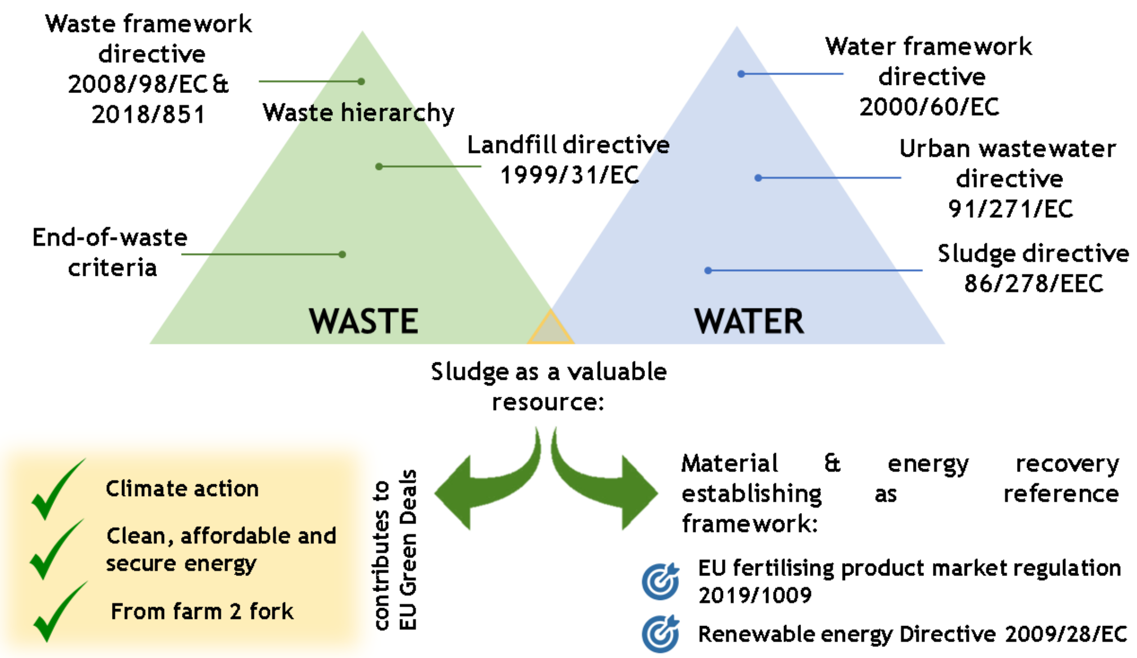


Figure 1-5 Legal and strategic framework concerning sewage sludge.

1.3.2 DEPENDENCE OF SEWAGE SLUDGE CHARACTERISTICS ON THE WASTEWATER TREATMENT

As it was mentioned before, the main purpose of a WWTP is the treatment of the wastewater to achieve a water effluent accomplishing some quality standards, avoiding, in turn, any harm in downstream systems due to its discharge. In fact, until recently, wastewater has been considered a waste rather than a valuable resource (Chrispim et al., 2019). Therefore, the classical design of a WWTP does not include resource recovery, and the implementation of technologies allowing the recovery of such resources is still poor (Kehrein et al., 2020). However, if the water sector is going to meet the current circular economy strategy, a change in the wastewater treatment paradigm is necessary starting by upgrading the existing WWTPs into water resource factories, generally known as biorefineries (Bora et al., 2020). Similarly, the design of new facilities must include this last concept, which tries to maximise the resource recovery in wastewater.

In this context, it is important to highlight that in the classical WWTP concept, sewage sludge and its downstream treatment, conversely to all the mainstream treatment, is not typically integrated into the plant design and it is considered as an unavoidable by-product. However, as it was already mentioned, the characteristics of the sludge produced are directly dependent on the operations and treatment that wastewaters undergo in the mainstream. Changes in mainstream operations or in the wastewater characteristics will greatly affect the characteristics of the sludge produced. As an example, Table 1-1 underlies the variable characteristics of different types of conventional sludges (urban and industrial). The maximum potential use of every sludge would vary also with its properties. Consequently, it would be worthwhile exploring the upgrading of the most attractive characteristics by adjusting some of the mainstream operations. That solution would allow a formulation of the sludges produced allowing their valorisation with a higher added value.

In this context, the integral wastewater treatment plants with resource recovery and the upgrading of potential characteristics of sewage sludge have led to the generation of new sewage sludge fractions with very specific characteristics. On one hand, as an example of primary wastewater treatment upgrading, the application of fine sieves is lastly attracting attention. This upgrading alternative is able to separate a sieved fraction rich in cellulose and hemicellulose which can be valorised in the chemical industry, used as asphalt, building material, or further transformed into biofuel (Ruiken et al., 2013; Ghasimi et al., 2016; Crutchik et al., 2018; Reijken et al., 2018). On the other hand, the implementation of Enhanced Biological Phosphorus Removal (EBPR) for the biological treatment stage of wastewater allows the biological storage of phosphorus present in the wastewater into a phosphorus-rich sludge (5-7% TS compared to around 2% TS in conventional sludges). This phosphorus-rich sludge might gather attractive quality parameters making them suitable for their agronomic valorisation (Yuan et al., 2012; Tyagi and Lo, 2013; Tarayre et al., 2016

Table 1-1. Main chemical characteristics of sludges produced from different wastewaters and operations.

Type of sludge	Origin (M/I)*	TS (%, wb)	VS (%, d.b.)	N-TKN (%, d.b.)	N-NH4 (%, d.b.)	TP (%, d.b.)	TK (%, d.b.)	pH	cE (mS cm^{-1})	DRI (gO_2 $kg^{-1}VSh^{-1}$)
Primary sludge	M	5-28	60-80	1.5-4		0.3-3.5		5.6-6.9		
Secondary sludge	M	15-25	52-76	3-6		0.8-2.9	0.2	6.4-7.9		
EBPR sludge	M	12-26	80	5	1	5-7	1	7-7.5		
Mixed sludge	M	26-38	60-70	2.5-4	0.5-1			5.9-7.1	1.2-1.8	6-7
Pulp & Paper mill sludge	I	19-26	80-85	0.5-5				6.2-7.8		
Slaughter house sludge	I	12.3-16.4	55-84.8	0.3-11.1	2.6	0.004-0.5	0.17-0.9	7.1-7.6		
Dairy processing industry sludge	I	6.8-13.3	59.7-62.9	5.7		3.7	0.7	7.3		
Sugar industry sludge	I	22.8-40	40-93.94	0.8-3.3		0.2-5.9	1.7-3.1	6.1-6.9	2.4-2.5	
Tannery sludge	I	17.8	35-68	0.2-1.6		0.01-0.05	0.007	7.2-7.9	0.9	

Origin: M is referred to sewage sludge from municipal WWTP and I is referred to industrial origin ones.

Data gathered from: Seyhan & Erdincler, 2003; Chen et al., 2004; Navaee-Ardeh et al., 2006; Pagans et al, 2006a; Rihani et al., 2010; Bayr & Rintala, 2012; Zhang et al., 2014; Oh et al., 2017; Crutchik et al., 2018; Grigatti et al., 2017; Longo et al., 2017; Hao et al., 2018; Zhang et al., 2018; Ashekuzzaman et al., 2019; Bhat et al., 2019; Mpofu et al., 2019; Toledo et al., 2019; Da Ros et al., 2020; Kwapinska et al., 2020; Kumar and Thakur, 2020; Laura et al., 2020; Mendieta et al., 2020; Ávila-Pozo et al., 2021

1.3.3 TECHNOLOGIES AND PRACTICES FOR SLUDGE MANAGEMENT

A wide range of different sludge management alternatives is described in this section. It is important to clarify that the selection of the best management method for sludge should be context-specific. The solution chosen should maximize the sludge recycling and or recovery of its resources besides implying benefits in terms of social, environmental, or economic aspects compared to other management methods (Kacprzak et al., 2017). Also, if a specific recovered product will be obtained, a market for such product should be explored to evaluate if the potential product would fit in the current market, considering also the market size with potential customers and possible competitors.

Before going deeper into sludge management alternatives, some challenges that sewage sludge presents should be highlighted. First, its low solid to liquid ratio makes its transportation and logistics difficult and expensive (Kurt et al., 2015). Thus, usually thickening, and mechanical dewatering are required to facilitate subsequent processes (Oladejo et al., 2019). Besides that, sludge drying, which is an energy-intensive process, is usually required before its further thermal use for energy generation, as up to 10% MS is required for some technologies (Bennamoun et al., 2013; Kurt et al., 2015). Second, as it was discussed in previous sections, sewage sludge might contain several organic and inorganic pollutants that should be taken into consideration when choosing the most appropriate management method. In fact, apart from the gradually more stringently regulated heavy metals, other organic pollutants are nowadays in the focus of discussion. Some of the most concerning organic compounds in sludge considered in reference documents as the end of waste criteria for biodegradable waste subjected to biological treatment are: polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB), dioxins, and furans (PCDD/F), absorbable organic halogens, linear alkylbenzene sulphonates, Nonyphenol and ethoxylates and Di(2-Ethylhexyl)phthalates (Saveyn and Eder, 2014; Mininni et al., 2015; Kacprzak et al., 2017; Colón et al., 2017) and more recently other emerging contaminants such as pharmaceuticals, personal care products (PPCPs) and endocrine-disrupting compounds together with antibiotic resistance genes (ARGs) (Dubey et al., 2021; Markowicz et al., 2021). Considering the current demands on the inclusion of the compounds under discussion in the regulatory framework, the appearance and certain concentrations of them in sludge might limit their best valorisation for agricultural purposes.

1.3.3.1 DIRECT DISPOSAL OF SLUDGE TO SOIL

In general terms, direct land application of sewage sludge has been the traditional low-cost way of management of sludges and it does not require any specialized equipment (Bicudo et al., 2011, Bora et al., 2020). Around 40% of the sludge produced in Europe is used for agricultural purposes (Kacprzak et al., 2017). In fact, it was the preferred or widespread practice for some European countries as France, Spain, Italy, Germany, Ireland, Sweden, Austria, Czech Republic, etc., while in the Netherlands, Belgium, or Switzerland is not significantly practised (Minini et al., 2015; Kacprzak et al., 2017).

This practice returns nutrients to the soil, improves the biological properties of soils, and their structure by increasing its organic matter content (Hudcová et al., 2019). However, the potential pollution via toxic elements or transmission of pathogens due to this practice seems to be a major threat to health, either human, animal, or environment itself (Minini et al., 2015; Markowicz et al., 2021). In this context, agricultural use of sewage sludge might be responsible for bioaccumulative trend of certain antibiotics and endocrine disruptors which could be introduced back into the food chain (Bora et al., 2020). In fact, land application of sludges is regulated by the Council Directive 86/278/EEC in European Union and the USEPA 40 CFR Part 503 in the USA (Mantovi et al., 2005). European Directive intended the encouragement of the use of sludge in agriculture, guaranteeing its safe use, mainly limited by the content of heavy metals in sludges and soils susceptive of their application (European Commission, 1986). However, as it was mentioned before, the sludge directive is certainly outdated and most of the European countries already set more stringent limitations for their agricultural use, considering also, sometimes, other pollutants of emerging concern (Colón et al., 2017; Hudcová et al., 2019). Consequently, due to the increasing concern on the risks that this practice might pose, this practice seems to be gradually being discouraged or even banned, particularly for food production purposes (Buta et al., 2021; Santos et al., 2021).

From the economic point of view, transportation of dewatered sludge might be also an important drawback. Dewatered sludge still presents a high water content, which limits the maximum distance in which sludge could be applied (Kehrein et al., 2020). It must be clarified, however, that agricultural use of sludge is considered a low-cost alternative, compared to the rest of the solutions described later in this section.

1.3.3.2 SOLUTIONS FOR ENERGY RECOVERY FROM SLUDGE

Given the increasing environmental and health concern due to the presence of potentially harmful substances in sludge (pathogens, pharmaceuticals, hormones, heavy metals, and persistent organic pollutants), thermochemical alternatives for energy and fuel conversion are considered nowadays attractive (Syed-Hassan et al., 2017). However, as mentioned before, the high moisture content of sludge greatly limits its efficient thermochemical use.

Energy recovery from sludge is regarded as an attractive method for utilising the increasing quantity of sludge (Rada et al., 2009; Tambone et al., 2011).

- Incineration

The incineration process is a well-established technology that aims for the complete oxidation of organic compounds at high temperatures (reaching 800-900ºC in the furnace in the case of fluidized bed reactors), allowing, in turn, the reduction of waste volume. In this case, sludge is burned in a combustion chamber in the presence of oxygen in excess (Tyagi and Lo, 2013; Syed-Hassan et al., 2017). CO_2, H_2O, and ash are the only intended products although other trace pollutant gaseous emissions (particulate material, NOx, SOx, heavy metals, PAHs, PCDD/F, chlorinated compounds, etc.) might appear and must be monitored and treated (Adar et al., 2016; Syed-Hassan et al., 2017; Gao et al., 2020). In this context, this alternative would be in line with the Council Directive 2009/28/EC on the promotion of energy from renewable sources and the hierarchy defined in the waste framework directive (Directive 2008/98/EC), (Tyagi and Lo, 2013; Ponsá et al., 2017). The Incineration Directive 2000/76/EC (European Commision, 2000) and Directive regarding industrial gaseous emissions 2010/75/EC (European Commission, 2010) establish the regulatory framework affecting this technological solution.A considerable volume of ash is produced after combustion thatshould be managed accordingly to its quality and degree of risk. Several uses have been explored for the innocuous ash being the most accepted one its use as supplementary cementitious material (Gao et al., 2020). Agricultural use of ash is also an alternative given their likely high phosphorus contents, although the heavy metals concentration effect in the process is a major drawback of this alternative (Raheem et al., 2018). Besides, the bioavailability of phosphorus in the ash for its efficient use by plants remains unclear (Meng et al., 2019). In this regard, some European countries are prioritising this sludge management solution, for instance, Austria, Germany, and the Netherlands. However, according to the latest national regulations and in line with the current circular economy action plan, some countries (Switzerland, Austria and Germany) also made mandatory phosphorus recovery in certain types of WWTPs (Hudcová et al., 2019; Santos et al., 2021).

The typically high water content limits enormously the energetic and thus, the economic feasibility of the process. In fact, reduction of moisture content below 30% is required to achieve a positive balance, although combustion can occur with a wider range of moisture content in sludge (41-65% MC), although the supply of auxiliary fuels is often required (Syed-Hassan et al., 2017; Oladejo et al., 2019).

Lastly, following the strict gaseous emission limits established in the industrial emissions Directive, complex flue gas treatment is required, which usually requires chemicals and expensive technologies (Oladejo et al., 2019; Gao et al., 2020). Operational costs derived from auxiliary fuels required also increase the overall costs of this technology (Syed-Hassan et al., 2017).

- Anaerobic digestion

Anaerobic digestion is the most popular sludge stabilisation technology in the market, and also the most widely used energy recovery method for sludge at different scales (Tyagi and Lo, 2013). It does not require sludge dewatering (Oladejo et al., 2019). In this process, organic compounds contained in sludge are treated in an anaerobic environment and are converted biologically into biogas that contains methane, carbon dioxide, and traces of other gases (Vatachi, 2019). The conventional biogas with an average methane content of 60% contains 5.8-7.5 kWh Nm^{-3} (Piao et al., 2016; Volschan Junior et al., 2021). Moreover, considering that about 80% of the biodegradable COD fraction in the sludge can be converted into harvestable biogas in completely mixed reactors, 16-20 kWh of electricity per inhabitant and year could be roughly produced from sewage sludge (Volschan Junior et al., 2021). Nevertheless, anaerobic digestion usually requires long retention times (7d- 5 weeks) and the conversion efficiency of organic carbon can be rather low (about 40-70%) (Oladejo et al., 2019; Bora et al., 2020). Anaerobic digestion also produces a digestate as a by-product that should be managed appropriately. Usually, this digestate is rather stable and its content in organic matter is high; thus, its main use is the agricultural application, although strict pollutant control is required for this practice.

From the economic point of view, anaerobic digestion is more advantageous than other thermochemical conversion technologies, although investment costs are still high (Oladejo et al., 2019; Gao et al., 2020). Currently, most studies focus on chemical, physical and biological sludge pre-treatment technologies to enhance biogas and methane production (Volschan Junior et al., 2021). The energy balance and economic feasibility of an anaerobic digestion system is highly dependent on its scale and the conversion efficiency of the carbon into methane. In this context, several innovative biogas upgrading technologies are being investigated and implemented to enrich the biogas produced towards biomethane to inject it in the natural gas grid (Deschamps et al., 2021). However, biogas upgrading technologies require usually high investment costs; and some of them are recognised to be energy-intensive (Kvist and Aryal, 2019; Baena-Moreno et al., 2020). Additionally, if the biogas or biomethane produced will be used off-site, its transportation requires pressurisation, which might turn into increasing the price of the recovered resource over the price of natural gas (Kehrein et al., 2020).

- Pyrolysis

Pyrolysis is the thermal conversion of organic wastes in oxygen-deficient atmosphere (usually in N_2 and CO_2 atmosphere) at 350-900ºC (Ponsá et al., 2017; Raheem et al., 2018; Gao et al., 2020). Three products can be obtained from the pyrolysis process making a process close to “zero waste”: i) a bio-oil rich in aliphatic hydrocarbons, aromatic hydrocarbons, long carbon chain organic acids and alcohols (Adar et al., 2016). The bio-oil is the product that is normally intended to maximise, can be used as an energy source or substrate for chemicals production, a solid product or char that can be used as a solid fuel, or absorbent and a gaseous stream with no apparent specific use, although its heat can be recovered (Tyagi and Lo, 2013; Ponsá et al., 2017; Raheem et al., 2018; Oladejo et al., 2019; Gao et al., 2020). The further use of the char produced depends on its energy, nutrient and heavy metals content, being the last one a serious concern nowadays (Raheem et al., 2018).

Conversely to incineration, low operating temperatures minimise melting and evaporation of heavy metals and thus, the complexity of post-treatment processes for flue gases is lower (Raheem et al., 2018).

Given the processing equipment required for this technology and the suboptimal stage of its performance (technology still rather immature), its economic feasibility might be challenging (Raheem et al., 2018). First, the drying requirement of sludge is higher than for other thermochemical technologies (to below 10% MC) (Oladejo et al., 2019; Bora et al., 2020), and high moisture contents might decrease the quality of products (Syed- Hassan et al., 2017; Gao et al., 2020). Related to that, energetic self-sustainability of sewage sludge pyrolysis is still on research and use of catalysts and co-substrates is being studied in the last years (Pan et al., 2021). Second, given the complexity of processing equipment, the economic feasibility might be limited. However, it seems that bio-oil and biochar commercialisation could be possibly enough to support the system economically, provided that production yields and quality of products are maximised (Raheem et al., 2018). Still, how the products obtained from sewage sludge pyrolysis can be available in the current markets is still unclear (Gao et al., 2020).

- Gasification

Gasification is a thermochemical technology carried out at temperatures between 800 and 1000ºC in oxygen-deficient environment and very short residence periods (Tyagi and Lo, 2013; Syed-Hassan et al., 2017; Kehrein et al., 2020). It can be considered an extension of the pyrolysis process where a gasifying agent is needed (air, oxygen, steam, CO2, and mixture of air-stream) to transform organic material into an energy-rich gaseous mixture mainly composed of H_2, N_2, CO, CO_2 and CH_4 (Kalogo & Montheit, 2012; Tyagi and Lo, 2013; Raheem et al., 2018; Oladejo et al., 2019; Gao et al., 2020). Apart from the energy-rich gaseous mixture or syngas (synthetic gas) (containing 1.1-7.8 kWh m^{-3}) obtained heat is also produced, which can be used to generate power and process heat (Adar et al., 2016; Oladejo et al., 2019). As mentioned, the syngas obtained is highly energetic and it can be used for heating or generation of electricity through a heat engine or can be further processed to produce chemicals or liquid fuel (Syed-Hassan et al., 2017; Oladejo et al., 2019). However, its quality is highly dependent on the sludge properties that are valorised through this technology, being the process particularly sensitive to high ash contents, which could lead to sintering, agglomeration, and clinker formation (Oladejo et al., 2019).

From the material recovery point of view, the high operating temperatures applied in gasification were demonstrated to promote phosphorus mitigation into the gaseous phase whereas during pyrolysis phosphorus is most likely to be retained in the char produced (Meng et al., 2019).

Implementation of gasification technology for sewage sludge faces some challenges that might hamper its economic feasibility. Sewage sludge with too high moisture content might be challenging as it lowers the overall process efficiency and syngas quality. Thus, usually, a moisture content below 30% is recommended for gasification (Syed-Hassan et al., 2017). Accordingly, the drying step is a highly energy-demanding step and the higher drying need increases the overall cost of the system (Raheem et al., 2018). Apart from that, if sewage sludge-derived syngas is used as a fuel, it needs to be cleaned as it contains undesirable impurities (mainly tars and nitrogenous and sulphurous compounds) that may damage fuel cells, engines, or turbines (Syed-Hassan et al., 2017; Raheem et al., 2018; Oladejo et al., 2019). Some authors demonstrated the techno-economic feasibility of the gasification process of sludge with promising results (You et al., 2016; Alves et al., 2021), although those authors did not include sludge drying steps and studies were conducted considering co-gasification processes.

Gasification is a complex process which makes technology more expensive (Gao et al., 2020).

- Supercritical Water Oxidation

Supercritical water oxidation is a thermochemical process that occurs in a water phase at high temperatures and pressures (Kalogo & Montheit, 2012; Tyagi and Lo, 2013). When water temperature and pressure are simultaneously raised and the critical point of water is reached, a change of phase takes place in which water becomes “supercritical” (Kamler & Soria, 2012), which presents some unique characteristics such as the ability to quickly dissolve oxygen. Thus, when applied to sludges, very fast and complete oxidation of organic compounds occurs (up to 99.9%) (Tyagi and Lo, 2013; Ponsá et al., 2017). Once the process is finalised, inorganic compounds, including highly appreciated phosphorus compounds or coagulants, can be recovered from the ash (Kalogo & Montheit 2012, Tyagi and Lo, 2013). Energy can also be recovered by heat exchange in the reactor or from the effluent flow from the reactor, although the process itself is not able to produce any forms of biofuels (Kalogo & Montheit 2012; Bora et al., 2020). A major advantage of this technology is that there is no sludge dewatering or drying requirement to feed supercritical water reactors (Adar et al., 2020)).

As a major drawback, the high complexity of the process limits the scalability of the process, together with economic feasibility. Appropriate reactor design and construction to withstand high pressure, temperatures, and active corrosion difficult the development of the technology and increase its costs (Ponsá et al., 2017).

- Hydrothermal liquefaction and supercritical water gasification of sludge

Given the limitation of pyrolysis and gasification processes applied to wet samples, the two processes carried out under supercritical conditions have been suggested as promising technological alternatives for sludge valorisation. These technologies have attracted extensive attention due to their high energy recovery efficiencies and environmental benefit (Chen et al., 2020a; Do et al., 2020). The unique properties of water under supercritical conditions (above 374 °C and 22.1 MPa), can provide a homogeneous reaction environment to the sludge that can be transformed either into a hydrogen-rich synthetic gas or bio heavy-oil (Adar et al., 2020; Do et al., 2020; Qian et al., 2021). These innovative alternatives avoid the expensive drying step required for other thermochemical alternatives mentioned. Besides, in the case of supercritical water gasification, the syngas produced does not require further post-treatment (Adar et al., 2016; 2020; Zhang et al., 2021). Compared to previous thermal processes these technologies permit very high energy conversion efficiencies (40–55 %) for hydrothermal liquefaction and up to 62% for supercritical water gasification processes enhanced by the use of catalysts) (Adar et al., 2020; Do et al., 2020), obtaining highly energetic products with 30-35 MJ kg^{-1} and 15 MJ m^{-3} in bio heavy oil and syngas, respectively. Besides, the solid outlet produced is rich in nutrients (N, P, K, Mg, Ca) and can be used as bio-based fertiliser. Equal to the rest of the thermochemical processes mentioned, pathogens and organic toxic compounds are removed or degraded.

Recently published studies are mainly related to adjusting operational parameters and use of catalysts to enhance process efficiencies (Adar et al., 2020). However, according to the last author, works carried out in representative scales and treating real sludge are scarce. Considering the last, supercritical water gasification technology is clearly far from being implemented at real scale. Chen et al., (2020a) reviewed the existing works concerning supercritical water gasification, pointing out 3 works at industrial scale dealing with sewage sludge.

1.3.3.3 SOLUTIONS FOR MATERIAL RECOVERY FROM SLUDGE

- Composting

Composting of wastes is another widely applied treatment process. It is defined as the aerobic biological decomposition of organic substrates which produces metabolic heat allowing the development of thermophilic temperatures leading to the sanitization of the organic waste treated. The process, when properly managed, should guarantee a well stabilised product, free of pathogens and plant seeds (Colón et al., 2017) as well as an important volume reduction during the process.

As a matter of a fact, given the recent concern on the effect of emerging pollutants on the environment and human health, studies related to the mitigation of such pollutants through composting has been increasingly studied in the last years (Ezzariai et al., 2018; Zheng et al., 2020; Lü et al., 2021).

First, one of the main limitations of composting is related to land requirement, given its long retention time until the organic waste is fully stabilised to allow its use in agricultural land. Second, the usually low concentration of nutrients in sludge compost compared to mineral fertilisers is another weakness that should be considered (Kominko et al., 2019; Rehman et al., 2020). Third, similar to what is claimed for direct agricultural application of sewage sludge, heavy metal content in compost and most probably also organic pollutants in the future, may forbid its agricultural application. In fact, and as it was mentioned before, sewage sludge is not considered appropriate feedstock for fertilising products formulation, not even for the category of compost (CMC3) in the European regulation of fertilising products (2019/1009). Forth, the willingness of European farmers to switch from mineral to organic fertilisers, including compost. This limitation was reviewed by Chen et al., (2020b). Summarising the findings made by the authors, fertilising product cost, the efficiency of nitrogen content, and stable availability of the product seem to be some of the main factors affecting this switch. Additionally, the variable origin of the feedstocks used for their production raises doubts on the reliability, safety and efficiency of such products. This last limitation could be overcome by boosting the standardisation of the bio-based products and their promotion by their inclusion in relevant regulations (Chen et al., 2020b).

Composting is a simple process that requires relatively small capital investment and it is recognised to be an efficient and cost-effective method for sludge treatment (Zheng et al., 2020). However, operational costs due to high energy demanding forced aeration are considerable. Besides, the product obtained (compost) is considered a low added-value product which presents a rather low market price (Alibardi et al., 2020). Finally, the product might have low acceptance among stakeholders, mainly due to the potential presence of pollutants.

- Other technologies for the production of added-value products

Activated sludge has been demonstrated to be a suitable substrate for the specific production of several added-value products such as enzymes, biochemicals, bioplastics, or biopesticides.

Enzyme and biochemicals production from organic wastes, for instance, could be helpful for some industrial sectors (pharmaceutical, food, cosmetics or detergent industries).. Some works demonstrated the feasibility of producing a wide range of products: cellulases, ß-glucosidases, several types of short-chain carboxylic acids (caproic-, butyric-, lactic- acids) (Iglesias-Iglesias et al., 2021; Jing et al., 2021). As a counterpart, the high specific activity of enzymes and purity of the overall products obtained is required for the industrial uses mentioned and purification and concentration steps are usually needed (Raheem et al., 2018).

Bioplastics are produced from polyhydroxyalkanoates (PHA) which are naturally produced by some bacteria as carbon and energy reserve. PHA are polyesters of hydroxy alkanoic acids and are known to be biodegradable polymers which have been investigated as an alternative for petroleum-derived plastics to produce packaging films and disposable products (Tyagi and Lo, 2013). In recent years, literature regarding PHA production using municipal sewage sludge as the carbonous substrate has increased (Kumar et al., 2021), being the most recent works focused on the optimisation of process parameters to maximise recovery yields (Kumar et al., 2018). Compared to conventional substrates, the benefit of using residual feedstocks for this aim is the cheapening of its production costs (Bassi et al., 2021). However, although being technically feasible and environmentally more sustainable than the production of conventional plastics, their production costs are still high and production yields rather low. Furthermore, bioplastics production from sludges seems not to be yet cost-competitive and their market potential is limited (Raheem et al., 2018; Kehrein et al., 2020). Therefore, specific regulative actions should be considered to support these changes in the plastic market (Bassi et al., 2021).

Finally, beneficial metabolic products with bio-pesticidal or bio-stimulant potential have been produced from sludges. This kind of compound is functional and beneficial for agricultural use with no toxic residues (Raheem et al., 2018). For instance, recent works have demonstrated the production of edaphic biostimulants from the fermentation processes of sewage sludges (Rodríguez-Morgado et al., 2019; Tejada et al., 2022). Edaphic biostimulants show bioremediation effect as they can stimulate the microorganisms in the soils leading to the degradation of xenobiotic substances such as pesticides and hydrocarbons. Besides, some compounds such as betaines, chitin, humic substances, protein hydrolysates, and other bioactive substances have been demonstrated to present biostimulating effects in plants (Huang et al., 2021a; 2021b). Thus, bioconversion processes of different organic wastes with the subsequent extraction of the compounds mentioned have been raised as an alternative to valorise organic wastes and obtain compounds with added value. Similarly, the production of compounds with bio-pesticidal effect from sewage sludge has been also assessed in the literature where specific bacterial strains can produce intracellular inclusions containing insecticidal proteins (Zhuang et al., 2011; Chang et al., 2012). The use of bio-pesticides can lead to significant improvements in entomotoxicity and they are already being applied in agronomy, forestry, and public health sectors (Tyagi and Lo, 2013). The use of sewage sludge as cultivation media might result in a cost-effective alternative, although pre-treatment of sludge is most probably needed. Production yields are still rather low and highly dependent on process control and purification steps are required before entering in the global pesticide market (Raheem et al., 2018).

1.4 INNOVATIVE SOLUTION FOR SLUDGE VALORISATION

As already described in the previous section, there is a wide range of sludge management practices and technologies that could be appropriate in specific contexts.

Currently, innovative technologies are being developed aiming at the efficiency of wastewater treatment plants; reducing the amount of sludge produced together with the energy consumed and providing, therefore, clear environmental and economic benefits (Longo et al., 2017; Conca et al., 2020; Da Ros et al., 2020; Larriba et al., 2020). The change of the conventional linear wastewater treatment plants in which sludge is an unavoidable by-product towards a more integrated wastewater resource recovery concept permits the optimization of the valorisation strategies for all the secondary by-products produced. As a consequence, an attractive product with specific characteristics (sludge) can be expected and its valorisation route will be designed accordingly.

Additionally, in an existing WWTP upgrading context, some innovative resource recovery technologies are presented as easy to implement, where no major infrastructural modifications are required to valorise waste streams such as wastewaters, and more particularly sewage sludge.

Bearing in mind the paradigm described above, two innovative, easy-to-implement sewage sludge valorisation alternatives are presented in the current work. The solutions proposed can be implemented with minor changes in existing sewage sludge composting facilities and can maximise resource recovery (either energy or material recovery) in the sewage sludges. Selection of the most appropriate technology of the two innovative technologies presented will depend on the maximum potential of the sludge in study, either attractive for energy purposes or agronomic purposes.

On one hand, biodrying technology is suggested as an efficient alternative for those sludges with attractive energy content and those that due to the presence of certain toxic elements could not be valorised for agronomic uses.

On the other hand, advanced composting is suggested as the best valorisation route for those sludges rich in nutrients (particularly nitrogen and phosphorus) with no significant content of inorganic toxic elements, such as heavy metals.

1.4.1 BIODRYING

Biodrying technology is an aerobic biological process that uses the excess of heat produced during the decomposition of easily biodegradable organic matter to remove as much as of its water possible (Winkler et al., 2013). This process can be considered a composting-like process. However, its operation time is intended to be shorter (7-15 days) and its main objective is obtaining an end-product for energy recovery instead of waste biostabilisation (Navaee-Ardeh et al., 2010). In biodrying, most of the organic matter contained in the raw solid waste should be preserved in the final material, producing a biomass fuel with a considerable gross calorific value (Huiliñir and Villegas, 2014).

As a general description of the biodrying profile, it consists generally of 3-4 stages: (i) a first temperature increasing phase in which microbial community is activated and start to degrade most biodegradable organic compounds and produce metabolic heat, (ii) a thermophilic stage in which temperatures increase above 40-45ºC and up to 70-75ºC and very intense moisture removal is intended to occur and a (iii) mesophilic stage in which sludge drying rate slows down, decreasing sometimes even to below-mesophilic range of temperatures leading to (iv) a cooling stage. Figure 1-6 shows a typical profile of a biodrying process of sewage sludge where the stages mentioned are identified. In the figure, the decrease of moisture removal in the bulk mixture is also highlighted, which is together with the temperature profile, a key parameter in the biodrying performance monitoring.

The production of a high-quality biomass fuel from biodrying is the main target of the process. The product should be ready to be burned in a biomass boiler and enable the maximum energy recovery possible. As lower the moisture content (MC) and higher the calorific potential of the product, as the more efficient the combustion process and energy recovery potential it would present. Therefore, the moisture content of the product and its calorific value are the key parameters that are usually intended to be optimised in the biodrying process.

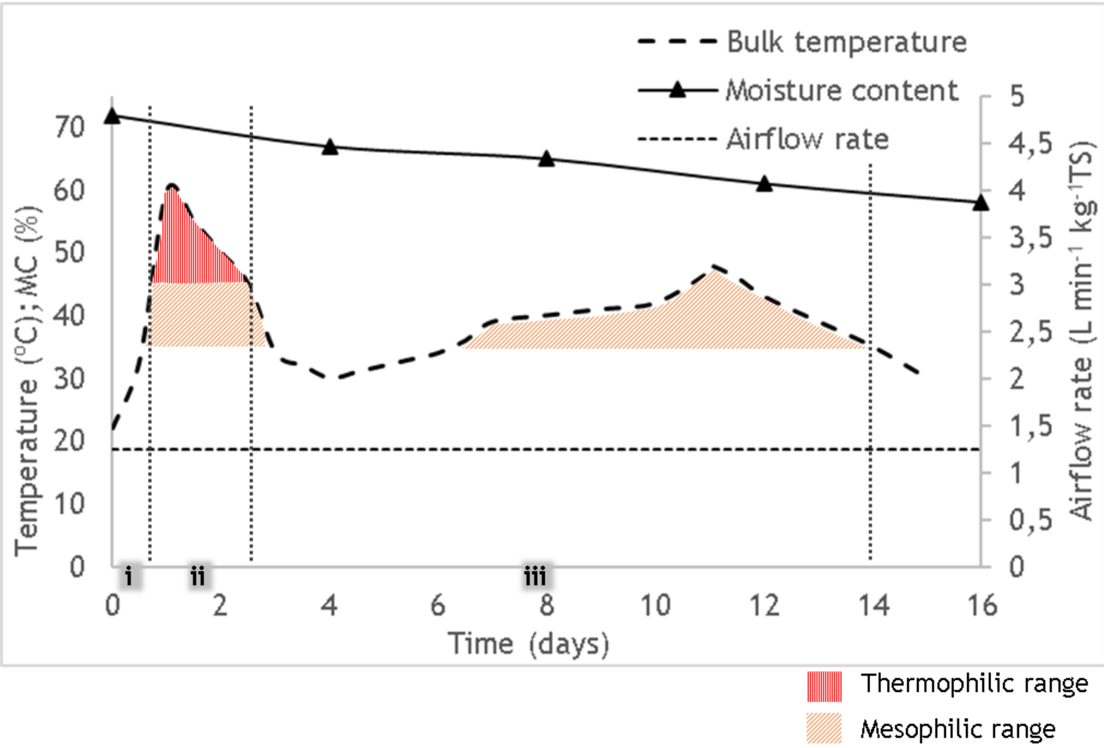


Figure 1-6. Typical profile of sludge biodrying process (adapted from Hao et al., 2018). Every stage mentioned is identified in the figure as (i) temperature increasing phase, (ii) thermophilic stage, and (iii) mesophilic stage.

Biodrying of municipal solid waste has been thoroughly studied and it has been even applied at industrial scale to produce solid recovered fuels (SRF) from mechanical and biological pre-treatment (MBT) facilities (Velis et al., 2009; Tambone et al., 2011; Ragazzi et al., 2011; Evangelou et al., 2016). Sewage sludge, though, presents some difficulties over other organic wastes for its valorisation through biodrying, being the most significant its low porosity and high moisture content, which can hamper the proper air diffusion throughout the raw material.

For the process optimisation, some initial and operational parameters are assessed in the literature, being those parameters equally critical for both MSW and sludge biodrying. On one hand, initial adjustment of material structure is crucial to provide an optimal aerobic environment for an appropriate process, particularly given the low porosity of sludges (Yang et al., 2014). This adjustment is usually done by mixing the sludge with a bulking agent (B.A.) in a known mixing ratio. Conventionally, lignocellulosic agricultural and forestry by-products, such as sawdust, straw, woodchips, rice husks, and pruning wastes are used as bulking agents (Zhao et al., 2010; Huiliñir &

Villegas, 2014; 2015; Yang et al., 2014; Cai et al., 2018), although other authors also assessed the use of dried sludge (either conventionally or through biodrying) as bulking agent (Yang et al., 2014). Initial mixture adjustment mainly implies modifications in the moisture content of the mixture, which is directly related to its free air space (FAS). However, most of the authors studying initial mixture parameters focus mainly: (i) on the effect of its moisture content in the drying performance (Yang et al., 2014; Huiliñir & Villegas, 2015; Ma et al., 2016); (ii) the effect of the typology of bulking agent used (Yang et al., 2014). It is generally accepted that the proper adjustment of initial moisture content is critical for the effective performance of biodrying. An excessive moisture content would fill pore spaces with water, hampering the efficient aeration of the mixture (Sadaka et al., 2012; Villegas & Huiliñir, 2014), leading to delayed temperature increases and lower maximum temperatures, achieving ultimately limited drying performance (Yang et al., 2014; Villegas & Huiliñir, 2014). Apart from that, given the inefficient aeration of the mixture, equal to composting processes, an excess in moisture content could lead to increased aeration costs and the development of unwanted anaerobic conditions (Richard et al., 2002;). Conversely, a lower than recommended moisture content could make difficult the maintenance of heat inside the matrix (Barrena et al., 2011a), shortening particularly the thermophilic phase necessary for the efficient biological drying.

On the other hand, among performance-related parameters, airflow rate and turning are the most affecting parameters in biodrying. First, turning allows the homogenisation of bulk mixtures, especially when working with sewage sludge which tends to compaction (Zhao et al., 2010; Cai et al., 2012). Redistribution of moisture content and biodegradable organic compounds may also occur after turning, and thus, mixtures can be sometimes re-activated for that reason, although excessive turning might also provoke a drop in temperatures, leading to inefficient drying performance.

Second, the airflow rate effect is rather complex as it affects the microbial activity, outlet air and mixture temperature, rate of vaporization, and its water holding capacity (Sharara & Ahn 2012). Mainly, aeration is closely related to temperature profile achieved by means of (i) affecting the satisfactory microbial activity leading to biodegradation and heat production and (ii) affecting the temperature cumulation or loss leading to a satisfactory or unsatisfactory moisture removal and ultimately, biodrying process. Therefore, the delicate balance between airflow rate supplied and heat cumulation is extremely important. A too low aeration rate does not supply sufficient oxygen for microorganisms, favours biodegradation, and can lead to the development of undesirable anaerobic conditions (Navaee-Ardeh et al., 2006). Also, excessively low aeration would facilitate the early saturation of the airflow with water vapor and would lead to inefficient moisture removal. Conversely, excessive aeration can cause delayed microbial activity, lower pile temperatures by cooling effect not reaching thermophilic conditions resulting in unnecessary power consumption and ineffective drying performance (Navaee-Ardeh et al., 2010; Cai et al., 2013).

Compared to composting, air-flow rates in biodrying processes are higher to facilitate water removal and reduce organic matter consumption (Sadaka & Ahn, 2012; Huiliñir & Villegas, 2014). Air-flow values between 0.5 and 6 L $min^{-1}kg^{-1}VS$can be typically found for effective biodrying processes (Zhao et al., 2010; Huiliñir & Villegas, 2014; 2015). Although adaptive aeration systems have been also successfully assessed (Cai et al., 2013; Zhang et al., 2015). In the current situation, the biodrying process of sewage sludge is not fully understood and it is far from being optimised.

As mentioned before, biodrying is already being applied for some wastes generated at MBT pre-treatment facilities to produce SRF. There might be some organic wastes, and in particular, sludges, that don't gather the properties to be valorised for agricultural purposes, mainly given the presence of organic or inorganic pollutants or lack of market for this kind of fertilising product. In those cases, and provided that their higher calorific value is high enough, or at least equivalent to conventionally used SRF or agroforestry wastes, biodrying of this sludge could be a promising alternative. Biodrying technology is expected to have a considerable impact in the water and waste sectors given the recent trend to limit the agricultural application of sewage sludges, even after appropriate biological treatment. It is important to highlight that the implementation of biodrying technology in existing sludge composting plants is feasible, and seems to require minor modifications of the existing infrastructure.

1.4.2 ADVANCED COMPOSTING

Composting has been widely applied to mainly stabilise and sanitise sewage sludges before their application as an organic amendment into soils (Colón et al., 2017). Composting of sewage sludge shows a profile that could present similarities with the profile of sewage sludge biodrying. It also presents usually four stages including (i) a temperature increasing stage, (ii) a thermophilic stage, (iii) a mesophilic stage, and (iv) a cooling stage, being the curing stage the one differing from the biodrying process. The length of each stage varies with the sewage sludge treated, the working mixture, and the configuration of the system itself, including its working capacity. Figure 1-5 shows a typical profile of sewage sludge composting where the stages mentioned are shown. Regarding initial and operational parameters, critical parameters affecting composting process are quite the same as those mentioned in biodrying. However, given the purpose of maximum biostabilisation of organic matter and sanitisation of the waste of the composting process, biological activity maximisation is required for the successful completion of the process, which could last up to 4-6 months. Regarding parameters related to the initial mixture, it is generally accepted an optimal moisture content between 50- 60% (Haug 1993; Trémier et al., 2009) while a suitable FAS of 50-60% is normally reported for composting processes. The C/N ratio has a decisive effect on aerobic degradation processes as well as in odour emissions of the process (Li et al., 2013).

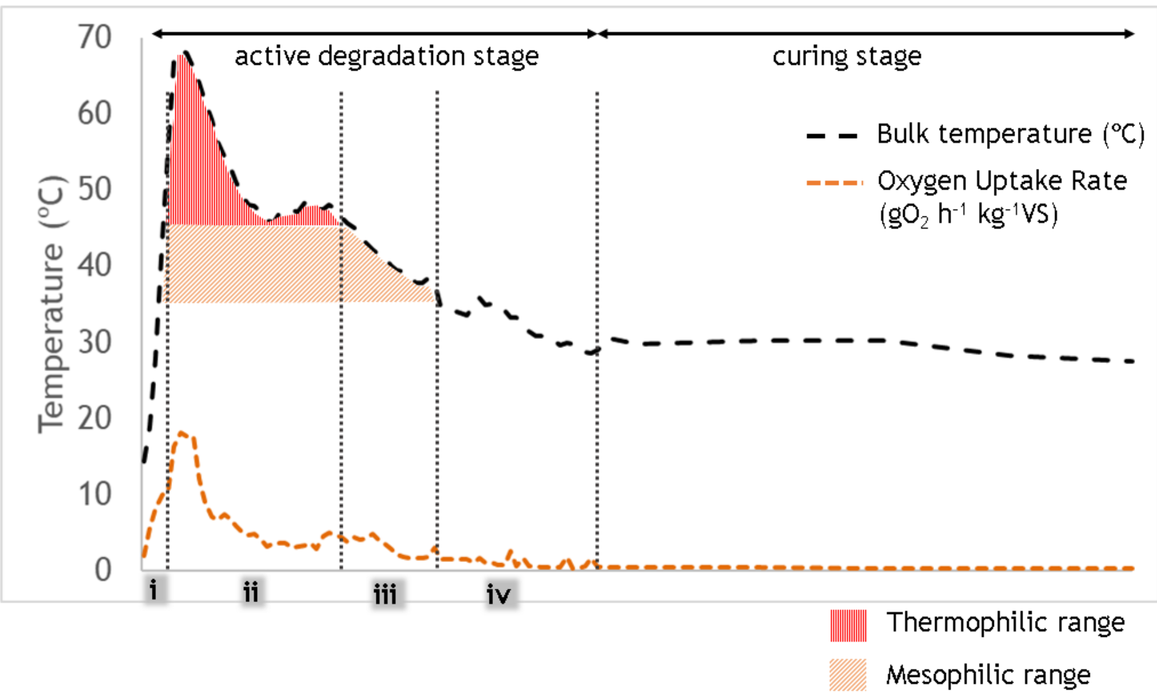


Figure 1-7. Typical profile of sewage sludge composting process.

It is accepted an optimal range of C/N ratio between 25 and 30 (Haug, 1993; Navaee- Ardeh et al., 2010). Values above could mean a limiting content of nitrogen leading to a hampered microbial activity (Li et al., 2013). Likewise, an excessively low C/N ratio can suggest a lack of carbon source and probably increased nitrogen leaching or volatilisation in form of ammonia, generating also undesirable odours and atmospheric pollution (Ogunwande et al., 2008; Puyuelo et al., 2011). Also, pH is relevant for the proper development of composting process and some wastes require its initial adjustment to near neutrality, although a wider range of pH has been successfully used (5.5-9) (Bernal et al., 2009; Navaee- Ardeh et al., 2010).

Analogously to the biodrying process, aeration is a critical operational parameter in composting (Han et al., 2018). Conventionally, composting piles are aerated either actively with forced aeration or passively through turning. The aeration strategies applied in composting affect the temperature profile of the bulk mixture, biodegradation, the biostabilisation level and the overall length for the completion of the process (Kulikowska and Gusiatin, 2015). Typically, in sewage sludge composting processes airflow rates supplied are in the range of 0.1 and 2 L min^{-1} kg^{-1}TS (Yuan et al., 2016; Rincón et al., 2019; Wang et al., 2018). The vast majority of the sludge composting literature found work in continuous air regime (Kulikowska and Gusiatin, 2015; Yuan et al., 2016; Han et al., 2018), although there are some works at industrial scale applying auto-controlled air supply, either dependent on temperature or oxygen content in exhaust gases (de Guardia et al., 2008; Puyuelo et al., 2010; Shen et al., 2011). Both temperature-based and oxygen-based controls rely on the assumption that temperature and oxygen consumption indicate the biological activity of the bacterial community. However, the mentioned strategies usually adjust aeration rates per stage of the process, per certain ranges of fixed values, or fixing some set-points. Airflow rate adjustment strategy based on biological activity feed.b.ack was suggested as a reliable and robust control system that allows the (i) optimisation of energy consumption, (ii) increasing stability of the final product, and (iii) minimisation of GHG emissions (Puyuelo et al., 2010; Maulini-Duran et al., 2013; Puyuelo et al., 2014).

Composting of conventional sewage sludge is widely applied in Europe, although, as previously mentioned, some barriers regarding its safe use and limited fertilising value are arising.

Advanced composting is a process that permits the upgrading of the existing composting in two forms. First, advanced composting adjusts the air supply according to the biological consumption (Oxygen uptake rate, or OUR) required by the bulk mixture, supplying only the air that is strictly needed. This is expected to allow lowering the energy consumption of the process and shortening to the minimum the overall process completion length, which is translated into the economic attractiveness of the process (Puyuelo et al., 2010). However, the challenge that faces advanced composting is linked more to the selection of appropriate feedstock rather than the technology itself. Therefore, second, optimisation of feedstock is necessary for the production of a high-quality product. The quality of this product is assessed in terms of (i) product stability, (ii) nutrient content, (iii) absence of unwanted pathogens, and (iv) presence of toxic pollutants, including those of emerging concern. In this context, the upgrading of the technologies allowing the obtention of sludges with attractive nutritive characteristics and low content of pollutants would permit the production of a safe new-generation bio-based fertiliser ready to be used in agricultural fields, thus fulfilling the requirements for the so-called advanced composting process.

1.5 SMART PLANT

The present thesis was developed in the framework of the European Horizon 2020 SMART Plant project. The SMART Plant project was an innovation action in the program Enabling the transition towards a green economy and society through eco-innovation.

The main target of SMART Plant was the scaling-up in real environment of eco-innovative and energy-efficient solutions to renovate existing wastewater treatment plants and close the circular value chain by applying low-carbon techniques to recover materials that are otherwise lost. In this framework, seven plus two (7+2) pilot systems were implemented and optimized for more than two years in real environment in five municipal water treatment plants. The pilots implemented pursued the maximum resource recovery from wastewaters by upgrading their treatment at three levels: mainstream, sidestream and downstream. Figure 1-7 summarises the technologies assessed in the project in each abovementioned treatment level.

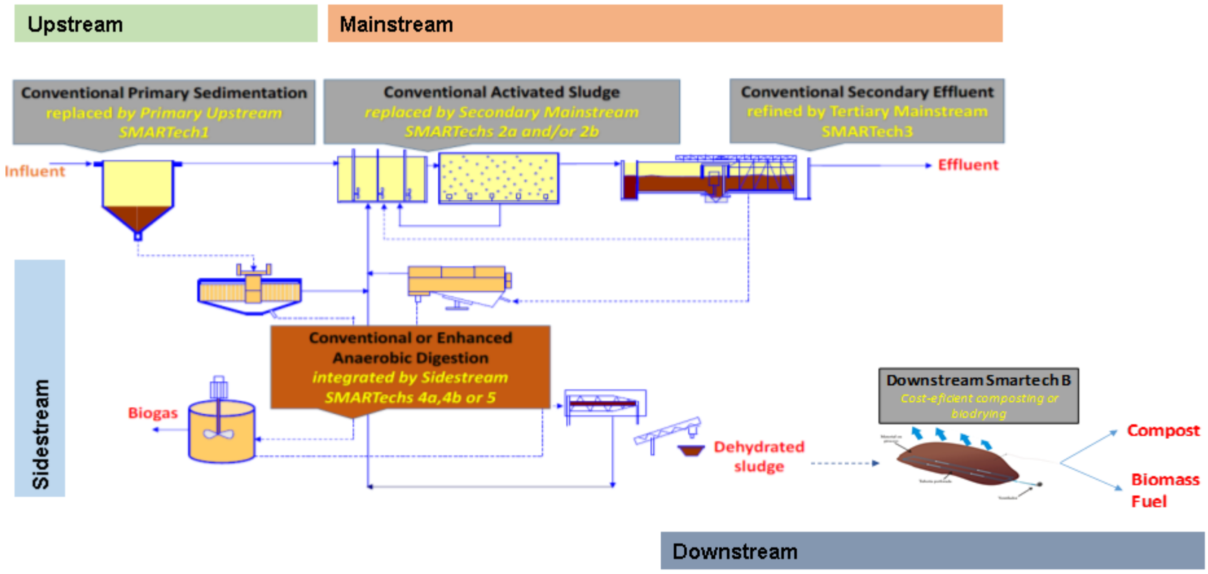


Figure 1-8. Summary of the technological approaches implemented and optimised in SMART Plant project.

As one of the main outputs of the project, a comprehensive SMART portfolio of recovered products comprising biopolymers, cellulose, fertilizers, and intermediates was developed up to the final marketable end-products. Moreover, the feasibility of circular management of urban wastewater and environmental sustainability of the systems were also assessed in the project, through Life Cycle Assessment and Life Cycle Costing approaches as well as the global benefit of the scaled-up water solutions.

BETA Technological Center was involved in SMART Plant project as a participant partner together with a total of 26 partners (19 SMEs and 7 R&D Organizations) from 9 EU countries.

BETA Tech Center was involved in SMART Plant mainly by developing advanced composting and biodrying technologies for the postprocessing of sludges produced within other SMART technologies. Thus, phosphorus-rich dewatered sludge was valorised into nutrient-rich biobased fertiliser through advanced dynamic composting, while cellulose-rich sludge was valorised as an attractive biomass fuel through biodrying. The quality of the products was demonstrated in terms of fertilising value and calorific value, respectively. Environmental and economic sustainability of the technologies and products was also demonstrated in close collaboration with other partners of the project.

2. OBJECTIVES

2. OBJECTIVES

The general objective of this thesis was to increase the resource recovery (both, energy and materials) from sewage sludge through innovative technologies (biodrying and advanced composting) to produce biomass fuel (BMF) and bio -based fertilisers (BBF) to ultimately increase the circularity of WWTP.

The achievement of this general objective was tackled by means of the following specific objectives:

- To establish an updated state of the art on the topic and identify potential challenges of the technologies, feedstocks and products that would allow their upgrading and entry into the market.
- To construct a pilot installation able to monitor relevant operational parameters and develop and validate two control systems for the satisfactory completion of biodrying and advanced composting of sewage sludges.
- To identify and select the sludges with maximum energy recovery potential for biodrying and with maximum nutrient recovery potential for advanced composting.
- To satisfactorily perform biodrying and advanced composting trials of selected sludges and to obtain representative products.
- To improve biodrying performance by implementing and assessing different aeration strategies.
- To develop and validate process monitoring criteria to evaluate the satisfactory biodrying process completion.
- To develop innovative indexes to improve the biodrying process efficiency assessment.
- To evaluate and compare the quality of the products obtained.
- To assess the environmental impacts of the processes studied by means of monitoring of relevant gaseous emissions and energy consumption along the processes.
- To assess the economic performance of both processes at industrial scale by means of developing specific economic models and applying them to potential case studies.
- To identify the overall strengths, weaknesses and opportunities of the processes highlighting also the potential threats that need to be overcome to bring both, technologies and products obtained, to the real market.

3. MATERIALS AND METHODS

3.MATERIALS AND METHODS

3.1 SLUDGES STUDIED AND THEIR SAMPLING PROCEDURE

In order to achieve reliable and impactful results from the sludge valorisation alternatives tested, a set of sludges with different characteristics was assessed. The valorisation alternative suggested for every sludge studied was dependent on the maximum potential for resource recovery that was presenting, either for material or energy recovery.

Raw sludge samples tested were typically taken from composting industrial plants or wastewater treatment plants (WWTP), either municipal or industrial such as pulp & paper mills. Besides, biodrying and advanced composting processes were assessed with innovative sludge fractions produced within SMART Plant project, the so-called cellulosic sludge (CS) and phosphorus rich sludge (PS).

3.1.1 CELLULOSIC SLUDGE (CS)

Cellulosic sludge was collected from the Wastewater Treatment Plant (WWTP) of Geestmerambacht, The Netherlands which serves to 262,000 population equivalents (P.E). Cellvation® cellulose recovery technology was assessed in the framework of SMART Plant project and treats in situ 30-80 m^3 h^{-1} of wastewater. The cellulose recovery system consists of an initial grit and hair removal in a rotating drum filter, to apply after a 350 μm fine sieve (Salsnes Filter, Norway). Fine sieve filter is able to recover a sieved fraction with 4-8% TS (containing up to 80-90% of cellulose and hemicellulose) which is then dewatered in a screw press for subsequent drying through infrared drying to achieve a product based on recovered cellulose that is intended to be marketable for different uses. Apart from the recovered cellulose, a dewatered filter cake from now on called cellulosic sludge is produced. Figure 3-1 shows a diagram of a WWTP where Cellvation® technology is implemented for cellulose recovery. Given its expected high content in cellulosic and hemicellulosic fibres, which present considerable calorific value (up to 18.6 MJ kg^{-1}TS) (Dietenberger & Hasburgh, 2016; Kim et al., 2017).

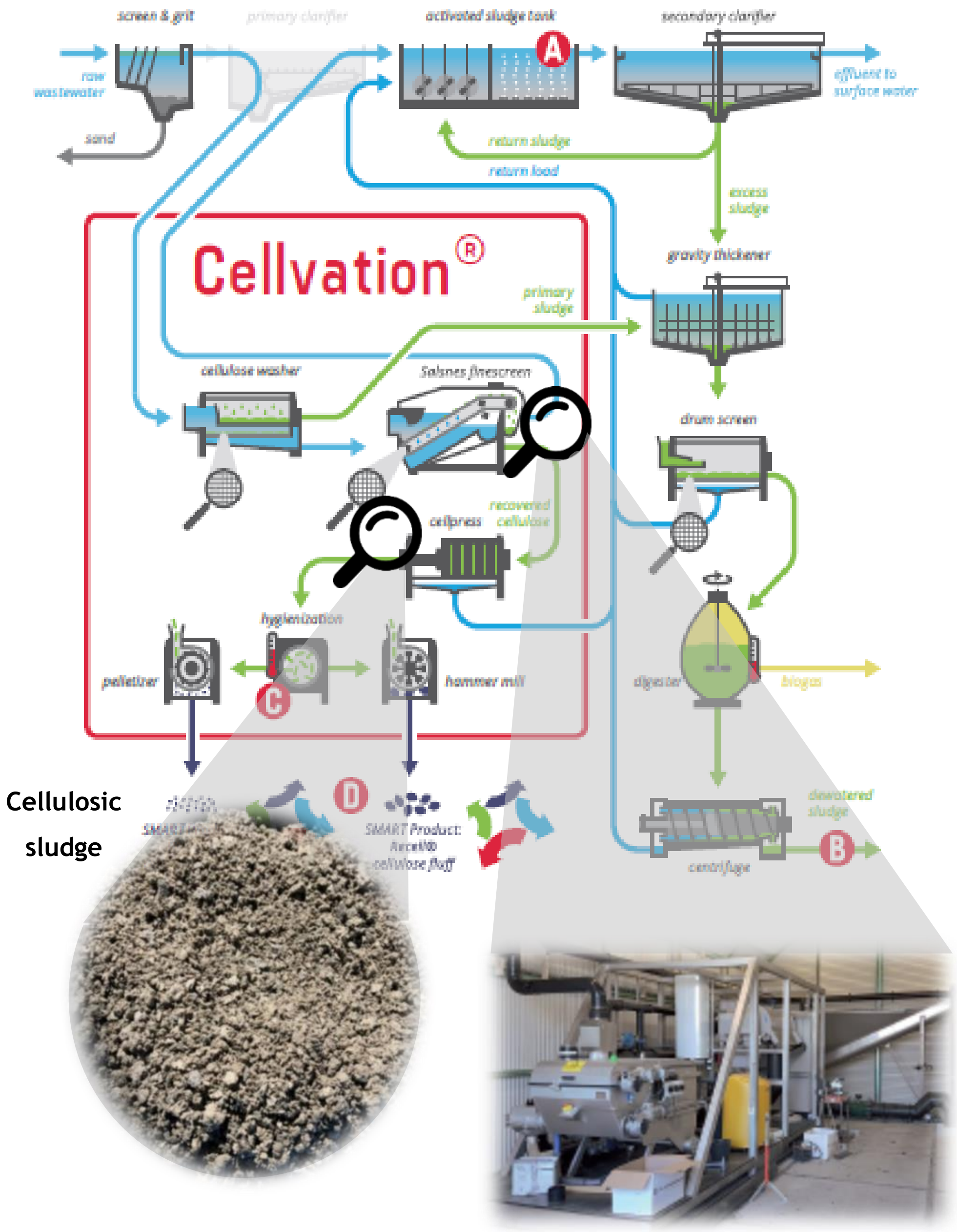


Figure 3-1. Cellvation® technology for cellulosic sludge production.

3.1.2 PHOSHORUS RICH SLUDGE (PS)

Phosphorus rich sludge was obtained from Carbonera WWTP, managed by Alto Trevigiano Servizi (ATS), where SCENA process was implemented and assessed in the framework of SMART Plant project.

SCENA process is an innovative technology implemented in sidestream wastewater line aiming the via-nitrite nitrogen and phosphorus removal from the anaerobic supernatant. Figure 3-4 shows schematically the SCENA system.

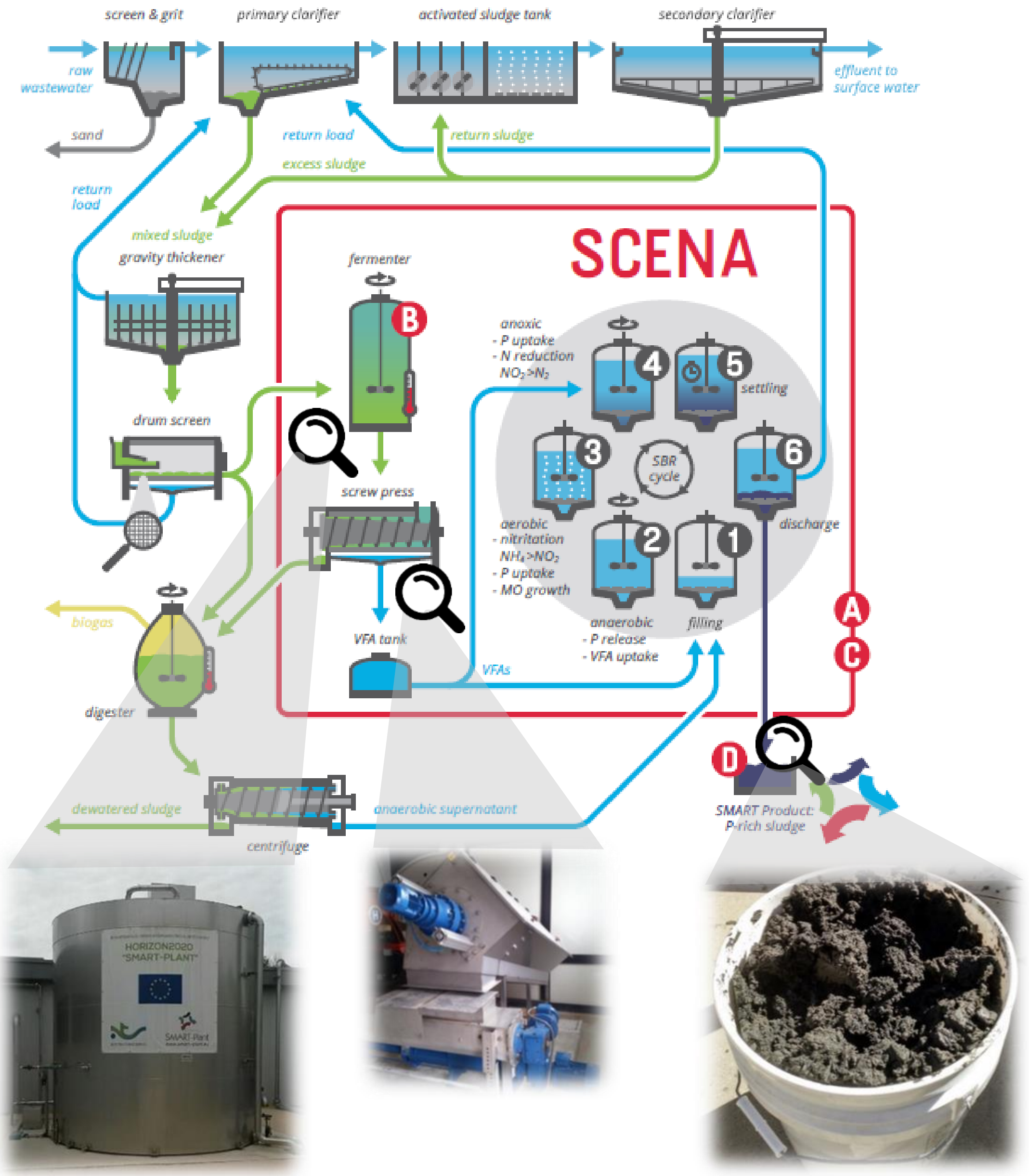


Figure 3-2. Diagram explaining SCENA system and sample treated through advanced composting.

In general terms, SCENA system consists of three main processes: (i) fermentation of thickened sewage (mixed) sludge at mesophilic conditions (37°C) to obtain a fermentation liquid rich of volatile fatty acids (VFA) and nutrients, (ii) solid/liquid separation of the fermented sewage (mixed) sludge by a screw press and (iii) Short-Cut Sequencing Batch Reactor (scSBR) performing via-nitrite nitrogen removal and enhanced biological phosphorus uptake (EBPR process) through the dosage of the sewage sludge fermentation supernatant.

Given their sidestream and EBPR nature, the sludges produced were expected to contain a considerable nutrients concentration. Therefore, advanced composting was selected as the best valorisation route of this sludge.

3.1.3 PRIMARY SLUDGE (PRS)

Primary sludge was collected from the WWTP of Almendralejo, Spain, which is managed by Socamex (Figure 3-2 a and b). WWTP is able to treat 18,000 m^3 of municipal wastewater per day (which equals 54,545 population equivalents).

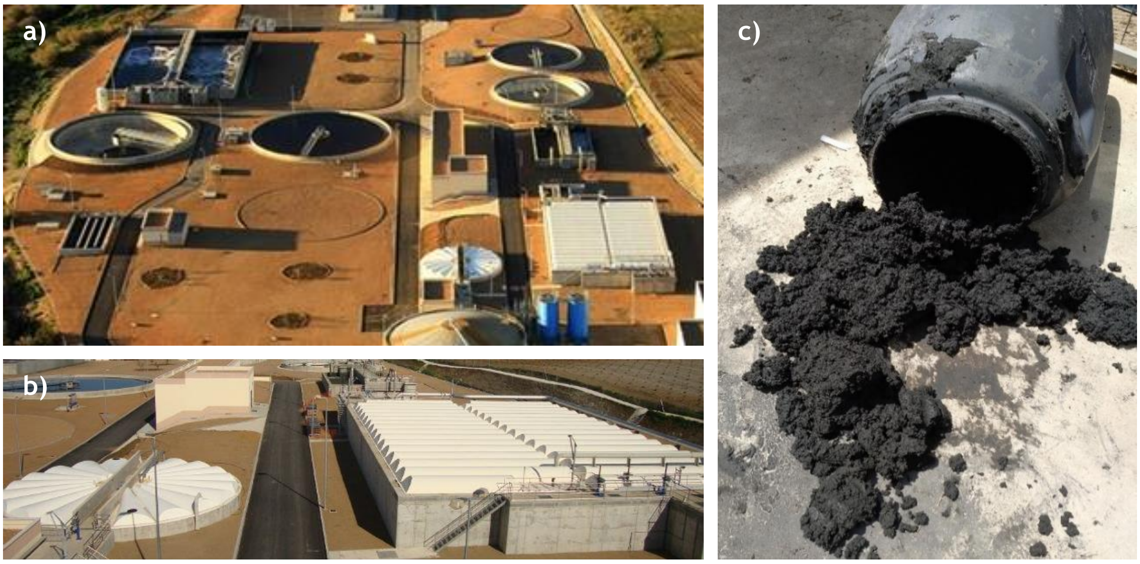


Figure 3-3. Almendralejo WWTP (a and b) and primary sludge equivalent obtained for biodrying.

Briefly, in Almendralejo, wastewater treatment starts with fat, grit and sand removal. Afterwards, wastewater is treated in two successive biological stages with settling stage in each of them. The sludge recovered in the first stage is equivalent to primary sludge which was dewatered using a centrifuge before sending it to BETA for its valorisation (Figure 3-2c). From conventionally produced urban wastewater sludges, primary sludge was the one expected to be analogous to cellulosic sludge, expecting, therefore, a considerable content of cellulosic fibres. Thus, the valorisation route assessed with this sludge was biodrying technology.

3.1.4 SECONDARY SLUDGE (SS)

Secondary sludge was obtained from several small WWTP and conventional sludge composting plants. For instance, the samples were collected from: municipal WWTP of Olot (17,000m^{3}WW d^{-1} and 99,166 P.E treatment capacity) and its subsequent composting plant (10,000 t sludge y^{-1} treatment capacity), composting plant of Jorba, composting plant of Fervosa (overall capacity of 80,000 t y^{-1} mainly for sewage sludge and pig slurry treatment), municipal WWTP of Navàs (8,750 P.E treatment capacity) and municipal WWTP of San Salvador de Guardiola (4,167 P.E treatment capacity). Figure 3-3 shows some examples of the samples collected during that period.

Figure 3-4. Secondary sludge dewatering through belt filter (a and b) and sludge storage in WWTP of Navàs (c). hopper for sludge storage in WWTP of Olot before composting plant (d), composting windrows of composting plant of Aigües de Manresa.

Different origin secondary sludges were used mainly along biodrying and composting start-up periods in order to investigate the effect of raw material characteristics in the biological activity profiles (temperature and oxygen consumption profiles) of aerobic reactors. Additionally and to compare the biodrying performance of different origin, secondary sludge from Jorba sludge composting plant was valorised through biodrying.

3.1.5 MIXED SLUDGE (MS)

Mixed sludge was collected from the WWTP of Almendralejo, Spain, managed by Socamex. As mentioned before, Almendralejo WWTP main treatment process consist of two successive biological stages with settling stage in each of them. Mixed sludge contains mixed sludges from both settlers, and it is normally valorised in plant through anaerobic digestion. The mentioned mixed sludge was dewatered just before entering in the anaerobic reactor in order to be sent to BETA TC to assess its valorisation through biodrying.

3.1.6 PULP AND PAPER MILL SLUDGE (PPS)

Pulp and paper mill sludge was obtained from a cardboard recycling facility in the region of Barcelona, Spain. The WWTP of the facility treats all the rejected pulp and the wastewater produced in the recycled paper production. WWTP consists of a first step of fat and grit removal, primary and secondary treatment and dewatering through belt filer. Dewatered sludge is normally externally disposed for composting. Roughly talking, dewatered sludge consists of 20% biological sludge, 20% floating material and 60% rejected pulp. Figure 3-5 shows an example of sludge gathered in the mentioned pulp and paper mill. Given its origin from pulp & paper industry, it was expected to be highly fibrous, and thus it was expected to be highly energetic. The valorisation route selected for pulp & paper mill sludge was biodrying technology.

Figure 3-5. Samples from Pulp and paper mill sludge.

3.1.7 BULKING AGENTS USED

The sludge was mixed with an appropriate type of bulking agent (e.g. pruning waste, straw, etc.) and in a proper mixing ratio in order to (i) provide a mixture structure to guarantee the proper airflow circulation and (ii) adjust the high initial humidity content of sludges.

In order to select the most appropriate bulking agent to achieve the best start-up conditions, different bulking agents and different sludge to bulking agent mixing ratios were evaluated. Among all bulking agents considered (straw, pine chips, crushed wood…) pruning waste was selected as the most appropriate bulking agent for the process since it presented an appropriate size and shape variability which could allow the efficient aeration of mixture, the correct integration of the bulking agent into the sludge minimizing the compaction and limiting the formation of anaerobic zones. Also, pruning waste was capable of adjusting the moisture content of the mixture and it was easily available for its use in the trials. Finally, pruning waste was highly available and easy to collect in nearby composting plants. An example of pruning wastes used within biodrying and composting trials is shown in Figure 3-6.

Figure 3-6. Pruning waste used as bulking agent in different advanced composting and biodrying trials.

Different sludge to bulking agent mixing ratios were used in biodrying and composting trials. Mixing ratios were selected according to the adjustment of moisture content needed which was calculated by mass balance.

3.1.8 STORAGE OF SAMPLES

Once collected, raw samples were stored at 4ºC for maximum 3 days before setting up the biodrying or composting experimentation. Samples were taken out from the cold storage room hours before setting up composting or biodrying trials, leading in all the cases to an initial bulk temperature between 15 and 20ºC.

3.2 ANALYTICAL METHODS

All solid samples were always characterized for main physic-chemical and biological parameters. Analysis results are always given as an average of triplicate samples with standard deviation.

Physic-chemical analysis were done following, except for specified methods, using the “Test Methods for the Examination of Composting and Compost” (US Department of Agriculture and US Composting Council, 2001).

3.2.1 MOISTURE CONTENT, TOTAL SOLIDS AND VOLATILE SOLIDS

For moisture content (MC) and dry matter (TS) analysis solid samples were dried in an oven at 105ºC for minimum 24h using ceramic crucibles. Weight difference between fresh and dried sample was then used for MC and dry matter calculation (Equation 3.1 and Equation 3.2). Finally, to determine volatile solids (VS), dried samples were burned for 5h at 550ºC in a muffle furnace (ELF 11/14B, Carbolite, Hope Valley S33 6RB, United Kingdom). This temperature allows the total burning of organic matter, thus only the inorganic fraction or ash remains in the crucible. Equation 3.3 details the VS calculation method based on weight difference.

$$MC\ (\%) = \frac{(W_f - W_d)}{(W_f - W_c)} * 100 \qquad \text{Equation 3.1}$$

$$DM\ (\%) = 100 - MC\ (\%) \qquad \text{Equation 3.2}$$

$$VS\ (\%) = \frac{(W_d - W_{ash})}{(W_d - W_c)} * 100 \qquad \text{Equation 3.3}$$

Where: MC (%) is the Moisture content of organic sample, W_f is the weight of fresh sample, W_d is the weight of dried sample, W_c is the weight of empty crucible and DM (%) is the dry matter content of sample. VS (%) is the Volatile Solids content of the sample, W_{ash} is the weight of the ash left after burning.

3.2.2 PH AND ELECTRICAL CONDUCTIVITY

For pH and Electrical Conductivity (EC) analysis, solid samples were diluted to 1 to 5 (w:v) with deionized water. After mixing for 20 minutes and another 30 minutes of settlement, pH and E.C. were measured in supernatant where solubilized salts and ions were present. For pH measurement a pH-metre (Cyberscan Ion 510, XS-Instruments, Carpi MO, Italy) was used, while the E.C. was measured through an electrical conductivity meter (Cond-510, XS-Instruments).

3.2.3 NITROGEN COMPOUNDS: AMMONIACAL NITROGEN AND TOTAL KJELDAHL NITROGEN

Nitrogen compounds were measured by titration procedure after distillation of solid samples (Pro-Nitro M distiller, J.P. Selecta S.A., Barcelona, Spain) in the presence of base in excess (Sodium Hidroxyde at 40%). Additionally, for Kjeldahl Nitrogen (TKN) measurement, a previous acid digestion (with Sulphuric acid at 65% at 400ºC for 2h) step of organic sample was added (Block Digest 20, J.P. Selecta S.A.). Product distillate was collected in boric acid at 3% and titration was carried out using hydrochloric acid at 0.5N. Nitrogen content calculation procedure is detailed in Equation 3.4.

$$N - NH_4^+ or\ TKN\ (g\ kg^{-1}) = \frac{(V_{HCl\,sample} - V_{HCl\,blank}) * N_{HCl} * 14}{W_{sample}} \qquad Equation\ 3.4$$

Where: V_{HCl} are referring to volume of hydrochloric acid consumed in titration in both blanc and sample treatments, N_{HCl} is referring to the normality of the titration acid and W_{sample} is referring to the weight of the sample.

3.2.4 TOTAL PHOSPHORUS CONTENT

Total phosphorus (TP) was determined from the extract obtained by the acid digestion (with one part of sulphuric acid at 65% and 3 parts of nitric acid 3N using a Block Digest 12, J.P. Selecta S.A.) at 100ºC for 2h.

The Watanabe- Olsen colorimetric method based on ammonium molybdate and ascorbic acid was used for the determination of phosphorus. For its quantitative determination, a calibration standard curve between 0.2 and 1.2 mg L^{-1} was built using mono-potassium persulphate (Figure 3-7). After 30 minutes of reaction time the absorbance was measured by a spectrophotometer (DR3900, Hatch) at 890nm.

Regression parameters obtained from the calibration curve (with a minimum acceptable R2 of 0.99) were then used to calculate the concentration in the aliquot and then total phosphorus in the sample was calculated by applying the Equation 3.5.

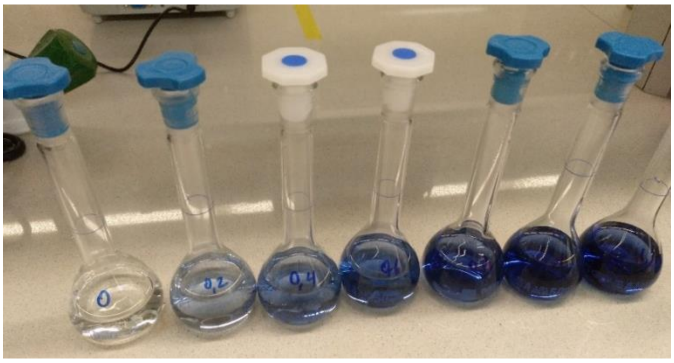


Figure 3-7. Calibration standard for phosphorus determination with Watanabe-Olsen method

$$TP\ (g\ kg^{-1}) = C * \frac{V_f}{V_A} * \frac{V_E}{W_{sample}} * D \qquad Equation\ 3.5$$

Where: C is the concentration measured in spectrometer for the reaction volume prepared (V_F), V_A is the volume of aliquot added for measurement, V_E is the total volume of the extract after acid digestion, W_{sample} is the total sample weight analysed and D is the dilution factor if any extra dilution was required.

3.2.5 TOTAL POTASSIUM CONTENT

Total potassium (TK) was determined from the extract obtained by the acid digestion already mentioned in section 3.2.4.

For the quantification, the Tetraphenylborate Method- based kit provided by HACH® was used (reference 1432399). The working range of the kit was of 0.1 to 7mg/L. After 5 minutes of reaction potassium concentration (in mg/L) was measured through a spectrophotometer (DR-3900, HACH ®) and then corrected with the sample mass extracted.

3.2.6 BULK DENSITY AND FREE AIR SPACE

Bulk density (BD) is the volume occupied by a certain amount of sample. Bulk density of raw organic wastes and mixtures was measured in wet samples following Equation 3.6.

$$BD\ (kg\ L^{-1}) = \frac{W_{sample}}{V_{sample}} \qquad Equation\ 3.6$$

Where: BD is the bulk density, W_{sample} is the weight of sample (in g) and V_{sample} is the volume filled by the sample (in mL).

Free air space (FAS) or air-filled porosity, ε, was estimated from is the water, organic matter and ash content and density in the sample as previously used (Richard et al. 2002). Calculation procedure is detailed in Equation 3.7.

$$\varepsilon_a = FAS = 1 - \ \rho_{tot}\left(\frac{(1-DM)}{\rho_w} + \frac{DM * VS}{\rho_{VS}} + \frac{DM * (1-VS)}{\rho_{ash}}\right) \qquad Equation\ 3.7$$

Where: ε_a is equal to FAS, ρ_{tot}, ρ_w, ρ_{VS}, and ρ_{ash} are densities of mixture (in kg L^{-1}), water (assumed to be 997 kg L^{-1}), organic fraction (assumed to be 1600 kg L^{-1}) and inorganic fraction or ash (assumed to be 2500 kg L^{-1}), respectively. DM is the percentage of dry matter content in the sample (fresh basis) and VS is the volatile solids content (% in dry basis).

3.2.7 TOC AND C/N RATIO

Total organic carbon was estimated from the organic matter by the correction factor suggested by Barrington et al. (2002) and already used by other authors (Villegas & Huiliñir, 2014). The calculation procedure is detailed in Equation 3.8.

$$TOC\ (g\ kg\ TS^{-1}) = \frac{VS}{1.83} * 10 \qquad Equation\ 3.8$$

Where: VS is the percentage of VS content in the sample (in dry basis) and 1.83 is the correction factor suggested by the author mentioned.

Afterwards estimated TOC value was used to calculate C/N ratio (Equation 3.9).

$$\frac{C}{N} ratio = \frac{TOC}{TKN} \qquad Equation\ 3.9$$

Where: TOC is the estimated value of total organic carbon following the Equation 3.8 (in g kg^{-1}TS) and TKN is the total nitrogen measured by the procedure described in section 3.2.4 (in g kg^{-1}TS).

3.2.8 BIOLOGICAL INDICES: RESPIROMETRIC TEST

Assessing biodegradability of organic wastes is enormously helpful for the proper design of aerobic processes (Barrena et al., 2009; Ponsá et al., 2010). To this aim, several chemical and biological processes have been suggested of which aerobic respiration indices seem to be the most suitable, which allow discerning and discriminating those organic samples with high enough biodegradability for their aerobic treatment. Among aerobic respiration indices, the information provided by the combination of dynamic respirometric index (either as an average of 1h or 24h) and cumulative oxygen consumption at 4 days of maximum biological activity (AT_4) seem to be reliable and most complete information (Ponsá et al., 2010). In this context, both indices mentioned are calculated through respirometric test by measuring the amount of oxygen consumed by the material analysed per time unit (i.e. the oxygen consumption speed) (Adani et al. 2006).

Accordingly, all the sludge samples collected in this work were subjected to a dynamic respirometric test. Dynamic respirometer used for this procedure was constructed based on the design described in Ponsá et al., 2010 (Figure 3-8). Between 100-150g of sample were placed in Erlenmeyer flasks with a working volume of 500mL at a controlled temperature of 37ºC. Mass controllers (Mass-Stream, Bronkhorst, Ruurlo, Netherlands) continuously supplied between 10 and 25 mL/min of compressed air (20.9% O2) into biological reactors after wetted. Airflow supplied was selected according to the expected biological activity in the sample. Electronical oxygen sensors (O2-A2 Oxygen Sensor, Alphasense, Exxes, United Kingdom) were used to measure oxygen content of outlet gas after its dehumidification. Arduino acquisition system (Arduino UNO, Arduino, Ivrea, Italy) was used for data acquisition and for monitoring, a software developed in the centre by using Labview 2017 (National Instruments, Austin, Texas) was used.

Figure 3-8. Schematic overview of dynamic respirometer (left) and real picture of equipment (right).

Dynamic Respirometric Index (DRI) represents the average oxygen consumption rate along 24h in the maximum biological activity period. DRI gives an idea of the extent in which organic sample is easily biodegradable. It can give information about the biological reaction burst during aerobic biological processes such as biodrying. DRI is calculated as follows (Equation 3.10):

$$DRI\left(mg\frac{O2}{h\cdot kgVS}\right)=\frac{Q\cdot(O_{2i}-O_{2o})\cdot 31.98\cdot 60}{22.4\cdot W_{VS}} \qquad Equation\ 3.10$$

Where: DRI is the dynamic respirometric index in mg of O_2 per hour and per kg VS analysed, Q is the airflow supplied (L min^{-1}), O_{2i} and O_{2o} are oxygen contents in inlet and outlet air (molar fraction), respectively, 31.98 is the molecular weight of O_2, 60 minutes to hour conversion factor, 22.4 is the value of molar volume of air under normal conditions and W_{VS} is the weight of VS introduced in each biological reactor.

Cumulated oxygen consumption index (AT_4) represents the cumulated oxygen consumption within the 4 days of maximum oxygen consumption rate. 4 days of maximum consumption rate are assumed to start whenever 25% of maximum oxygen consumption rate (calculated as an average of 3h) is reached, assuming the period before this 25% of maximum oxygen consumption rate to be the lag phase of the organic waste itself. This parameter gives an idea of the period in which high biological activity is being occurred. Cumulated oxygen consumption is calculated following the Equation 3.11.

$$AT_n = \int_{t1}^{t1+n} DRI_t \cdot dt \qquad \textit{Equation } 3.11$$

Where: AT_n is the cumulated oxygen consumption in n period (being usually this period of 4 days), t1 is the time in which lag phase is finished, t_{1+n} is the period in which cumulated oxygen consumption is intended to be measured.

Figure 3-9 shows a typical oxygen consumption profile in terms of DRI and cumulated values of a cellulosic sludge.

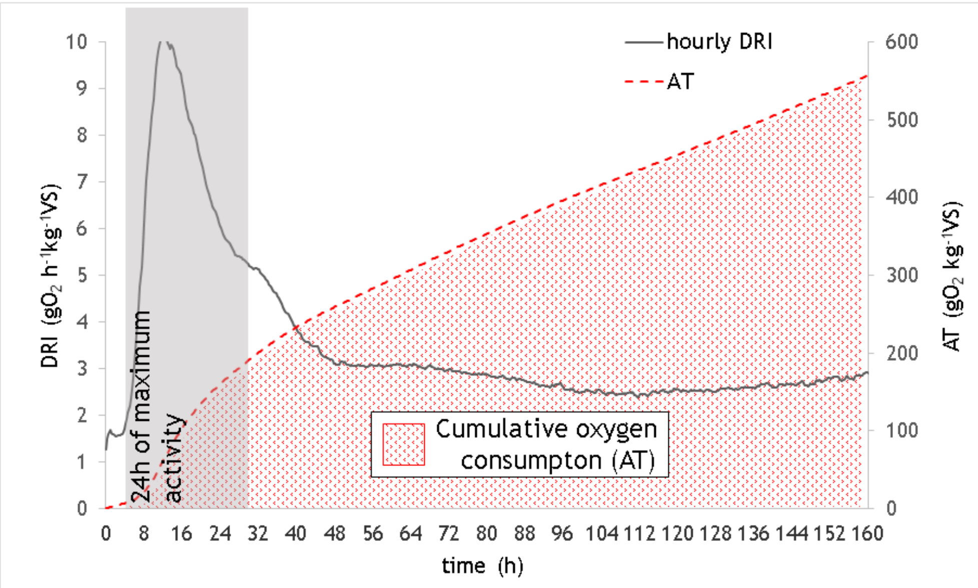


Figure 3-9. Typical oxygen consumption profile (in terms of DRI an AT) of a sludge

3.3 SAMPLING AND ANALYSIS OF GASEOUS EMISSIONS

As mentioned in Chapter 1, emission of pollutant gases is one of the main drawbacks of aerobic processes. The gaseous compounds emitted during composting or biodrying processes might contribute to greenhouse effect (GHG emissions), eutrophication (for instance NH_3) or generation of particulate matter (for instance NH_3 or VOCs). Moreover, emission of odorous compounds (NH_3 and VOCs among others) might be partly responsible of complains from composting or biodrying surrounding neighbourhoods, contributing to a lower social acceptance of the treatment plants.

Therefore, with the aim of evaluating the processes in study from a more holistic point of view, gaseous compounds of interest were monitored in selected trials. Moreover, since gaseous emission data are very scarce in biodrying literature (Zhang et al., 2017, González et al., 2019), all the new information gathered in this regard are expected to be helpful for the research community.

To this aim, gaseous samples were taken daily during biodrying and composting trials. Sampling and analysis methods followed the procedures described elsewhere (González et al., 2019; González et al., 2020). Samples for their subsequent GHG, VOCs characterisation and odour units analysis (group 1) and determination were taken into Nalophan ® bags, while NH_3, tVOCs and H_2S (group 2) were measured in situ (Figure 3-10a). Given the different configuration of biodrying and composting reactors, the sampling methodology differed.

For the composting reactor, which was hermetically closed, gas sampling of group 1 was done directly from the outlet gas pipe to obtain the gaseous samples. All Nalophan® bags were flushed twice before obtaining the final gaseous sample for analysis. For gaseous emissions grouped in group 2, analysis was directly done from the outlet gas pipe into a flux chamber (Figure 3-10a) that allowed the homogenisation and representativity of exhaust gases.

Total volatile organic compounds (tVOCs), NH_3 and H_2S concentration of the outlet gases of reactors were measured daily in situ with a MultiRAE Lite portable analyser (RAE Systems).

Detailed characteristics of the sensors installed in the MultiRAE analyser are given in Table 3-1.

Sometimes, in particular during thermophilic stage, dilution of gas with environment air was required due to too high NH_3 emission rates which were over the detection limits of monitoring equipment.

Figure 3-10. Sampling procedure of gaseous emissions: (a) sampling of NH_3, tVOC and H_2S emissions using a pump to dilute gaseous sample with external air, (b) semi-spherical stainless-steel flux chamber used for biodrying gaseous emissions sampling, (c) suction pump and hermetic box for biodrying gaseous emissions sampling (group 1), (d) gaseous emissions sampling device for biodrying samples (group 2).

Table 3-1. Sensor details of MultiRAE Lite analyser.

Parameter	NH_3	tVOC	H_2S
Sensor type	Electrochemical sensor	10.6 eV PID lamp	Electrochemical sensor
Detection range	100 ppmv	0 - 1000 ppmveg isobutylene	0 - 100
Sensibility	1 ppmv	1 ppmveg isobutylene	1 ppmv

Conversely, for biodrying reactor, auxiliar equipment was used to collect samples: to collect gaseous emissions grouped in the first group, a semi-spherical stainless-steel flux chamber (0.443m of base diameter, 0.154m^2 of base area and 0.045m^3 of volume) (Scentroid, IDES Canada Inc., Whitchurch-

Stouffville, ON, Canada) was used (González et al., 2019) after daily turning (Figure 3-10b). Two theoretically calculated retention times were let to pass before sampling analysis whenever aeration rate was changed. A suction pump and a hermetic sampling chamber, where a Nalophan® bag was placed, were used for sampling (Figure 3-10c). Sampling methodology described above is shown schematically in Figure 3-11a.

For the case of gaseous emissions from group 2, a suction pump was used to extract of exhaust gas from the headspace of biodrying reactor (below straw cushion). 3 devices made by plastic net were used to collect exhaust gases from 3 different sampling points in the headspace of biodrying reactor (Figure 3-10d). This sampling method was assumed to be representative of what would be emitted in a full-scale biodrying plant treating sludge. Analysis was done on site according to the equipment already described above. Sampling methodology and analysis described above is shown schematically in Figure 3-11b.

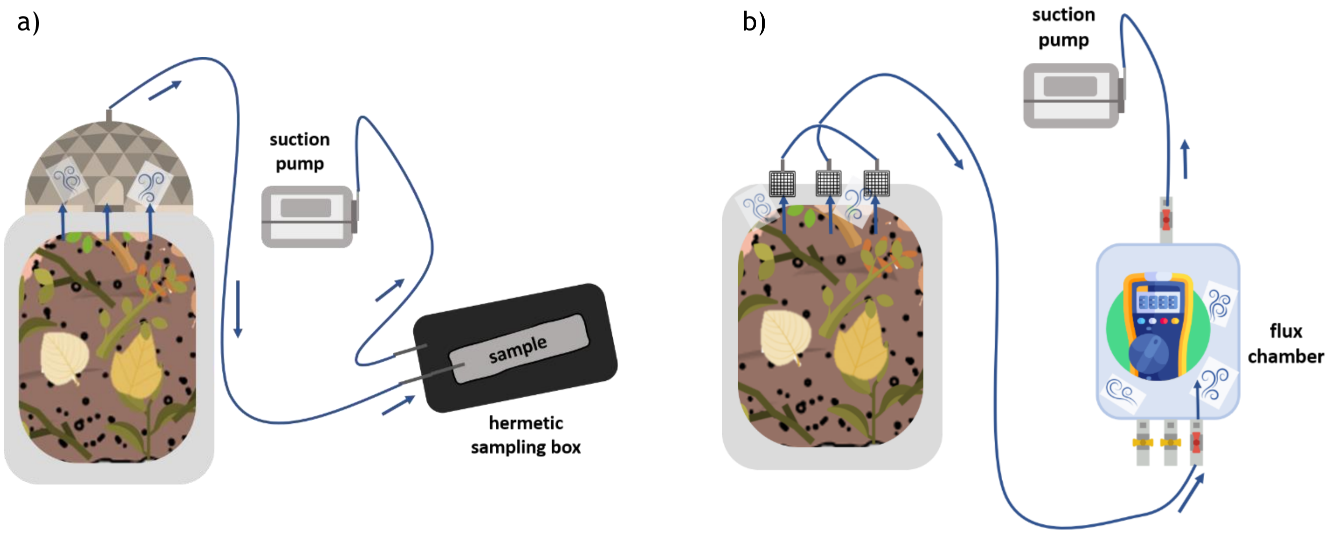


Figure 3-11. Sampling procedure of gaseous emissions during biodrying experimentation: (a) sampling for gaseous emissions of group 1, (b) sampling for gaseous emissions of group 2.

All CH_4 and N_2O analysis were performed in the samples collected in Nalophan® boxes through gas chromatography (GC). The equipment used was an Agilent 6890 N Gas Chromatograph (Agilent Technologies, Inc.) equipped with a HP-PLOT Q semi-capillary column (30m×0.53mm×40.0μm, Agilent Technologies, Inc.) with N_2 as carrier gas (2 psi). Details of the analytical parameters used in chromatography are detailed in Table 3-2. Further details can be found in González et al., 2019a and b.

Table 3-2. Operational parameters for GHG analysis through gas chromatography.

Parameter	CH_4	N_2O
Injection volume (μL)	500	500
Total time of analysis (min)	4	6
Detector	Flame ionisation detector (FID)	Electron capture detector (ECD)
Temperature of injector (°C)	240	120
Temperature of detector (°C)	250	345
Oven temperature (°C)	60	60

For VOCs characterisation, analysis was done in duplicate the very same day of sampling. 1L of sample was pumped through sorbent tubes which were then desorbed to be analysed through gas chromatography- mass spectrometry (GC-MS) using a pre-calibrated Agilent 7820 GC coupled to an Agilent 5975 Mass Spectrometer. Chromatograph was equipped with a D.B.-624 capillary column (60m x0.25mm x1.4 μm, Agilent Technologies, Inc.). Qualitative determination was done according to mass spectra of compounds in the Wiley275 mass spectrum library available in the GC–MS system. Quantitative calibration was done using 35 liquid standard compounds representative of composting processes in increasing concentrations. Further details of the analytical method can be found in González et al., (2019; 2020).

3.4 PERFORMANCE EFFICIENCY ASSESSMENT

To properly differ between the trials assessed when applying any technological solution, indices enabling the evaluation of the successfulness of trials are necessary. Those indices can combine parameters based on the operation of the technology and parameters related to the environmental and economic sustainability of the process, including also parameters related to the quality of the end-products obtained. Although for composting process, the indices to better assess the overall process efficiency point of view exist and more extended (Colón et al., 2012), the indices suggested for biodrying technology are incomplete. To our best knowledge, although indices assessing the technical performance efficiency exist (Huiliñir & Villegas, 2015; Hao et al., 2018), no indices assessing the overall process efficiency from a holistic perspective have been suggested.

3.4.1 BIODRYING PERFORMANCE EVALUATION

3.4.1.1 CONVENTIONALLY USED PERFORMANCE INDICES

Until now, to assess the technical evaluation of a biodrying process, moisture removal, organic matter mineralisation and the relation between the abovementioned have been used. From the process efficiency point of view, drying rates are typically used to evaluate to evaluate daily moisture removal. However, this method is clearly scale dependent and it does not take into account the organic carbon consumed for water removal, making it difficult the comparison of results to other trials or works.

Apart from drying rates, the combination of moisture removal ratio and VS consumption ratio seem to be more accurate to evaluate the efficiency of a biodrying process (Equation 3.11 and 3.12, respectively).

$$Moisture\ removal\ ratio = \frac{m_{MC} * 100}{m_{MCi}} \qquad Equation\ 3.11$$

$$VS\ consumption\ ratio = \frac{m_{VS} * 100}{m_{VSi}} \qquad Equation\ 3.12$$

Where: m_{MC} is the absolute mass of the moisture removed and m_{VS} is the mass of VS consumed in the process and m_{MCi} and m_{VSi} are the initial mass of moisture and VS, respectively, both in absolute terms.

Additionally, (ii) Biodrying Index (BI) is usually reported in literature as performance efficiency index which relates absolute water removal to absolute VS consumption (Equation 3.13) (Huiliñir & Villegas, 2015; Hao et al., 2018). BI indicates the real process efficiency by revealing the organic matter investment for a certain moisture removal ratio (Zhang et al., 2008; Ma et al., 2016).

$$Biodrying\ Index = \frac{m_{MC}}{m_{VS}} \qquad Equation\ 3.13$$

In the present work, moisture removal ratio, VS consumption ratio and Biodrying index were used as a reflect of performance efficiency in conventional way. To this aim, organic matter mineralisation during biodrying was calculated according to the ash conservation principle (Cai et al., 2012). Accordingly, final Volatile Solids (VS) mass was calculated from VS content of representative products after homogenisation and grinding. The VS loss ratio was estimated for every stage (lag, thermophilic and late mesophilic-cooling stages) from the percentage of cumulative O_2 consumption monitored in each stage. Then, those values were used to calculate moisture content removal, correcting it from monitored mass loss. Biodegradation of the bulking agent was assumed to be negligible (Ponsá et al., 2011) as it was confirmed through dynamic respirometry tests.

3.4.1.2 INNOVATIVE INDICES PROPOSED

Besides the abovementioned performance indices, two innovative indices are described and discussed in this work as a reflect of the overall biodrying process. Energetic Biodrying Index (EBI) is a new index integrating an energy consumption parameter (EC) into performance efficiency assessment. Hence, this parameter could give an idea of the energetic and thus, economic and environmental viability of a certain biodrying process performance. EBI is also calculated in absolute terms (Equation 3.14).

$$Energetic\ Biodrying\ Index = \frac{1}{m_{VS}} \cdot \frac{1}{EC/m_{MC}} \qquad Equation\ 3.14$$

where EC is the overall energy consumption of the process and it is given in kWh.

Both BI and EBI indexes are also analysed daily in the present work to achieve the early interpretation and optimization of the biodrying process by modifying control parameters accordingly.

Additionally, energy production per energy consumed (EP/EC) (Equation 3.15) and the Specific production ratio (SPR) as a reflect of mass conservation efficiency (Equation 3.16) are presented as suitable indicators for the evaluation of the process by means of end-product quality. End-quality perspective is then combined with the BI to introduce the innovative and most complete biodrying performance index (BPI) (Equation 3.17). BPI presented here could be used as an overall biodrying process efficiency indicator, facilitating the comparison of results as it considers all the main factors involved in the biodrying performance, including the quality, in terms of energy recovery potential, of the end-products obtained.

$$EP/EC = \frac{EP}{EC} \qquad Equation\ 3.15$$

$$SPR = \frac{m_{TS}\,product}{m_{TS-CS}\,fed} \qquad Equation\ 3.16$$

$$BPI = \frac{1}{m_{VS}} \cdot \frac{1}{{EC}/{m_{MC}}} xEP \qquad Equation\ 3.17$$

Where: EP is the energy potential production in terms of HHV of sieved product considering its specific production ratio (corrected per dry mass of sludge treated). $m_{TSproduct}$ is the total mass of TS in the product after the process (calculated by mass balance) and $m_{TS\text{-}CSfed}$ is the total mass of dry cellulosic sludge fed to the system.

3.4.2 COMPOSTING PERFORMANCE EVALUATION

3.4.2.1 CONVENTIONALLY USED PERFORMANCE INDICES

DRI reduction has been used in numerous works as a reflect of the efficiency of the composting process (Colón et al., 2012; Puyuelo et al.,2014). Besides, combining the DRI reduction with any performance indicator, such as overall energy consumption of the process, permit the normalization of performance data, making it possible the comparison of efficiency results with other works.

To this aim, first, respirometric activity was measured in both initial raw sludge and final sieved end-product (bio-based fertiliser, BBF) and then reduction percentage in both respirometric indices were calculated (Equation 3.18 and 3.19 for DRI and AT_4, respectively).

$$\%\ of\ DRI\ reduction = 100 - \left(\frac{DRI_f * 100}{DRI_i}\right) \qquad Equation\ 3.18$$

$$\%\ of\ AT_4\ reduction = 100 - \left(\frac{AT_{4f} * 100}{AT_{4i}}\right) \qquad Equation\ 3.19$$

where DRI is the dynamic respirometric index calculated for initial (i) and final (f) samples and is given in $gO_2\ kg^{-1}VS\ h^{-1}$ and AT_4 is the AT_4 calculated for initial (i) and final (f) samples and is given in $gO_2\ kg^{-1}VS$.

Second, DRI reduction was used to assess the composting performance efficiency in energetic terms as it was suggested elsewhere (Colón et al., 2012; Puyuelo et al., 2014). The Respiration Index Efficiency (RIE) in terms of energy consumption (RIE-EC) was suggested as a fair indicator of process yield comparing stabilisation degree of organic matter and energetic resource invested for it (Equation 3.20).

$$RIE - EC = \frac{overall\ energy\ consumption}{DRI_i - DRI_f} \qquad Equation\ 3.20$$

where RIE-EC is the respiration index efficiency in terms of energy and it is given in kWh (gO_2 $kg^{-1}VS$ $h^{-1})^{-1}$, the overall energy consumption is given in kWh and DRI is the dynamic respirometric index calculated for initial (i) and final (f) samples and is given in gO_2 $kg^{-1}VS$ h^{-1}.

3.5 END-PRODUCT QUALITY ASSESSMENT

Considering the initial characteristics of sludges, the valorisation routes selected for them differed. Accordingly, the parameters related to the quality analysis of the products obtained were also different. Hence, the product quality evaluation of the biodried products were based on their moisture content and calorific value, at both higher heating value (HHV) level and lower heating value (LHV) level. Conversely, for BBF, stability degree and nutrients content were analysed, together with the measurement of several relevant organic and inorganic pollutants in a process representative sample. Additionally, agronomic quality of composting was assessed trough pot test in a process representative sample.

3.5.1 SAMPLING AND PRE-TREATMENT PROCEDURE OF END PRODUCTS

For end-products sampling, first, all the final bulk- mixture was discharged and thoroughly homogenised. An aliquot of final mixture was taken by the coning and quartering method either for biodried product or composting product for its subsequent analysis (Figure 3-13a).

Sample used for laboratory analysis was sieved through a 10mm mesh for biodrying and 10mm first and 2mm after for composting. Finally, samples taken for calorimetric test were dried in oven at 105ºC and then grinded for their laboratory analysis as schematically shown in Figure 3-13b.

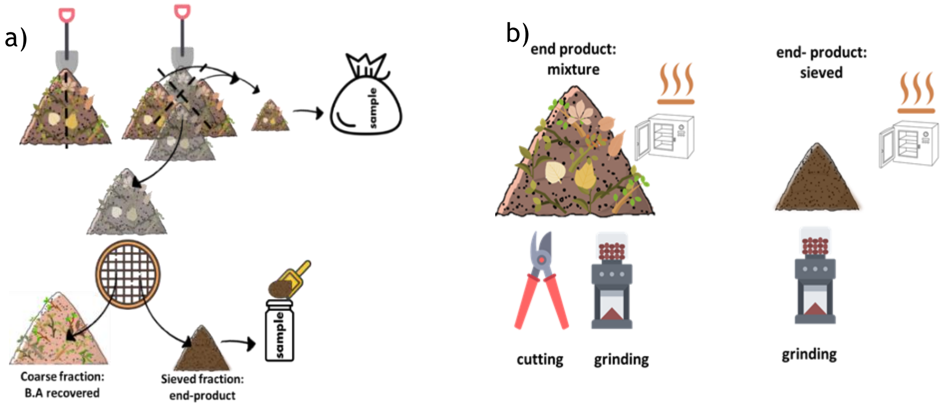


Figure 3-12. a) Sampling by quartering method for mixed products and b) Pre-treatment of biodrying products for calorimetric test.

3.5.2 QUALITY OF BIOMASS FUEL

Biomass fuel (BMF) quality was evaluated taking into account the moisture content of the end-products together with their heating values, both higher and lower heating values. Moisture content was measured accordingly to the procedure described above.

3.5.2.1 HIGHER AND LOWER HEATING VALUES

Higher Heating value (HHV) of wastes was determined separately using a bomb calorimeter (1341 Plain Jacket Calorimeter with the 1108 Oxygen Combustion Vessel, Parr, Illinois, USA) (Figure 3-15a).

Briefly, a pellet between 0,6-1g was placed inside oxygen vessel and then it was filled with oxygen (99.99%) at about 30 atm pressure (Figure 3-15b). The bomb was introduced in the adiabatic calorimeter filled with water and the sample was ignited electrically (Figure 3-16c). The HHV was calculated using corrected temperature rise of each test, previously calibrated using benzoic acid as reference substance.

Lower heating value (LHV) was afterwards calculated from HHV by correcting it (Equation 3.21) taking into account the moisture content of final product and its estimated Hydrogen content (Koppejan & Van Loo, 2012).

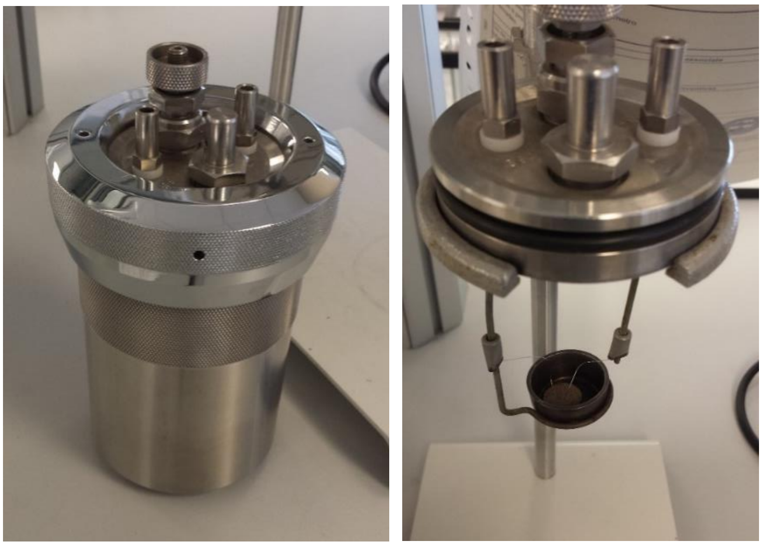


Figure 3-13. (a) Calorimeter used for the determination of the calorific value of biodried products; (b) oxygen bomb used and (c) BMF pellet placed for its ignition in contact with ignition wire.

$$LHV = HHV * \left(1 - \frac{\%\,MC}{100}\right) - \left(2{,}444 * \frac{\%\,MC}{100}\right) - 2.444 * \frac{\%H}{100} * 8.936 * \left(1 - \frac{\%MC}{100}\right) \qquad \text{Equation 3.21}$$

Where HHV is the higher heating value of BMF in MJ kg^{-1}ST, LHV is the lower heating value of BMF in MJ $kg^{-1}{}_{product}$, %MC is the moisture content of biomass fuel (BMF), %H is the estimated hydrogen content of BMF for which 5% was selected as 4-5% content of hydrogen was proved to be content in livestock wastes, while the value was of 6% for conventional biomass fuels (Choi et al., 2014), 2.444 is the enthalpy difference between liquid water at 25ºC and gaseous phase and 8.936 is the molecular weight relationship of both molecules (mH_2O/mH_2).

3.5.3 QUALITY OF BIO-BASED FERTILISER

3.5.3.1 CONTENT OF NUTRIENTS AND POLLUTANTS IN THE BIO- BASED FERTILISER

Nutrients' content and stability by means of DRI and AT_4 were performed as mentioned above. Additionally, to guarantee the safety of the product obtained sanitisation degree and a list of different pollutants were tested and analysed. Sanitisation degree was assessed in terms of thermophilic temperature achievement and maintenance, following the fertilising products regulation 1009/2019 (European Commission,2019).

Concerning pollutants, a representative sample of BBF obtained was sent to external laboratories for the measurement of heavy metals and different emerging pollutants. Cadmium (Cd), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and magnesium (Mg) were analysed by microwave digestion (MARS, CEM) followed by inductively coupled plasma mass (ICP-MS) spectrometry (Agilent Technologies). Mercury (Hg) was analysed by thermal decomposition followed by atomic absorption spectrophotometry (Perkin Elmer). The analysis of heavy metals was performed by the laboratory Servei d'Anàlisi Química in Autonomous University of Barcelona during Jun and September 2019.

108 pesticides and the 16 Polycyclic Aromatic Hidrocarbons (PAH) added to priority list of US EPA were analysed after extraction using the commercial extraction salt packet of QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe). Then, a clean-up step was conducted with primary-secondary amine (PSA) and C18. Finally, the extract was diluted with water (1:2) before GC analysis.

Finally, 15 antibiotics and estrogens included in the European watch list (European Commission, 2018) were also analysed. In this case, the Department of Chemistry in University of Rome performed the analysis described below. In general terms, QuEChERS procedure was applied for the extraction and purification of the analytes. For analysis, a multi-class screening method was developed applying a high-performance liquid chromatography coupled to tandem mass spectrometry (HPLC–MS/MS). Ultimate 3000 chromatograph (Thermo Fisher Scientific) was used as LC system which was equipped with a C18 Kinetex XB column (Phenomenex) to allow the separation of selected compounds. Previously calibrated TSQ Vantage mass spectrometer (Thermo Fisher Scientific) was used for the identification of analytes. Detailed methodology development for the extraction and analysis of antibiotics and estrogens can be found in Benedetti et al., (2020). A complete list of the pollutants determined is shown in Appendix IV.

3.5.3.2 CUANTITATIVE QUALITY INDICES FOR BIO-BASED FERTILISER: FERTILITY INDEX AND SAFETY INDEX

Fertility indicator (FI) and clean indicator (CI) were previously used to define the overall quality of composts (Saha et al., 2010; Puyuelo et al., 2019). Those indexes are calculated by scoring different quality parameters of BBF into several categories.

In this work, a stability degree parameter was included adapting the one suggested by Saha et al., 2010, by a more restrictive scoring strategy based on DRI values of end-products according to criteria adopted by Adani et al., (2004).

With regard to Clean Indicator, apart from the already suggested heavy metal contents, sanitisation degree and PAH parameters were added to Clean Index, and it was re-named as Safety Index (SI). Heavy metal scoring was adapted from Puyuelo et al, (2019) according to the maximum permissible limits established by Spanish Royal Decree for fertilising products 506/2013 (BOE, 2013). Due to its obligatory compliance established by Regulation 1009/2019 on fertilising products, maximum weighing factor was given to sanitisation degree and either compliance or non- compliance was accepted for scoring. Conversely, low weighing factor was given to PAHs due to their lack of limitations in regulations, only considered in some member states such as Austria or Belgium (Saveyn and Eder, 2014). Scoring of PAH16 was done following several regulatory criteria given in Europe (Austria, Denmark, Germany…) (Saveyn and Eder, 2014). Scoring values from 5 to 0 given to the different parameters are shown in Appendix II.

Fertilising Index (FI) and Safety Index (SI) are calculated by summing the multiplication of each scoring value (SV) and weighing factor (WF) for each of the parameters (i) associated to each index (Equations 3.22 and 3.23).

$$FI = \frac{\sum_{i=1}^{n} SV_i WF_i}{\sum_{i=1}^{n} WF_i} \qquad Equation\ 3.22$$

$$SI = \frac{\sum_{i=1}^{n} SV_i WF_i}{\sum_{i=1}^{n} WF_i} \qquad Equation\ 3.23$$

Where, SVi is the scoring value of i parameter and WFi is the weighing factor of i parameter.

3.5.3.3 ASSESSMENT OF AGRONOMIC QUALITY OF BIO-BASED FERTILISER

The agronomic quality assessment was carried out within the framework of the SMART Plant project in collaboration with the agronomy group of University of Verona. The agronomic test was performed using pot tests of the BBF obtained in an advanced composting trial of PS considered representative of the process. The performed the pot tests (4.5L) using pregerminated Maize seedlings (hybrid P0943, Pioneer). Seeds were planted in a silty loam soil with a low available P content and then fertilised with the BBF provided. In order to compare the parameters assessed and mentioned below, negative (not fertilised with P) and positive controls (triple superphosphate, TSP, 46% P_2O_5) were also used. Phosphorus content of the soil-sample or soil-TSP mixture was adjusted by the phosphorus in sample at the recommended dose for maize plants produced in field, 220 kgP_2O_5 ha^{-1} (Perelli, 2009). Nitrogen and potassium contents were further adjusted in all the samples and controls with NH_4NO_3 and KCl following the dosing recommended of the mentioned author.

Five replicates were prepared for all controls and experimental samples and they were grown for 8 weeks under controlled conditions (35ªC and 27ºC of mean day and night temperatures, respectively, relative humidity of 45%, 16/8h light/dark photoperiod, 35%MC maintained in soil). Replicates and controls were organised following the randomised block scheme and they were re-organised every week to avoid any difference in plant growth due to different light conditions. A diluted solution of micronutrients (including ZnO_4, $CuSO_4$, HBO_3, $(NH_4)_6Mo_7O_{24}$, Fe-EDTA and $CaSO_4$) was added every two weeks.

Plant growth was assessed by means of height of the plant, the number of leaves and fresh and dry biomass of shoot and root tissues. Besides, as indicative of nutritional status, particularly N status, of the plant and leaf chlorophyll content, the SPAD (Soil-Plant Analysis Development) was determined using a SPAD-502 (Konica Minolta) and following manufacturer instructions. Macronutrients content, in particular P, micronutrients and heavy metals accumulated in roots and shoots of plants were determined after acid digestion through elemental analysis using an ICP-MS (Agilent 7500e ICP-MS Detection system, Agilent). Nutrient and heavy metal contents in plant tissues were used as indicators of status of plants, efficiency of P use of this source and as well as of possible impacts on plant production and quality.

3.6 TECHNO-ECONOMICAL ANALYSIS

The economic parameters of the implementation of the technologies assessed were estimated for a hypothetical scenario in which biodrying and advanced composting technologies would be implemented in the treatment plants of study: WWTP Geestmerambacht and Carbonera respectively.

For each of the scenarios Capital Expendidures (CAPEX), Operational Expendidures (OPEX), product selling revenues and avoided costs were estimated from mass balance calculations according to experimental data obtained (defined in Appendix III) and construction and equipment budgets facilitated by providers.

Economic model was built under Analytica 5.0 architecture (Lumina Decision Systems). This tool was used to simulate and design the biodrying and composting plants together with their management associated costs, mass balances and revenues calculations.

After a base case study in which each technology would be implemented in the plants of study, a breakeven analysis was carried out to find the minimum capacity of the plant for a zero-profit scenario. Finally, a new scenario for the implementation of the technology for analogous and more conventional primary and EBPR PS for biodrying and advanced composting, respectively was assessed. To this aim, WWTP management data were provided by WWTP of Almendralejo (Badajoz, Spain) and El Prat (Barcelona, Spain), respectively.

3.6.1.1 ECONOMIC PARAMETERS

(i) CAPITAL EXPENDIDURES (CAPEX)

CAPEX was calculated by summing all the construction and equipment costs estimated and included in the economic model. The categories included in this parameter were: (i) windrows construction costs assuming concrete as building material, insulation material, cover of windrows and 2m columns made of stainless steel; (ii) other infrastructure costs for auxiliary material, instrumentation and control, (iii) blower cost, (iv) mixing truck purchase cost, (v) cost of civil works, (vi) cost of engineering design, (vii) biomass boiler purchase for the model developed for biodrying. A detailed description of the calculation method for each budget allocation are given in Appendix III. Real budget data from technological providers were used for the calculation of expenses.

Briefly, CAPEX calculation was done assuming a biological (biodrying or advanced composting) process in windrows built of concrete between 16 and 120m^3. Additionally, a cover made of polycarbonate or similar was assumed together with a cover structure made of stainless steel. The number of windrows to be built was estimated from the total amount of mixture to be treated. Other infrastructure costs included piping, valves, instrumentation and control, and the installation costs of all the instrumentation. Forced aeration was assumed for which a unique blower was considered per each windrow. Their working capacity was estimated by the maximum specific airflow required for the system. The model model was designed to estimate unitary price of blower according to its required capacity, which was calculated according to the maximum specific air consumption required by the mixtures.

Civil works included transportation of building materials and working time of construction worker and it was calculated assuming a minimum cost plus a certain percentage of the partial construction costs depending on the plant size. Engineering costs were estimated in the same way.

(ii) OPERATIONAL EXPENSES (OPEX):

For operational costs calculation, various categories were included in yearly basis: energy consumption cost (both electricity and diesel consumption), bulking agent purchase cost, mixing truck rental for certain treatment capacity plants, personnel costs, maintenance costs, insurance costs and pelleting cost in the model developed for biodrying. Main parameters used for OPEX estimation are summarised in Appendix III.

(iii) Sludge management fee:

Either if the scenario is interpreted as a treatment plant directly installed in the WWTP or it is an external sludge management plant to be constructed, both situations would obtain economic profit from sludge management avoidance in the first case or management fee per sludge treated in the second case. 20€/t of sludge were assumed for sludge management fee, a value facilitated by Aigües de Manresa. However, it should be clarified that this fee is known to be highly variable, depending on the application country and the disposal method of such sludge (management methods which are lower in waste treatment hierarchy present higher management fees).

(iv) Product market price:

For biodried product selling, market price was estimated according to conventional biomass prices reported by Avebiom (2019) according to their energy content: 0.025€/kWh. Thus, a fair price for a metric ton of end products was calculated.

The price of the BBF was calculated based on the nutrient content (NPK) of the BBF that would be substituted as mineral fertiliser. Nutrient content of the product was calculated by mass balance. Therefore, the value would be given by the substitution unit prices and according to the current market price of those fertilisers (nutrient substitution prices provided by Sharara et al., 2018 and detailed in Appendix III).

3.6.1.2 ECONOMIC FEASIBILITY INDICES

To evaluate the economic feasibility of a project some economic indicators can be used, being Net present Value (NPV), Internal Rate of Return (IRR) and Payback period some of the most applied.

(I) Net Present Value

Net present value (NPV) determines the economic feasibility of the project. It represents the balance between the inflow and outflow cash throughout the study period at the present time, considering also a discount rate and the initial investment cost (Zhao et al., 2016; Abdallah et al., 2018). A project is economically feasible if NPV > 0. The higher the NPV value the better economic feasibility would the project have. When NPV is < 0, then the project is not feasible from an economic point of view. The calculation of NPV is done by means of the Equation 3.24.

$$NPV\ (€) = \sum_{t=1}^{T} \frac{B_t - C_t}{(1+r)^t} - K \qquad Equation\ 3.24$$

where: B_t are the benefits or incomes coming from the project implementation, in this specific case product selling and sludge management fees (in time t); C_t are the annual costs of the implementation of the project (OPEX) (in time t, t is each year; T is the lifespan of the project; r is the discount rate and K are the investment costs expressed in € (CAPEX).

For discount rate, a 7% was selected in this case, already used to assess this type of projects (Imeni et al., 2019). A lifespan period of 25 years was assumed for subsequent calculations.

(II) Internal Rate of Return

The Internal rate of return (IRR) is the interest rate at which the NPV equals zero. It is a direct indicator of the profitability of a project, being an IRR higher than the discount rate an indicator of a profitable project (Abdallah et al., 2018; Imeni et al., 2019).

To calculate the IRR of the scenarios assessed the built-in function provided by excel based on annual cash flows was used.

(III) Payback period

The payback period (PP) is the number of years required to recover the initial investment done (Abdallah et al., 2018). PP is calculated through the sum of annual cash flows over time until a positive value is achieved, once this time is reached, net profit period would start.

3.6.1.3 BREAKEVEN ANALYSIS FOR THE DETERMINATION OF MINIMUM WWTP TREATMENT CAPACITY

The breakeven point indicates the zero-profit point. In this case this analysis was used to determine the minimum capacity of a treatment plant to allow neither loss of profit from the project. From that point on the technology could start gaining profit.

The model was built plotting the different estimated NPV for variable WWTP treatment capacities. Then the best fitting regression model was found through Sigmaplot analysis tool and finally SOLVER tool provided the breakeven point.

3.6.1.4 EXPANSION OF THE SYSTEM IN THE FRAMEWORK OF IMPLEMENTED SMART SOLUTIONS

Since biodrying and advanced composting technology can't be considered isolatedly, the expansion of the system boundaries was contemplated in the economic model to include the whole WWTP where Cellvation® system and SCENA process would be implemented, respectively. This scenario with a bottom-up approach of coupled technologies was assessed in Deliverable 4.5 of SMART-Plant entitled "Socio-economic assessment, incl. LCC and CBA reports" (SMART Plant G.A 690323, 2020). Three different sub-scenarios were assessed in each case by modelling three WWTPs with rising treatment capacities, namely, small, medium and big. Insights about the economic viability of the combined SMARTechs were given by comparing its resulting economic parameters upon the base case LCC of WWTP. For the calculation of LCC of the base case WWTP Equation 3.25 was used.

Here, it was considered that the divestment costs amount to 10% of the capital expenditure. Operational costs sum up to contribute to the plant's life cycle costing during the entire lifetime (LFT) (25 years).

$$LCC_{plant} = CAPEX_{plant} * 110\% + OPEX_{plant} * LFT \qquad (Equation\ 3.25)$$

To calculate the LCC for combined scenario, $CAPEX_{SMART}$ and $OPEX_{SMART}$ were assumed to be the additional capital and operational costs considered, while subtracted revenue streams from recovered resources ($REVENUE_{SMART}$) were also considered in the LCC_{SMART} calculation (Equation 3.26).

$$LCC_{SMART} = LCC_{plant} + CAPEX_{SMART} * 110\% + (OPEX_{PLANT} - REVENUES_{SMART}) * LFT \ (Equation\ 3.26)$$

Finally, the savings in life cycle costs when implementing SMART Plant solutions, were then calculated applying Equation 3.27. If $LCC_{Savings}$ result to be positive, the SMART Plant solutions implemented would have a positive impact in the lifetime of the WWTP.

$$LCC_{Savings} = 1 - \frac{LCC_{SMART}}{LCC_{Plant}} \qquad (Equation\ 3.27)$$

4. DESIGN, CONSTRUCTION AND START-UP OF PILOT PLANT

4. DESIGN, CONSTRUCTION AND START-UP OF PILOT PLANT

One of the main challenges faced in the framework of this thesis was the design, construction and development of the pilot plant to feed with the requirements of the technologies in study. To this aim 2 twin reactors were designed and constructed, one per each technology to be implemented. All the specific monitoring and operating devices required were installed for each of the individual sub-pilots (biodrying and advanced composting). Likewise, specific advanced control systems for aeration were developed and installed for each technology, based on the monitored parameter used to control the aeration, namely bulk temperature for biodrying and oxygen uptake rate for advanced composting.

4.1 PILOT PLANT: COMPONENTS DESCRIPTION AND WORKING PROCEDURE

Schematic 3D view of the mobile bench scale plant designed is shown in Figure 4-1. The construction and optimization of its operation was carried out at BETA Technological Centre facilities, while an operation period of around 9 months was undertaken at Manresa's WWTP. Pilot plant consists mainly of the following components, identified with numbers in the Figure: reactor (1), scale (2), air compressor (3), monitoring & controlling devices (4), energy monitoring system (5), metallic structure (6) and mixer (7). The monitoring and controlling equipment of each consists of an airflow meter, oxygen and carbon dioxide sensor, and 2 temperature probes, one for inlet airflow and the other for bulk mixture. Scale (Gram Precision/k3-k3i, Gram group, l'Hospitalet de Llobregat, Spain) placed under the biodrying reactor allows the monitoring of weight loss during biological process, which is afterwards used to calculated corrected moisture loss and process efficiency.

Main components of bench scale plant are briefly explained technical and functionally in Table 4-1 and further detailed in sections between sections 4.1.1 and 4.1.3.

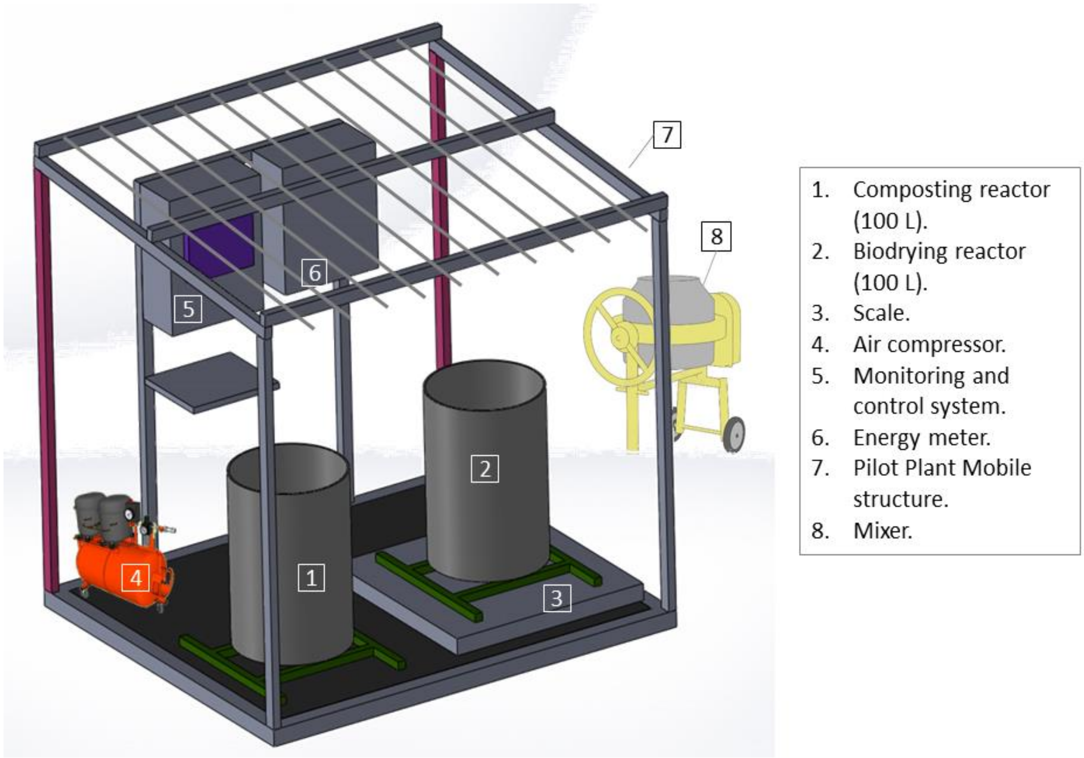


Figure 4-1. Schematic overview of bench scale biodrying and composting plant.

However, troubleshooting section (section 4.4) describes the evolution of plant from first construction and while process optimization was being occurred.

Table 4-1. Description of the main components of the Downstream SMARTech B.

Component	Technical Description	Picture of component
01-02 biological reactors (for biodrying and advanced composting)	Cylindrical biological reactors with a total volume of 120L. They have a diffusion grid in the bottom to allow efficient aeration of mixtures in study. Reactors were thermally isolated to avoid to the maximum extent temperature loss and maintain adiabatic conditions. More details are given in section 4.1.1.	

Table 4-1 cont.

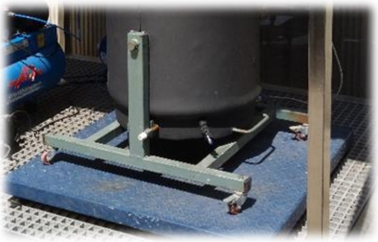
03-Scale (only for biodrying system)	The scale (max. 600 kg ± 0.1) placed under the biodrying reactor allows determining the mass reduction of the process. This mass reduction is an indirect measure of the drying rate. This component was installed only for biodrying reactor.	
03-Air compressor	Air compressor provides the air required to maintain aerobic conditions in the processes.	
04-Monitoring and control system	Biological process is monitored and controlled continuously. Specific and advanced airflow regulation was developed and implemented to optimise process efficiency. Pilot plant monitoring system based on Arduino UNO and Labview 2017® can be visualized and controlled remotely by an on-line application. More details are given in section 4.1.2.	 
05-Energy monitoring system	All electric and electronic devices are connected to the energy meter system to assess the specific and global energy consumption of the pilot plant. In this way, for process optimization steps, energy consumption is also taken into account.	

Table 4-1 cont.

06-Pilot plant structure	A mobile structure was designed and built specifically for the plant. The structure was built with stainless steel (AISI 316L) to support the external conditions (low and high temperatures depending on the season and a high presence of corrosive gasses like NH_3 generated during the aerobic biodegradation of organic matter).	
07-Mixer system	An industrial mixer is included in the plant in order to facilitate the initial mixing of the sewage sludge with the bulking material (normally wood chips or pruning wastes).	

4.1.1 BIOLOGICAL REACTORS

Biodrying and composting reactors, designed as twin reactors, except for the cover, are shown in detail in Figure 4-2a and b, respectively. They have a cylindrical geometry, with a total height of 770mm and a diameter of 550mm. Reactors were built with stainless-steel (AISI 316L) and lateral walls were isolated with stone wool between both stainless-steel layers to maintain adiabatic conditions. Additionally, due to an excessive surface to reactor volume leading to an excessive heat loss through reactor walls, an extra elastomeric insulation foam layer was added in the outside surface of reactors to further increase thermal insulation.

In order to facilitate the operations to charge and discharge wastes and products, as well as the cleaning of the internal structure of the bioreactors, a horizontal rotatory axis (points 01 in Figures 4-2a and b) was included.

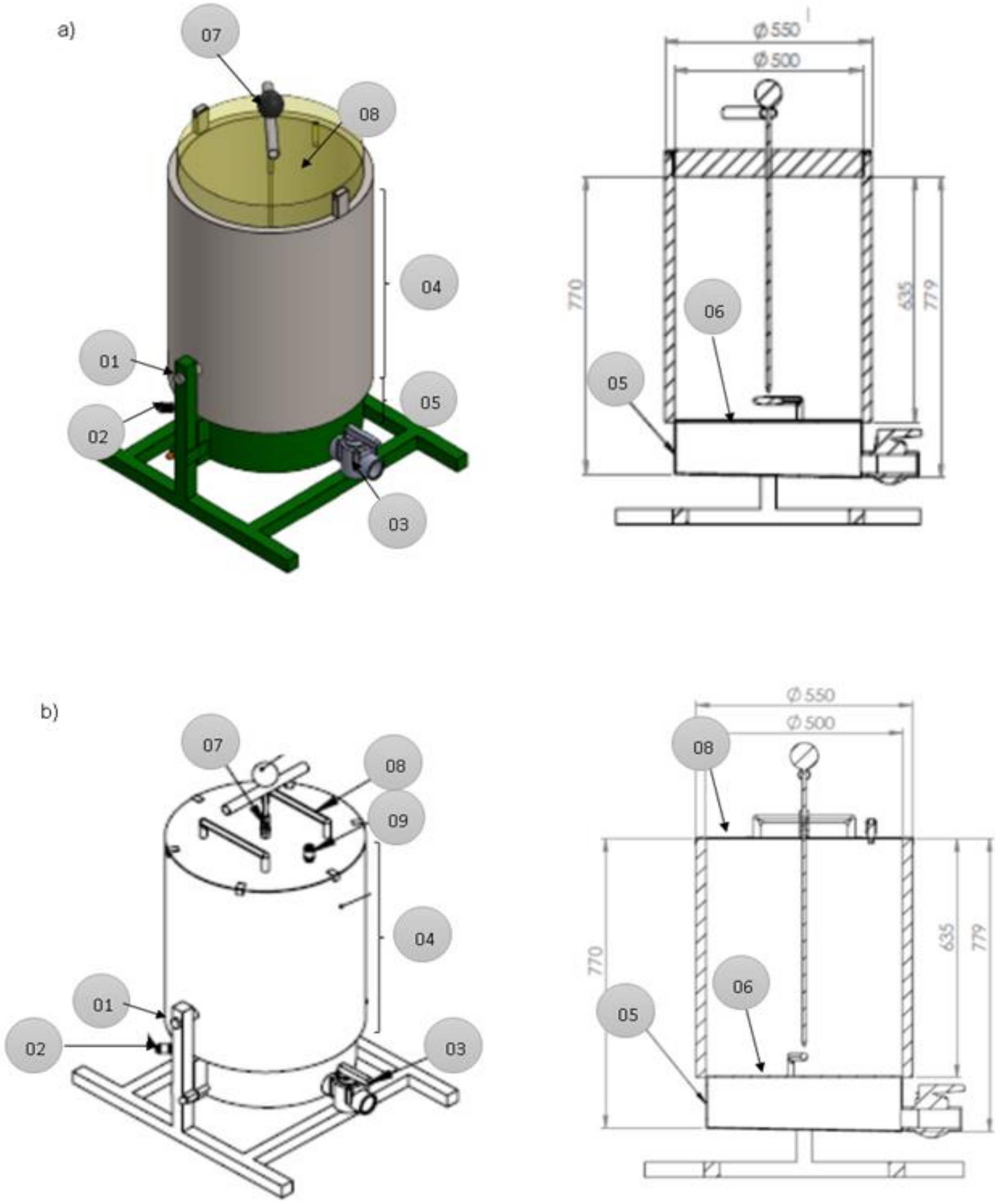


Figure 4-2. Schematic (a) advanced composting, and (b) biodrying

At the bottom of the lateral wall, two orifices are situated: a hole of 6 mm diameter (Figures 4-2a and b, points 02, and Figure 4-3b) for inlet airflow and in the other side, a hole of 24 mm diameter for leachate removal (Figures 4-2a and b, points 03 and Figure 4-3c). In the bottom plate of the reactors, a 45-degree slope was added to facilitate the leachates removal. Reactors have an operative volume of 100 L (Figures 4-2a and b, points 04), with extra 20L of air chamber in the bottom of reactor

(Figures 4-2a and b, points 05). A perforated plate is fitted into the reactor (point 06 in Figures 4-2a and b, and Figure 4-3a) to separate both chambers.

Regarding data monitoring system, a multipoint pt-100 temperature probe (Figure 4-2, point 07 and Figure 4-3d) allows temperature monitoring in 3 mixture points, in the bottom, in the middle and on the top of the mixture. Inlet air temperature is also monitored through a pt-100 probe. Oxygen and CO_2 monitoring systems were installed to analyse outlet gases for both reactors. Further description of monitoring devices and both control systems are given in section 4.1.2 and 4.3, respectively.

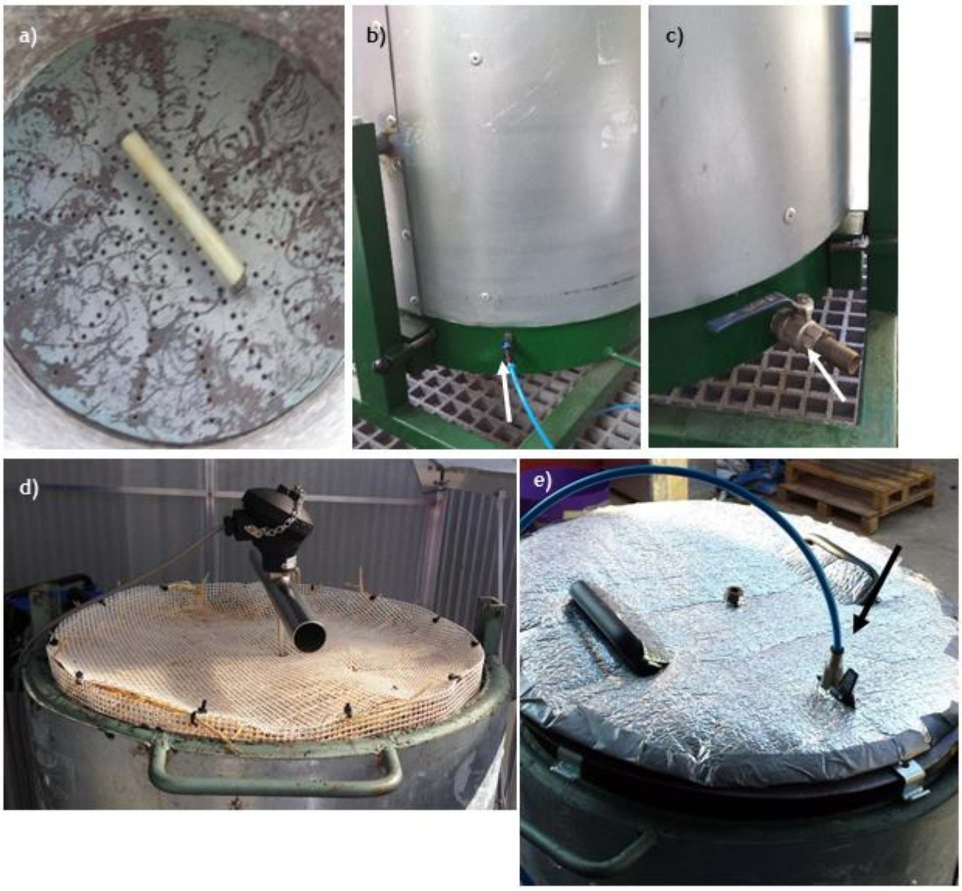


Figure 4-3. Some of the mentioned parts of bioreactors: (a) perforated plate for air diffusion, (b) inlet air supply hole identified by white arrow, (c) leachate recovery valve marked with white arrow, (d) top cover of biodrying reactor made by straw and (e) exhaust gas collection hole in hermetic cover of composting reactor, hole marked with black arrow.

The difference between the two systems, apart from the scale installed for the biodrying reactor, is the cover of the reactor. A permeable cover is placed at the top of biodrying reactor (Figure 4-2a, point 08), it consists of a net cushion of 10 cm width filled with straw. This cover hampers excessive heat loss while allowing the evaporation of the water. Gases generated during biodrying process were suctioned through an external fan in a controlled gas flow (0.4L/min) to monitor its oxygen and CO_2 content. For composting reactor, a hermetic cover (Figure 4-2b, point 08) avoided heat loss and allowed the collection of exhaust gases in headspace for O_2 and CO_2 monitoring (Figure 4-2b, point 08 and Figure 4-3d) to be subsequently used by the control system based on biological oxygen uptake.

4.1.2 MONITORING AND CONTROL SYSTEM: COMPONENTS

Air was supplied from an air compressor (Dixair DNX 2050, WORTHINGTON CREYSSENSAC, Pinto, Spain), through a mass airflow meter/controller (Mass-Stream D-6341-DR and D-6361-DR, Bronkhorst, Netherlands for composting and biodrying systems, respectively) (Figure 4-4, point 01 & Figure 4-5a).

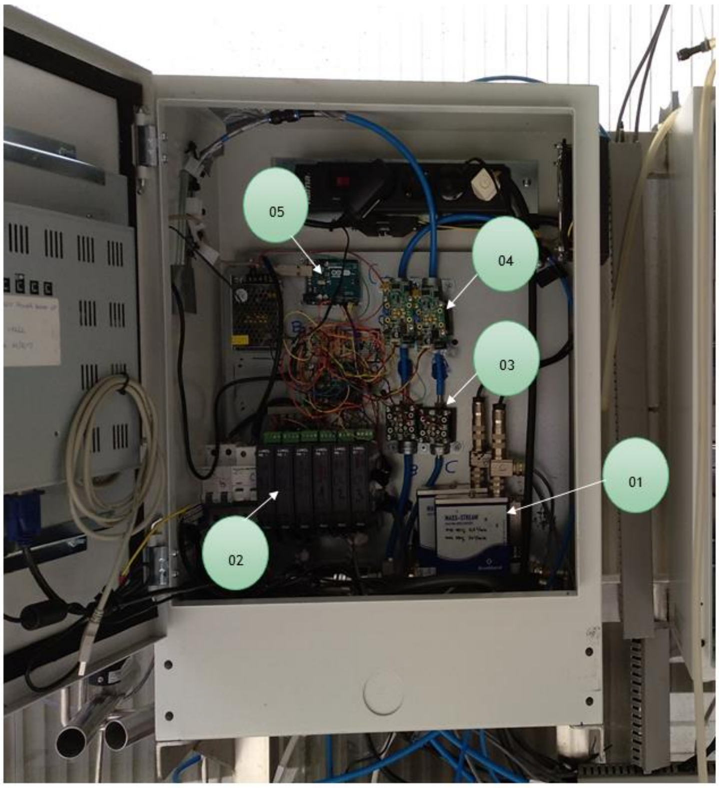


Figure 4-4. Signal acquisition and transformation of monitored and control parameters.

Optimum working conditions of airflow controller were guaranteed by controlling air pressure to 2.5 bar using a manometer and dried through a series of condensers and filters (Figure 4-5b).

Inlet air temperature was monitored continuously through a 50mm long and 4mm diameter temperature probe (Iserntech S.A., Vilanova i la Geltrú, Spain) (Figure 4-5c). Mixture temperature was monitored in 3 points (bottom, middle and top of reactor) using a multipoint pt-100 temperature prove of 700mm long and 8mm diameter (Iserntech S.A., Vilanova i la Geltrú, Spain) (Figure 4-5d). Exhaust gases composition monitoring was carried out using oxygen and carbon dioxide sensors (models O_2A_2 and IRC A_1, respectively, Alphasense, Essex, United Kingdom) (Figure 4-4, points 03 and 04, respectively and Figure 4-5e, A and B, respectively).

Figure 4-5. Monitoring and control devices of biodrying and composting systems developed.

All electrical signals generated in monitoring equipments were read through data acquisition system (Arduino UNO) (Figure 4-4, point 05) and then transferred to the PC in the plant though the software developed for monitoring and control (Labview 2017®, National Instruments). All data are monitored, registered and saved in an excel file. Further details of control system are given in section 4.3.

4.1.3 ENERGY MONITORING SYSTEM

All electric and electronic devices were connected to energy meters in order to be able to monitor and register continuously their energy consumption. Energy consumption monitoring system was developed by Wellness SMART Cities, also partners in SMART Plant project.

Each of the technologies were able to monitor energy consumption of (i) monitoring devices (Figure 4-6, point 02), and (ii) airflow meter (Figure 4-6, point 03). General consumption of the plant including energy consumption of air compressor was also monitored, although in this case, consumption of each technology was not able to be monitored independently (Figure 4-6, point 01).

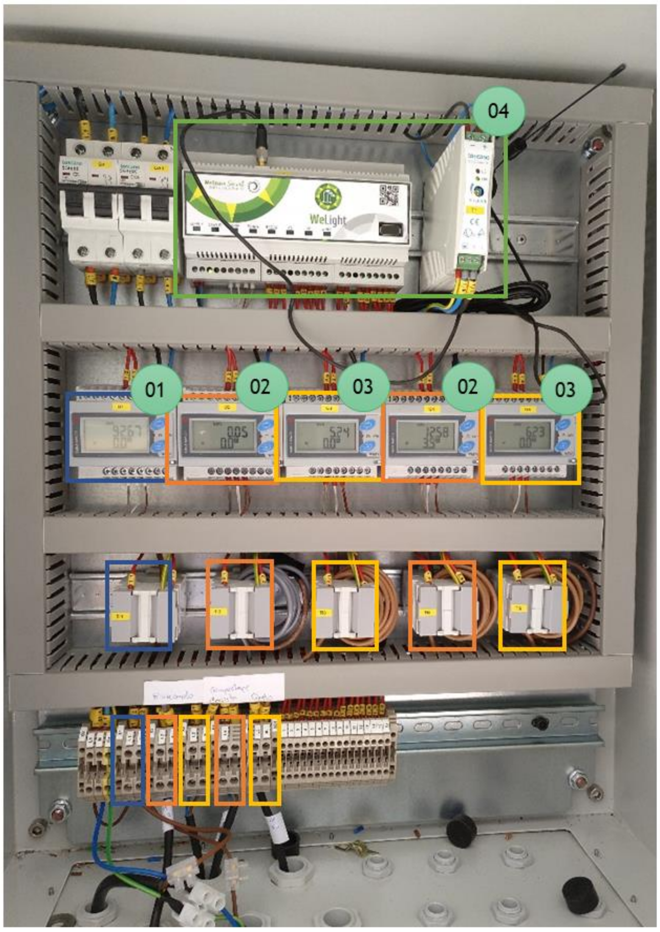


Figure 4-6. Energy consumption monitoring system.

The subtraction of energy consumption coming from electronic devices (sensors and airflow meter) from the general consumption values gives the energy consumption of air compressor, which is responsible of the main energy consumption. All data are registered and sent to the cloud. Registered data are available in WeSave system developed by Wellness SMART Cities.

4.2 FLOW MODEL DETERMINATION IN REACTORS

4.2.1 THE DISPERSION MODEL

Biological processes usually require hydrodynamic studies of their reactors to understand their heat and mass transfers as well as for making some operating decisions. In the case of biodrying and composting, biological processes in which oxygen availability is key for its proper performance, the better understanding of both heat and mass transfer are critical. According to other authors, describing a composting process based on an ideal flow mathematical model would not be accurate (Tremier et al., 2005) as the observed behaviour is not often ideal. Therefore, the axial dispersion model is often used to describe the non-ideal plug-flow behaviour in composting reactors. The axial dispersion model studies the longitudinal dispersion of a tracer introduced in a fluid (Figure 4-7) assuming there occurs a degree of back-mixing.

For the experimental characterization of non-ideal reactor, residence time distribution (RTD) method is typically used. In general terms, for RTD study an impulse is introduced punctually or continuously (step injection) in the reactor, typically called a tracer element which is afterwards monitored by a detection system. RTD permits the determination of a dimensionless parameter D/uL as a measure of the dispersion level of the tracer (Gòdia & López, 1998). Negligible dispersion with a value close to 0 occurs in ideal plug flow reactors while the dispersion is large, and the value of the parameter tends toward infinite in perfectly mixed reactors. After the tracer behaviour is monitored in the exit of the reactor, mean residence time (t_m) and variance (σ^2) can be calculated experimentally. t_m indicates the mean time which a fluid introduced will remain in the reactor while σ^2 indicates the degree of the spread of the tracer inside the reactor. Both values can then be used to calculate D/uL and define the hydraulic behaviour of the reactor. The calculation of all the parameters described above are given in section 4.2.2.

4.2.2 EXPERIMENTAL PROCEDURE

A plug flow-like pattern was expected for biodrying and composting reactors. Having this hypothesis as starting point, the flow model of the reactor was determined by the residence time distribution (RTD) and specifically step injection method by following the oxygen content as tracer gas. Oxygen is easily detectable, even at low levels, and it was possible to monitor it continuously though the actual monitoring system, moreover tracer in this particular case would be exactly the same element used for plant operation. Air supply, with a known oxygen content, at a particular airflow rate was monitored and controlled by the airflow meter installed in the pilot plant. As oxygen is easily consumed, stabilized compost was used as solid bed in the determination. In this way oxygen consumption was assumed to be negligible.

Stable compost was obtained in MSW composting plant of Parc Ambiental de Bufalvent, Manresa. Its stability was confirmed through dynamic respirometric test (DRI 0.1 ± 0.1 gO_2 $kg^{-1}VS$ h^{-1} and AT_4 29 ± 6 gO_2 $kg^{-1}VS$) and porosity was considered adequate for the procedure (59% estimated FAS value). 92% of the total reactor volume was filled with the selected organic bed, letting a headspace of 5 cm height. Reactor, then, was hermetically closed letting open only air supply and gas effluent valves.

As a first step, a constant pure nitrogen known flow was introduced in reactor until negligible oxygen concentration was detected in the outgoing gas. At that point, nitrogen flow was quickly replaced by clean air (20.9% of oxygen) at a known flow. Evolution of oxygen amount in the outlet gas was followed until its value was equal to the inlet gas. Data were monitored every 10 seconds. RTD study was repeated for 7 different airflow-rates, selected to be within the typical the range of the flows that would be used in biodrying and composting experimentation: 0.8 L/min, 1 L/min, 2 L/min, 4 L/min, 6 L/min, 8 L/min and 10 L/min.

For each RTD curve, the mean residence time (tm) and the variance (σ^2) were calculated as shown in Equations 4.1 and 4.2. As key parameters calculated, t_m indicates the average residence time of the compound monitored in the reactor (expressed in time units). σ^2 is the variance, an indicator of the dispersion of compounds in the reactor (expressed in square time units).

$$t_m = \frac{\int_0^{\infty} C\,\mathrm{t}\,\mathrm{dt}}{\int_0^{\infty} C\,\mathrm{dt}} = \int_0^{\infty} C\,t\,dt = \frac{\sum t_i\,C_i \Delta t_i}{\sum C_i \Delta t_i} \qquad \text{Equation 4.1}$$

$$\sigma^2 = \frac{\int_0^{\infty} (t - t_m)^2 C dt}{\int_0^{\infty} C\,dt} = \frac{\int_0^{\infty} t^2\,C\,dt}{\int_0^{\infty} C\,dt} - t_m^2 = \frac{\sum t_i^2 C_i \Delta t_i}{\sum C_i t_i} - t_m^2 \qquad \text{Equation 4.2}$$

Where: Ct is the oxygen content in outlet gas along time t.

Afterwards, the mentioned key parameters were used to calculate the axial dispersion coefficient (D/uL) according to the Van der Laan theory for the closed vessels (Levenspiel, 1999), (Equation 4.3).

$$\sigma_\theta^2 = \frac{\sigma^2}{t_m} = 2\left(\frac{D}{uL}\right) - 2\left(\frac{D}{uL}\right)^2 \left[1 - e^{-uL/D}\right] \qquad Equation\ 4.3$$

4.2.3 RTD PROFILES AND DETERMINATION OF REACTOR FLUX PARAMETERS FOR DIFFERENT AIRFLOWS

RTD or E(t) curves obtained from numerical derivation of monitored signals with all the airflow-rates assessed are shown in Figure 4-7. Highest airflow-rates (8 and 10 L/min) showed an almost immediate and narrow peak within the first minutes of analysing period. Lowest airflow-rates, though, showed a delay in the peak, which was not as sharp as the ones obtained with highest airflow rates.

Experimental parameters calculated for all the airflow rates assessed are shown in Table 4-2. Those parameters are required for the definition of the flow model: mean retention time, variance of the signal and dimensionless axial dispersion parameter.

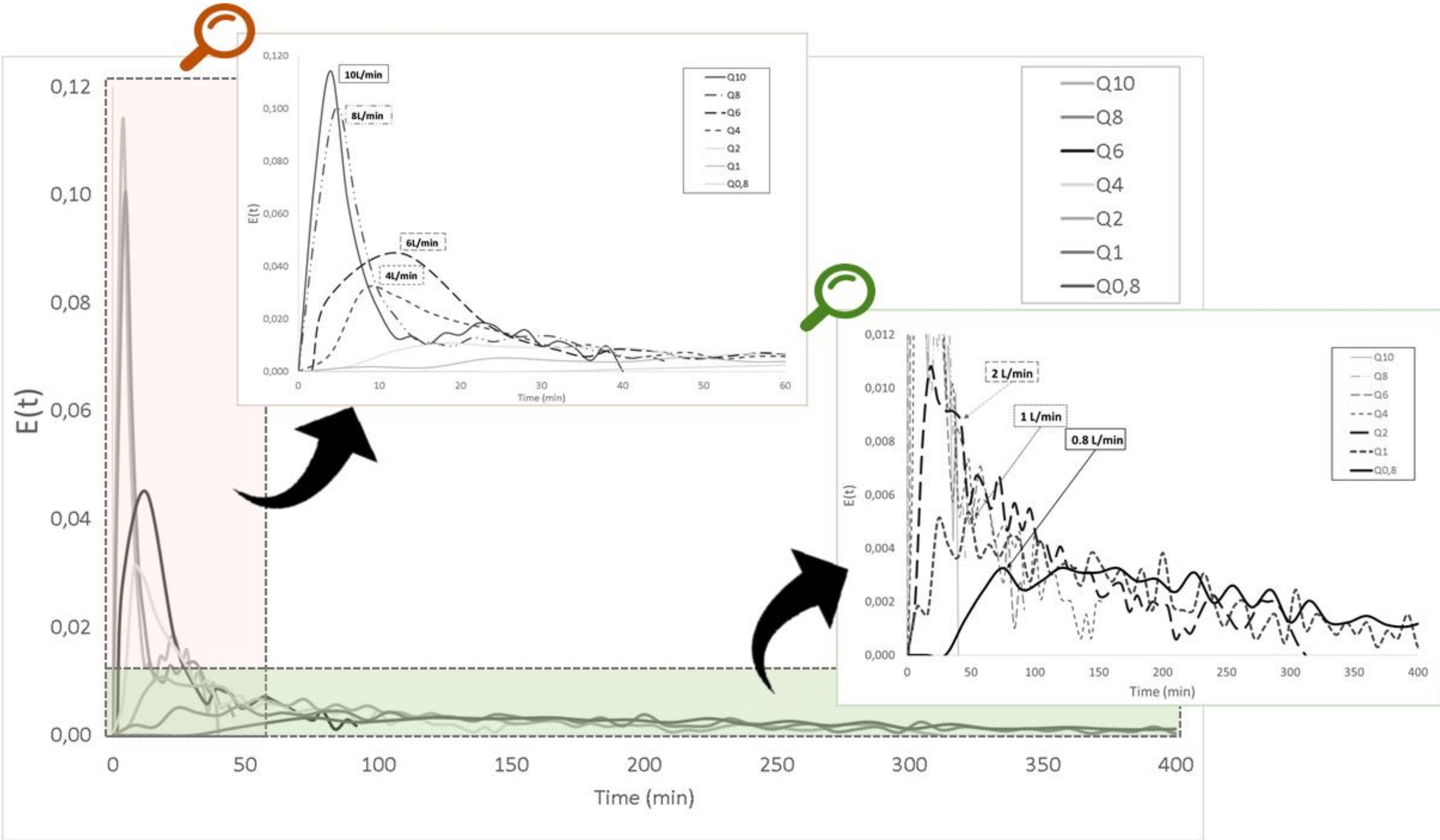


Figure 4-7. E(t) curves for the airflow rates studied in the RTD study. Curves for airflow rates from 4 to 10 L/min are shown in figure 4.15(a) while curves for airflow rates from 0.8 to 2 L/min are shown in figure 4.15(b).

As expected, highest airflows obtained a shorter mean retention time, meaning that the lower the airflow-rate the longer the period required for an element to pass through the solid bed and to be detected in the effluent. Conversely, highest variance values were achieved with the lowest airflows obtained, attributing this response to the higher persistence of the air in the organic bed and the likeable presence of dead zones. Finally, D/uL parameter showed an increasing tend concomitantly to airflow-rate. According to literature consulted, the higher the D/uL value, the more pronounced seems to be the asymmetry of the E(t) curve, being displaced typically with an early peak with a long tail. Minimum value was found for the 0.8L/min airflow-rate achieving a D/uL value of 0.1. For instance, D/uL values above 0.01 are considered as large deviation from plug-flow model as the signal is spread and a nonsymmetric curve is obtained (Levevenspiel 1999).

Table 4-2. Experimental data (t_m, σ^2 and D/uL) calculated from RTD studies for the airflow-rates assessed.

Flow rate (L/min)	tm (min)	σ2 (min^2)	D/uL
0.8	256.6	12829.7	0.11
1	164.2	14227	0.43
2	84.1	4162.5	0.54
4	38.0	1147.5	1.33
8	12.5	135.0	2.27
10	10.8	105.03	3.57

Thus, as a partial conclusion, the evaluation of D/uL parameter, along with the rest of the parameters, confirms that reactor works mainly following a plug-flow model, showing however a non-ideal behaviour with high variance values.

To finalise with flow model study section, theoretical retention time (τ) was compared to the experimentally obtained mean retention time (t_m). Comparative results are visually clear in Figure 4-8.

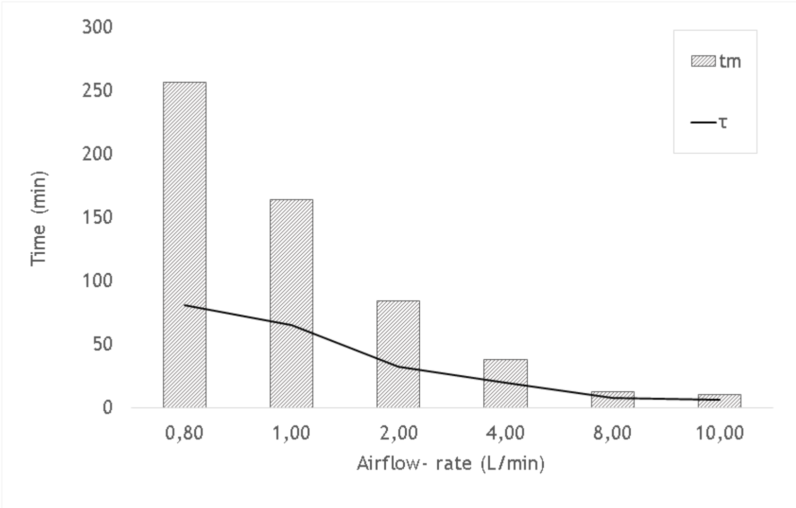


Figure 4-8. Comparison between mean retention time obtained experimentally by RTD study and the one calculated theoretically of the reactor in study.

The difference between both parameters is relatively low when working with high airflows. However, for airflow levels below 4L/min the difference between theoretical and experimental values is increasingly high. Thus, in the case of the lowest airflow-rate (0.8L/min), it would take 3 times longer than theoretically expected to pass through the reactor. This parameter was important for the establishment of a monitoring period when control system was designed (explained in section 4.3).

4.3 CONTROL SYSTEM DEVELOPMENT

As mentioned before (section 1.3.2), several parameters should be monitored and controlled during aerobic biological processes: temperature, moisture content, aeration rate, C/N ratio and raw material and mixture structure. Aeration rate was the main parameter manipulated in both technological solutions assessed in this thesis, although initial parameters of the bulk mixture were also adjusted. Additionally, for the evaluation of the processes, inlet air temperature, bulk mixture temperature, initial mixture characteristics and airflow rate supplied were monitored. However, aeration strategy was adapted depending on the objective of the process itself. As mentioned before, the aim of composting is the maximum mineralisation of organic matter in a waste in order to stabilise organic wastes before applying them into the soil. Conversely, the aim of biodrying is the maximum moisture removal of an organic waste by biogenic heat with the minimum organic matter biodegradation in the process. Control systems were developed according to each differential objective described for each process.

During composting, biological degradation was intended to be maximised in order to achieve the stabilisation of organic waste in the shortest possible period, thus shortening also process completion period and reducing land requirements. Mineralisation can be maximised by optimising biological activity in the bulk mixture, which is reflected by the respiration activity of bacterial community.

Biodrying literature treating different organic mixtures assessed different air-flow strategies and rates. Most of them compared various continuous aeration levels (Zhao et al., 2010; Sadaka et al., 2012, Sharara & Ahn 2012, Huiliñir & Villegas, 2014), without adapting aeration level to the process phase. However, given that bacterial community succession and concomitant temperature variation occurs along biodrying (Cai et al., 2016a), concurrent adaptation of the aeration level could lead to maximise the moisture removal. Cai et al., (2013) adapted aeration levels to each phase of biodrying process depending on the pile temperature and O_2 consumption.

In order to understand the biodrying process, it was necessary to understand the effect of the aeration in the bulk temperature and therefore in the drying performance. The objective of start-up period was the improvement of performance efficiency by adjusting the aeration strategy.

First trials, based on continuous or cyclic strategies achieved up to 30% of initial moisture removals, which were lower than expected values. Along the start-up period, changes in aeration, intending to be adapted to the specific biological phase of the process, led to improved moisture removal, which were clearly reflected in the drying rate profiles. Moisture removal behaviour was directly related to temperature and airflow rates supplied, maximum drying rates usually coinciding with maximum aeration rates, supplied during thermophilic range of temperature period. Also, the elongation of the period in which the mixture was actively being dried achieved better overall absolute moisture removal results. Therefore, the experimentation carried out during this start up period made clear the need of a phase-adapting control system. Also, it was concluded that it would be necessary to find the most correct balance between aeration rates and temperature. High enough aeration rates would indeed be necessary to achieve effective drying of the bulk material, although an excessive aeration could lead to an excessive heat loss hampering biological moisture removal (Huiliñir and

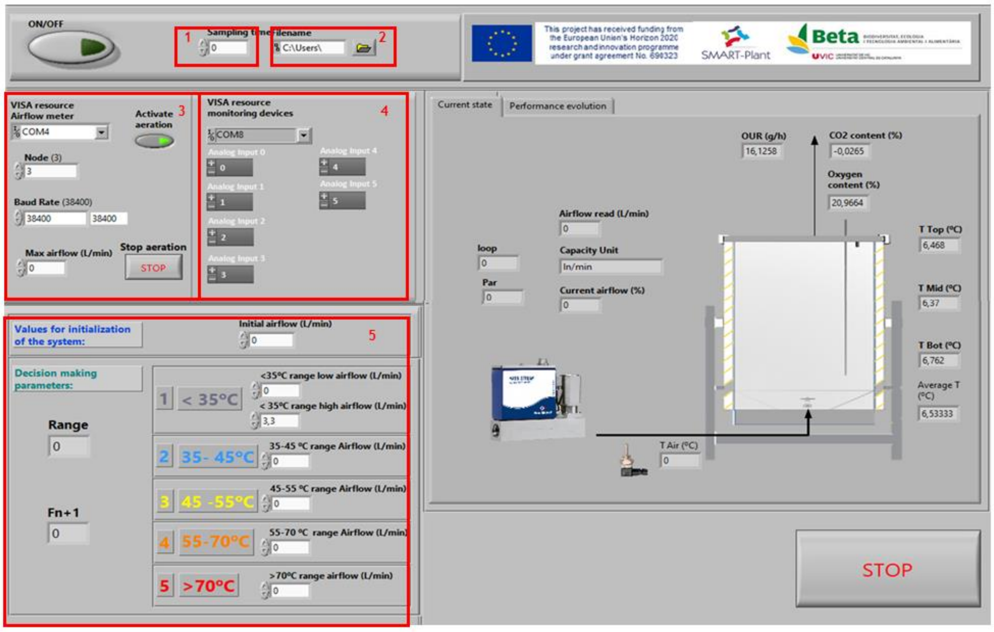
Villegas, 2014). Based on the information gathered during first trials, an automatic control system was developed based on temperature ranges. Using this control system, different aeration rates selected by the user were assessed in order to investigate in deep the effect of the aeration rate in the balance of bulk temperature and drying efficiency.

4.3.1.1 SOFTWARE INTERFACE AND CONTROL ALGORITHMS

In the development of pilot plant software basic and automatic aeration control systems were developed for both technological solutions. Basic systems were based on continuous or cyclic controls, as well as a control based on oxygen set-point in the case of composting control system. Automatic aeration control systems differed depending on the aim of each process as previously mentioned.

Figure 4-9 shows the user interface of biodrying (Figure 4-9a) and advanced composting systems (Figure 4-9b).

a)



b)

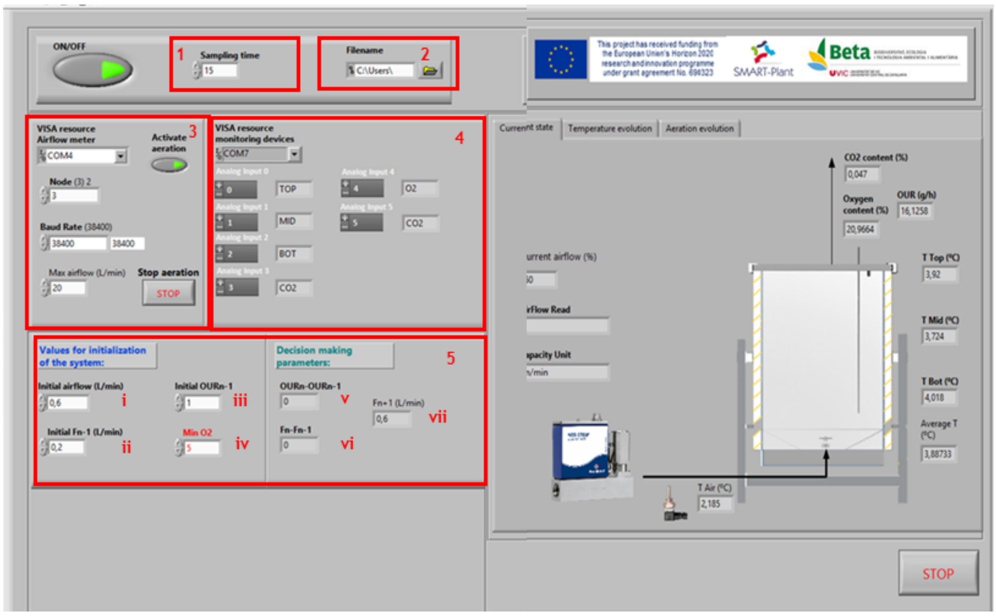
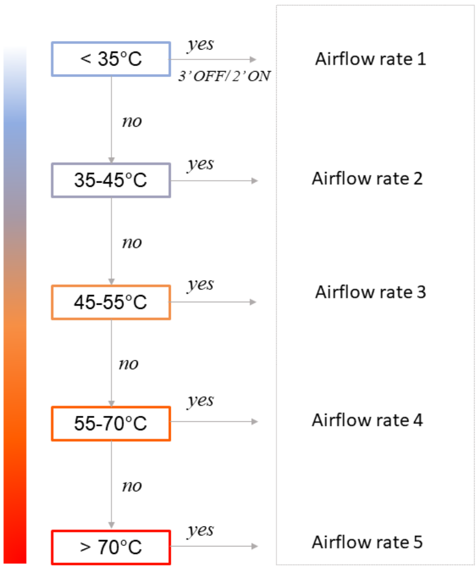


Figure 4-9. User interface of the automatic monitoring and controlling systems developed for advanced composting (a) and biodrying pilot plant (b). The parameters identified in red are (1) the data monitoring period; (2) the path and name of the excel file for data logging; (3) airflow meter communication parameters; (4) parameters for data acquisition communication (from Arduino UNO); (5) Parameters for automatic aeration systems, either based on OUR for advanced composting system or temperature range-based aeration system for biodrying.

Control algorithm diagrams for OUR based control system and temperature range-based control systems are shown in figure 4.10 a and b, respectively.

a)



b)

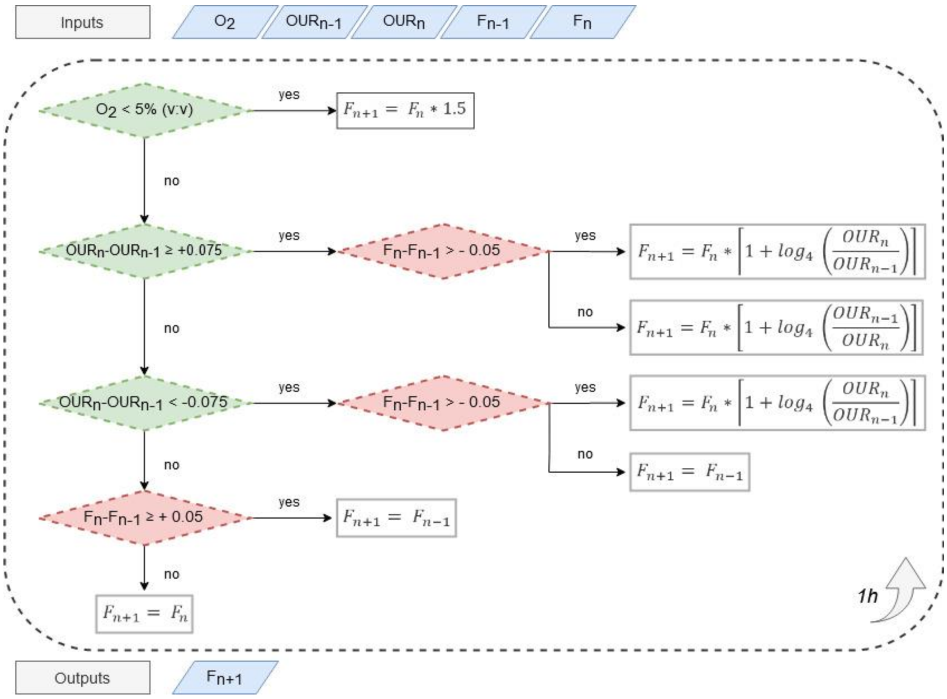


Figure 4-10. Control algorithms for (a) advanced composting process (adapted from Puyuelo et al., 2010), and biodrying process.

4.4 TROUBLESHOOTING

4.4.1 INFLUENCE OF ADVERSE WEATHER CONDITIONS IN BIOLOGICAL PROCESS

Due to excessive surface to volume relation and not enough thermal isolation (no thermal isolation in the air chamber) biological processes were adversely affected due to weather conditions, in particular during wintertime. Figure 4-11 shows an example of those adverse weather conditions faced during experimentation period.

To overcome this difficulty when biological processes were performed with sub-optimal conditions, some solutions were assessed:

1) Improve thermal isolation of reactors by adding an extra foam layer in the surface and bottom of reactor (50mm).

Figure 4-11. An example of the adverse weather conditions faced during experimentation

2) Setup of a permeable cover made by straw (Figure 4-12) for biodrying and by stainless steel for composting. The cover is able to retain part of the heat produced that otherwise would be lost. Straw allows evaporated water removal.

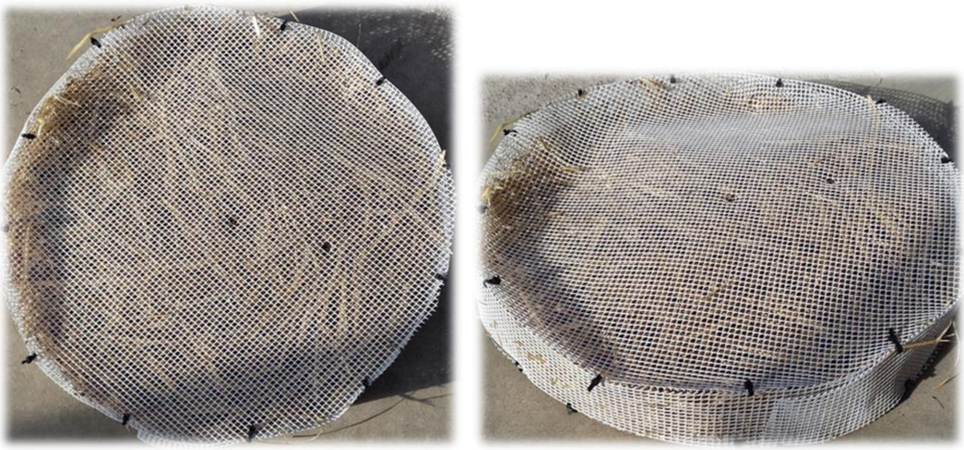


Figure 4-12. permeable straw cover built for biodrying reactor to avoid heat loss in

3) Pre- warming of influent air. Various attempts were done to warm the influent air before entering in the biological reactor. In general terms, electrical resistances were installed in metallic pipes made by steel or copper. This heating pipes were installed just before inlet air orifice. The final design of the air pre-warming system was set up as a copper spiral pipe (3m) shown in Figure 4-13.

Figure 4-13. Heating coil for inlet air pre-warming

Together with the previously mentioned modifications, warming of inlet air was assumed to be enough to help obtaining the thermal inertial that would allow the correct launch of the process.

5. ASSESSMENT OF BIOMASS FUEL PRODUCTION FROM SEWAGE SLUDGE THROUGH BIODRYING

5. ASSESSMENT OF BIOMASS FUEL PRODUCTION FROM SEWAGE SLUDGE THROUGH BIODRYING

Considering its rather high energy content (11-21 MJ kg^{-1}TS) (Frei et al., 2004; Navaee-Ardeh et al., 2006; Zhang et al., 2018), sludge shows the potential to be converted into a sustainable energy source provided that it is appropriately treated (Raheem et al., 2018). In this context, biodrying has raised as an innovative and presumably low-cost technology that allows the production of a biomass fuel (BMF) ready to be used in a biomass boiler. Although biodrying applied to sludges can be found in literature and its feasibility has been proved, the works published at a rather meaningful scale are scarce (Cai et al., 2012; Winkler et al., 2013) and in all the literature reviewed, the evaluation of the process efficiency is very limited (only based on MC reduction and VS consumption of bulk mixtures), and neither of them takes into account the quality of the product obtained and the overall energy consumed for the biodrying performance evaluation. However, if biodrying of sewage sludges will be regarded as an appropriate alternative to produce non-conventional biofuels able to penetrate the market, the maximisation of product quality and, thus, net energy recovery should be targeted. To this aim, the selection of the most appropriate raw organic wastes as feedstock for biodrying seems crucial, leading to satisfactory performances and potential end-products. Additionally, process scale-up is still limited due to techno-economic, environmental, and process monitoring and control burdens.

Bearing these objectives in mind, the application of fine sieves into the mainstream wastewater treatment assessed in SMART plant project through Cellvation® technology could be an example of a robust and easy to implement solution to make a promising feedstock available for its valorisation as BMF. Cellvation® technology is able to recover a sludge rich in cellulosic fibres, the so-called cellulosic sludge (CS), that presents high calorific potential (among other characteristics) and therefore it can be considered as an appropriate and promising feedstock for BMF production through biodrying.

In this Chapter, first, appropriateness of potential sludge feedstocks for biodrying process is explored in terms of process efficiency and product quality. Although CS is presented as an excellent candidate for being valorised through biodrying, it is important to remark the limited access to CS due to the still reduced application of Cellvation®-like technologies. Therefore, and in order to increase the impact and application of biodrying technologies, other existing conventional sewage sludge with presumable suitable characteristics have been also studied and assessed. Primary sludges from municipal WWTP and pulp and paper mill sludges from industrial WWTP were considered roughly analogous to CS, assuming to present attractive calorific potential. Finally, and with comparative purposes allowing the whole picture of conventionally produced sludges and studying the potential impact of this technology to the current sludge market, secondary and mixed sludges from municipal WWTPs were also evaluated.

Second, the use of fat-rich co-substrates is assessed as a possible solution for process enhancement for feedstocks not meeting the best characteristics.

Third, a deep study of biodrying processes applied to CS is specifically carried out, comprising the advanced and complete assessment of: i) process performance (including control and aeration strategies, monitoring parameters and performance indexes), ii) quality of end products, iii) techno-economic assessment and iv) polluting gaseous emissions. Special attention was given to improving and assessing different aeration strategies, assuming its major role in moisture removal during biodrying but also being aeration the major contributor to process energy consumption. At this point it is important to highlight that two innovative approaches and indexes have been proposed and applied, making possible a more complete and faster performance assessment of the process, including process energy consumption, quality of end-products and net energy recovery potential. These two indexes are the so-called energetic biodrying performance (EBI) and Biodrying Performance Index (BPI).

Finally, to provide a comprehensive picture of the process and facilitate and promote the scale-up and full application of biodrying technologies applied to low porosity-high moisture feedstocks, apart from the technical assessment of the biodrying process applied to several sludges, a thorough assessment of polluting gaseous emissions and economic feasibility of biodrying estimated for two potential case studies is also reported in this chapter. As a concluding remark, the strengths and weaknesses of the biodrying are described together with the windows of opportunities that this technological solution can open provided that some regulatory and market accessibility barriers are overcome.

Finally, it is worth mentioning that a relevant part of this chapter was published in the peer-reviewed journal Waste Management under the title “Biomass fuel production from cellulosic sludge through biodrying: Aeration strategies, quality of end-products, gaseous emissions and techno-economic assessment” (Guerra-Gorostegi et al., 2021).

5.1 BIODRYING PROCESS FEASIBILITY OF RAW SLUDGES OF DIFFERENT ORIGIN

5.1.1 COMPARATIVE ANALYSIS OF PHYSIC-CHEMICAL AND BIOLOGICAL CHARACTERISTICS OF RAW SLUDGES

In this section, comparative results of the targeted sludges are given. The sludges evaluated were: (i) cellulosic sludge (CS) (ii) primary sludge (PRS), (iii) pulp and paper mill sludge (PPS), (iv) secondary sludge (SS), and (v) mixed sludge (MS).

The physic-chemical characteristics of the sludges assessed are presented in Table 5-1. All the sludges studied were highly organic (60-80% VS in dry basis).

Respirometric tests are helpful since initial DRI and AT_4 values give insight on the biodegradability of raw organic wastes (Barrena et al., 2011b). Therefore, considering DRI values, PRS can be classified as highly biodegradable while the rest, showed moderate and low biodegradability (Barrena et al., 2011b). However, it is worth mentioning that the activation of biological activity was extremely sharp in CS and PPS, indicating a fast outburst followed probably by the depletion of most biodegradable organic material.

Additionally, calorific values (HHV) of PRS and MS were extraordinarily high (> 21 MJ $kgTS^{-1}$) compared to literature (Hao et al., 2018; Zhang et al., 2018) while CS and SS showed expected values (around 19 MJ $jgTS^{-1}$). The low HHV of PPS (13.1 MJ kg TS^{-1}) could be related to its high ash content due to the inclusion of deinking process sludge (Bajpai, 2015) which could lead to a biodried product with lower than desired HHV.

Table 5-1. Physic-chemical characteristics of the raw sludges assessed and a range of values of the bulking agent used in trials.

	Cellulosic sludge (CS)	Primary sludge (PRS)	Secondary sludge (SS)	Mixed sludge (MS)	Pulp & paper mill sludge (PPS)	Pruning waste (PW)
Dry mater (%, wb)	30.3 ± 0.6	18.64 ± 0.02	31.3 ± 0.1	24.9 ± 0.1	47 ± 1	64-91
Organic matter (%, d.b.)	92.7 ± 0.1	79.09 ± 0.05	74.6 ± 0.1	72.09 ± 0.08	65.7 ± 0.1	87-96
C/N	42.9	36.8	11.3	8.9	40.7	*
pH	6.89 ± 0.06	6.06 ± 0.02	6.50 ± 0.07	7.55 ± 0.06	6.77 ± 0.08	7.2-8.2
Conductivity (mS/cm)	0.542 ± 0.01	1.91 ± 0.04	3.22 ± 0.07	2.1 ± 0.1	1.07 ± 0.07	0.6-1.7
DRI ($gO_2\ kgVS^{-1}h^{-1}$)	2.8 ± 0.2	4.7 ± 0.1	2.5± 0.1	3.8 ± 0.3	2.4 ± 0.2	0.3-0.6
AT_4 ($gO_2\ kgVS^{-1}$)	191.5 ± 17.5	370 ± 7	234 ± 43	236 ± 23	139 ± 12	29-49
HHV	19.0 ± 0.1	21.67 ± 0.04	19.3 ± 0.2	21.45 ± 0.01	13.1 ± 0.1	17-19
LHV	3.73 ± 0.09	1.85 ± 0.01	4.01 ± 0.09	3.23 ± 0.00	5.18 ± 0.09	14-16

* Nitrogen content of PW was assumed to be negligible according to preliminary analysis performed

5.1.2 BIODRYING TECHNOLOGY APPLIED TO RAW SLUDGES OF DIFFERENT ORIGIN

Characteristics of biodrying mixtures assessed are detailed in Table 5-2. Mixing ratios were chosen according to the moisture content of sludges and pruning wastes aiming at the accepted optimum MC for biological aerobic processes (40-60%). Eventually, pruning waste used for the trial with PRS presented an excess of MC achieving, therefore, an initial MC over the targeted value (71.4%).

Table 5-2. Initial mixture characteristics of the biodrying trials performed with different types of sludges.

	Mixing ratio sludge: B.A (v:v)	Volume of sludge treated (L)	Experimental initial MC (%)	FAS (%)	Picture of mixture
Cellulosic sludge	1: 2.5	33.9	51.9	65,9%	
Primary sludge	1: 2.1	39.8	71.5	47.8	
Pulp & Paper mill sludge	1:0.9	52.3	40.7	78.0	
Secondary sludge	1:1.7	44.6	51.9	57.4	
Mixed sludge	1:3	25.3	50.6	75.1	

The pilot plant, the control system and the monitored parameters used are already described in Chapter 3. The main manipulated operational parameter to improve process performance was the airflow rate which was adapted to each temperature range (<35ºC, 35-45ºC, 45-55ºC, 55-70ºC and >70ºC) as defined in Figure 5-1. The specific aeration provided to each sludge assessed through biodrying was adapted according to the type of sludge, taking into consideration their organic matter content and the activity profiles obtained in respirometric tests in each case.

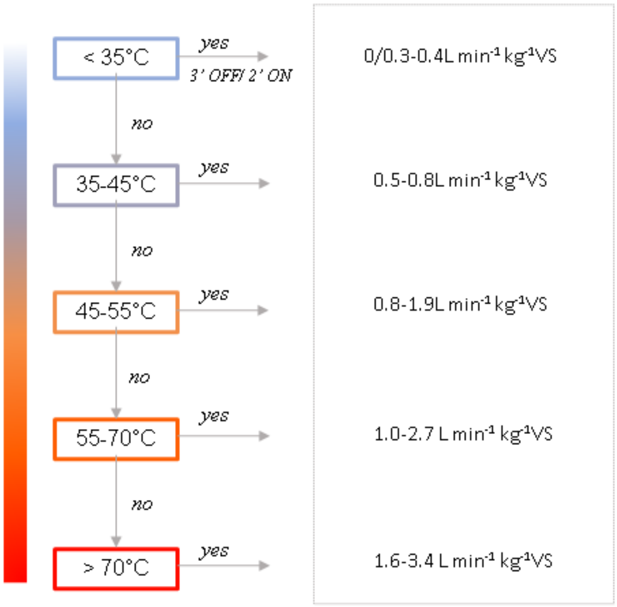


Figure 5-1. Selected aeration rates for biodrying trials of sludges from different origins.

Figure 5-2 shows the biodrying performance profiles of CS, PRS and PPS. Due to a lack of satisfactory biological process activation, the experimentation carried out with MS was discarded. Although the MS showed preliminary appropriate characteristics, the activation of the biological process did not occur, not being able to achieve thermophilic temperature ranges. This issue was unexpected for this kind of sludge and the presence of some toxic compound inhibiting the process was hypothesised. Due to unmeaningful results achieved in that case and since investigating the source of potential inhibition was out of the scope of the present thesis, the experiment was discarded.

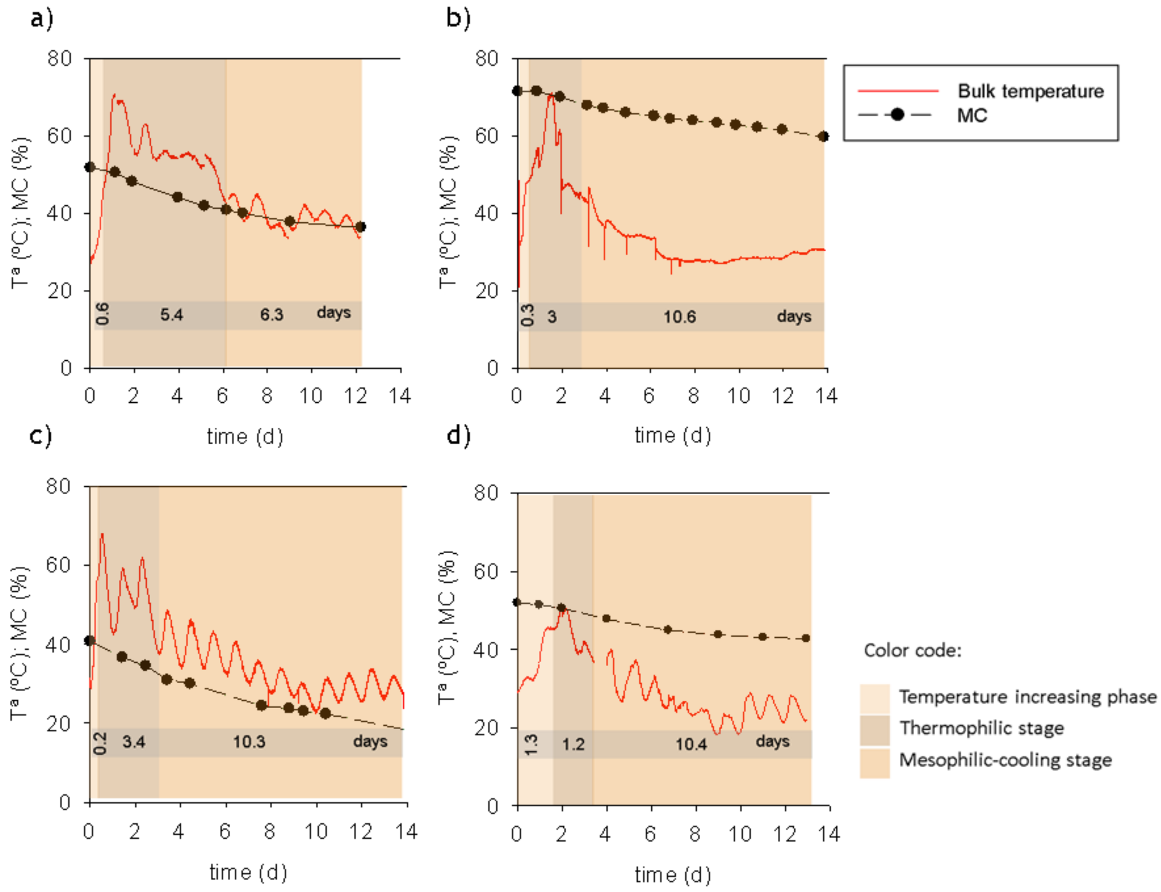


Figure 5-2. Biodrying performance profiles of a) cellulosic sludge (CS); b) primary sludge (PRS); c) Pulp and Paper mill sludge (PPS) and d) Secondary sludge (SS).

CS, PRS and PPS showed notably early and sharp temperature increases, with maximum temperatures close to 70ºC, and were able to maintain thermophilic bulk temperatures between 3 and 5.4 days. On the contrary, SS required 1.3 days of lag phase and the thermophilic stage was rather short, achieving maximum temperatures close to 45ºC. Taking into consideration the aim of the biodrying process, very fast activation (less than 24h) of the biological mixture would be desirable for a fast-drying process while conserving as much organic carbon as possible. Besides, a long maintaining highly thermophilic phase (more than 72h) would be preferable for an optimal biodrying performance.

Bearing these criteria in mind, SS biodrying did not achieve the minimum performance indicators to be regarded as a satisfactory biodrying trial. It could be hypothesised that the differences in the profiles described for SS compared to the rest of sludge might be due to the low availability of highly biodegradable organic matter due to the intensive treatment process in the wastewater treatment stage.

Enhancement of biodrying performance by using co-substrates could be a solution in those cases, being able to provide additional biodegradable organic matter and it is described in section 5.2.

To assess the technical efficiency, Table 5-3 summarises the values of (i) MC removal ratio and (ii) VS consumption ratio achieved for trials treating CS, PRS and PPS.

Table 5-3. Summary of biodrying efficiency indicators of CS, PRS and SS.

Sludge type		CS	PRS	PPS	SS
Moisture removal	absolute (total kg)	11.71	14.96	8.18	9.49
	ratio (of initial mixture)	55.0%	46.5%	68.0%	35.0%
VS consumption	ratio (of initial mixture)	14.9%	11.9%	10.6%	4.1%
	specific ratio (of initial sludge)	36.7%	27.8%	21.5%	14.3%
Heat required to remove moisture	Absolute (total MJ)	49.4	72.7	27.0	61.2

The experimentation carried out with PPS and CS achieved considerable moisture removals (68.0% and 55.0%, respectively) whereas PRS and SS trials were not as efficient (46.5% and 35.0%, respectively). Talking in absolute terms, the total mass of water in the trial treating PRS was considerably higher compared to the rest of the experiments. Moreover, compared to the best performing trials, 32-63% more energy would be required to remove the larger quantity of water in the case of PRS biodrying trial. However, the absolute mass of VS provided by the PRS that would be used for biological heat production was 19-32% lower than the rest of the sludges treated. Additionally, another explanation could be related to a sub-optimal bulk mixture structure of PRS trial (too wet with a relatively low FAS) that was previously related to a hampered drying performance, found to be independent of the airflow rate supplied at some MC point (Huiliñir and Villegas; 2015). Still, considering the overall results obtained with PRS, it could be considered a promising feedstock for biodrying. Finally, the results obtained in the trial treating SS were in accordance with the low temperature profile achieved in that trial.

Regarding biodegradation, Table 5-3 reports VS consumption ratios both referred to VS content from bulk mixture and specific VS content of the sludge fed. CS showed the highest VS consumption (20.9%) followed by PRS (11.9%) and PPS (8.5%). First, CS achieved the highest temperature profile with the longest thermophilic stage, which could be related to a very intense organic matter biodegradation. Second, the comparatively higher initial MC of PRS mixture could be the main responsible of the intense biodegradation that occurred in that trial (Navaee- Ardeh et al., 2006) and lower moisture removal capacity through convection due to the lower FAS of the bulk mixture.

Taking the results obtained altogether, authors dealing with dewatered sewage sludge at similar scales achieved equivalent moisture removal ratios between 49- 68% (Zhao et al., 2010; Zhao et al., 2011). However, in those cases, the VS consumption ratio was also comparatively high (19-37%). Other authors reporting lower VS consumptions, achieved also lower moisture removal ratios (Villegas & Huiliñir, 2014). Overall, it is important to remark that the results obtained permitted reaching a point beyond the state of the art. Particularly, improved sludge biodrying performances (in terms of moisture removal and VS conservation) were achieved at a representative scale of 100L confirming the reliability of the conclusions done.

Additionally, the current state of the art only refers to VS consumption ratio from both sludge and bulking agent. However, considering that the VS available for biodegradation would be almost exclusively from sludge, reporting consumption rates from the VS of the bulk mixture could be masking the results, not reflecting the reality of the process. In the worst cases, drawn conclusions might be false, since a lower VS consumption ratio from bulk mixture could be related to a higher sludge to bulking agent mixing ratio rather than to the appropriateness of the strategy used. Therefore, reporting a specific VS consumption ratio seems to be more suitable to assess the biodrying performance efficiencies.

5.1.3 QUALITY ASSESSMENT OF FINAL PRODUCTS

Apart from the process efficiency yields already discussed, the evaluation of the quality of BMF obtained is a key issue. Two main parameters are mainly considered in this regard, namely, a suitable (i) final MC in the end-product and (ii) calorific potential of the product by means of HHV and, particularly, LHV. Additionally, BMFs obtained in biodrying are usually described and characterised in the literature as a mixture with the bulking material used. In this work, the recovery of bulking agent and then, the separate production of sieved BMF, have been also studied. Moisture content and calorific values of mixed and sieved BMFs obtained in biodrying trials of CS, PRS, PPS and SS are shown in Table 5-4. The mixed BMF with the lowest MC was obtained from the PPS trial (17.9%) while the product from CS also achieved a rather low MC (36.4%) in the produced mixed BMF. For most conventional industrial boilers, the target MC values of the biomass fed need to be below 40% while the preferable values would be below 25-20%, specifically when pelletising of BMF is required (Rezaei et al., 2020).

It is worth mentioning that the trial treating PPS could have been stopped from day 9 on, as after those days MC was almost unchanged until the end, indicating probably that the hygroscopic limit (Vaxelaire and Cézac, 2004; Cai et al., 2012) could have been achieved by day 9. Accordingly, this could have prevented further VS consumption preserving a valuable part of the calorific value of the sludge.

Biodrying of PRS achieved the most humid mixed BMF (59.8% MC). Although this value has been suggested to be acceptable for sustained combustion in some biomass boilers (Navaee-Ardeh et al., 2010), the boiler efficiency would be low while it could be upgraded up to 74-80% when reducing the MC below 40% (Gebreegziabher et al., 2013).

Considering the results obtained, only the mixed BMF from PPS gathered the characteristics required for pelleting, while the rest need to be probably post-treated to finally meet the appropriate characteristics for densification processes, such as an additional extensive drying process or mixing with other materials (wood chips, etc).

The moisture content of sieved products was always higher than mixed products. Only sieved products of PPS achieved satisfactory moisture contents below 40%, while the rest of the sieved products would not reach the desired values and equally to the previously mentioned mixed products, other complementary MC adjusting strategies would need to be implemented.

Table 5-4. Quality assessment parameters in the sieved and mixed products of biodrying trials of different raw sludges evaluated.

Parameter	PRS		CS		PPS		SS	
	Sieved	Mixture	Sieved	Mixture	Sieved	Mixture	Sieved	Mixture
MC (%, w.b.)	72.0	59.8	56.3	36.4	27.8	17.9	61.6	44.9
VS (%, d.b.)	73.9 ± 0.5	77.8 ± 0.3	89.0	86.8	61 ± 1	68.8 ± 0.8	77.0	86.2
HHV ($MJ\ kg\ TS^{-1}$)	19.1 ± 0.3	19.94 ± 0.03	17.14 ± 0,05	17.08 ± 0.1	12.35 ± 0.04	13.2 ± 0.3	11.2 ± 0.1	13.6 ± 0.2
LHV ($MJ\ kg^{-1}$)	3.3 ± 0.1	6.12 ± 0.01	5.6 ± 0.03	8.7 ± 0.2	7.45 ± 0.04	9.5 ± 0.2	2.36 ± 0.06	5.8 ± 0.1




Only a few works were found reporting calorific values of biodried products. HHVs of mixed products were in the range of what could be expected for a biodried sludge (15-18 MJ $kgTS^{-1}$) (Zhang et al., 2018a; Hao et al., 2018). The LHVs of the BMFs obtained from CS and PPS were higher than those reported by Hao et al., (2018), except for the only product that they obtained when spent coffee grown was used as supplementary material, which most likely masked the result. Comparatively, pruning waste used could not mask product quality results in terms of calorific value as it always presented HHV values equivalent to the sludges assessed. Some of the products obtained achieved even calorific values almost as high as those found for thermally dried sludge and brown-coal (Winkler et al., 2013; Oladejo et al., 2019).

Except for the BMF from the biodrying of PRS, the HHVs of the rest of the mixed BMFs were equivalent to conventional biomass fuels such as forestry and agricultural wastes (nut shells, olive stone, sugar cane, etc.) and even some kind of wood chips and biomass pellets (Quiroga et al., 2010, Winkler et al., 2013; Gobierno de Navarra, 2015; Boumannchar et al., 2017). Additionally, the sieved BMFs of CS and PPS trials achieved LHV values above 4MJ kg^{-1}, considered satisfactory in literature (Hao et al., 2018) and comparable to the mixed biodried products in literature (Huiliñir and Villegas, 2014; Hao et al., 2018; Zhang et al., 2018b; González et al., 2019), meaning that the thermal valorisation of these materials, when appropriately treated through biodrying, could even be more competitive than previous works have demonstrated.

Summarising and concluding, (a) secondary sludge does not seem to gather suitable characteristics for a satisfactory biodrying performance. The use of complementary materials to enhance temperature profile and moisture removal is suggested as a good approach to improve performance efficiency in those cases and is described in the next section. (b) Conversely, primary sludge, cellulosic sludge and pulp and paper mill sludges, provided that initial and operational parameters are optimised for the most effective performance, seem to be suitable for their biodrying showing also high potential to obtain attractive biomass fuels.

5.2 BIODRYING PROCESS IMPROVEMENT TRHOUGH THE USE OF COMPLEMENTARY CO-SUBSTRATES

The use of adequate co-substrates can help to satisfactorily meet the biological and physic-chemical requirements to improve the process performance and reach the desired process efficiency and quality of end products (Barrena et al., 2011a). Some authors already improved the biodrying performance of sludges by adding food wastes, OFMSWs, or spent coffee ground (Ma et al., 2016; Zhang et al., 2018b; Hao et al., 2018). Apart from those, fat-rich wastes have been demonstrated to be useful highly biodegradable co-substrates in aerobic processes (García-Gómez et al., 2003; Gea et al., 2007), provided that their content in the mixture does not exceed a certain percentage (5-15%) (García-Gómez et al., 2003; Gea et al., 2007; Ruggieri et al., 2008). Thus, the use of fat-rich energetic amendments to enhance biodrying processes seems to be an interesting approach that has not been investigated yet. Moreover, satisfactorily enhancing the biodrying process efficiency of the sludges that otherwise could not be treated increases the impact of the alternative proposed in the current WWTP paradigm.

Thus, this section describes the first approach for the selection of potential lipidic co-substrates to enhance biodrying performance of secondary sludge, which by itself was not satisfactory and was considered the worst-performing sludge. To do so, first, different co-substrates were evaluated in terms of physic-chemical and biodegradability degree, and then 2 co-biodrying trials were conducted with the selected co-substrates. The co-substrates evaluated were (i) Diatomaceous earth (DE) from the oil refining process obtained from the OFMSW bio-methanation plant Can Barba (Terrassa, Barcelona); (ii) Winterization waste from an oil refining company (WW) (Santiga, Barcelona); (iii) Yogurt fat (YF) from the cow milk yogurt manufacture La Fageda (Santa Pau, Girona); and (iv) Coffee ground (CG) obtained from a local coffee shop (Vic, Barcelona). Table 5-5 shows a summary of physic-chemical and biological parameters considered for the selection of the most promising co-substrates for sludge biodrying.

Although YF and CG were the most organic wastes, their dry content was considerably lower than DE and WW, making low the actual content of VS available for microorganisms. Conversely, although DE and WW contained a high proportion of inorganic fraction (ash), the organic matter contained in WW and DE probably entirely consisted of fats and that was translated probably into considerable HHV. Regarding biological parameters, it seems that the use of co-substrates would significantly increase the biological activity of a sludge, either in terms of the DRI or cumulative terms.

Considering the results obtained, DE and WW were selected as the most suitable co-substrates to be assessed through the biodrying process. Additionally, their low density and low moisture content would potentially allow a significant enhancement of biological activity of biodrying mixture by adding a minimal amount of co-substrate, not substantially changing the major physic-chemical characteristics of sludge and biodrying mixtures. Similarly, the HHVs found for both co-substrates were similar to the values found for sludge. Thus, in our case, the addition of co-substrate would not mask the calorific potential of the product based on sludge.

Table 5-5. Physic-chemical characteristics of co-substrates and evaluated biological parameters of sludge-co-substrate mixtures: Diatomaceous earth (DE); Winterization waste (WW); Yogurt fat (YF), Coffee ground (CG).

Sample	%TS (w.b)	%VS (d.b.)	C/N	Density (kg L^{-1})	HHV (MJ kg^{-1}TS)	LHV (MJ kg^{-1})	DRI_{24} (g $O_2 kg^{-1}$SV h^{-1}) (increase %)	AT_4(gO_2 kg^{-1}SV) (increase %)
DE	97.7 ± 0.1	39.02 ± 0.01	683.7	0.631	12.66 ± 0.02	12.15 ± 0.02	6.9± 0.1* (139%)	820 ± 38 * (304%)
WW	95.1 ± 0.2	45.6 ± 0.6	1934.2	0.289	18.7 ± 0.2	18.52 ± 0.23	7.39± 0.09* (148%)	508± 31* (188%)
YF	5.39 ± 0.09	87.5 ± 0.2	8.5	1.004	n.a.**	n.a.**	6.50 ± 0.07* (130%)	332 ± 27* (123%)
CG	50.89 ± 0.03	98.18 ± 0.01	21.8	0.553	22.84 ± 0.03 ***		6.65± 0.04* (133%)	375 ± 9* (139%)

*Values given in the respirometric trials of sludge co-substrate mixtures. Values in brackets represent the enhancement (in percentage) of DRI_{24} and AT_4 values compared to sludge samples. ** Sample was too wet to analyse appropriately. *** Reported in Hao et al., (2018). n.a refers to not analised.

Specifically, 10% of the initial sludge weight was added in the mixture as co-substrate, which was in the range of the recommended addition for lipidic co-substrates for sludge composting (Ruggieri et al., 2008). Table 5-6 shows the main characteristics of the initial mixtures prepared for the assessment of biodrying performance of secondary sludge with the addition of co-substrates.

All co-biodrying trials were carried out using the dedicated pilot plant described in Chapter 3 together with the monitored parameters and control system used.

Table 5-6. Initial mixture characteristics of the biodrying trials performed with the addition of co-substrates to secondary sludge.

	Mixing ratio sludge: co-substrate: B.A (v:v)	Volume of sludge treated (L)	Initial MC (%)	FAS (%)	MIXTURE PICTURE
SS + WW	1: 0.3:1.7	45.8	67.3	51.8	
SS + DE	1:0.2:1.3	42.0	55.5	58.7	

Figure 5-3 shows the biodrying profiles and evolution of moisture content achieved during biodrying enhancement trials through the addition of co-substrates. As a reference, the profile of the biodrying trial performed with secondary sludge is also shown.

Temperature profiles achieved by the addition of co-substrates demonstrate the enhancement of the biodrying performance by maximising the peak temperature and maintenance of the thermophilic phase (Zhang et al., 2018), enhancing accordingly the MC removal. The temperature profiles obtained, though, were comparatively very different between them. When WW was used a satisfactory and very quick activation of the process was achieved, reaching 65°C by day 1.2 and maintaining thermophilic range of temperatures for 6.5 days.

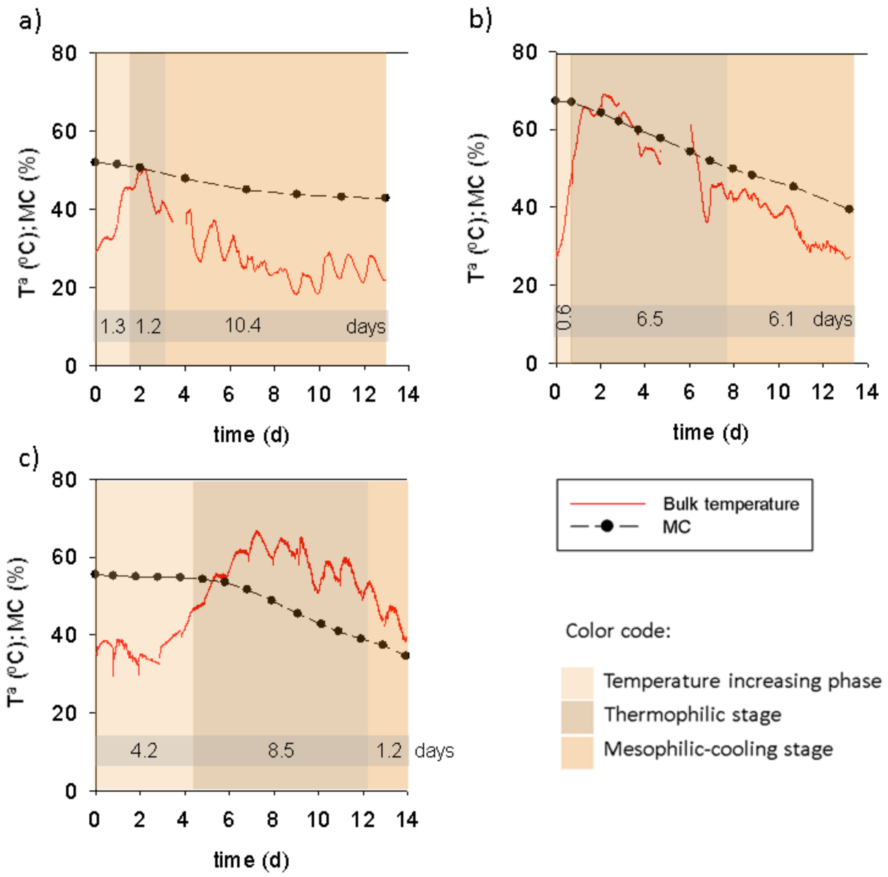


Figure 5-3. Comparative Biodrying performance profiles of the different sludge-co-substrate mixtures assessed: a) Secondary sludge (SS), b) Secondary sludge- winterization waste (SS-WW) and c) Secondary sludge - diatomaceous earth (SS-DE).

Conversely, in the case of DE use, the start-up of the process was delayed until day 3. One possible explanation for this differential behaviour compared with the supplementation with WW could be the adaptation time required by the microbial community to this material, or either the lack of immediate availability of the substrate. However, in this case, the thermophilic range of temperature was maintained for 8.5 days. In principle, this experimentation would not meet the criteria set before for satisfactory biodrying performance due to delayed activation of the process when using DE as co-substrate. However, using re-circulated bulking agent or part of the biodried product obtained as substrate-adapted-inoculum would be reasonable and it would probably reduce the lag phase to a satisfactory period shorter than a day.

The co-biodrying of SS with the considered materials led to reach moisture removal ratios above 60%. Moreover, compared to the rest of the trials assessed in this chapter, not only the temperature profile obtained but also the amount of moisture removed highlights the improvement of the process efficiency by the supplementation of SS with highly energetic co-substrates (Table 5-7). In fact, moisture removal ratios achieved were in the range of what other authors found for similar supplementation trials (Ma et al., 2016; Zhang et al., 2018; Hao et al., 2018; González et al., 2019). However, comparatively, the supplementation ratio presented here (10% in wet mass) is far below the supplementation done in literature (50-70% in wet mass) (Ma et al, 2016; Zhang et al., 2018) highlighting the value of the results obtained in the present work.

Table 5-7. Process efficiency parameters in sludge-co-substrate biodrying trials assessed.

Parameter		SS-WW	SS-DE
Moisture removal	absolute (total kg)	25.9	17.2
	ratio (of initial mixture)	68.9	60.8
VS consumption	ratio (of initial mixture)	4.6	8.3
	ratio (of initial sludge)	11.4	13.9

Parallelly, higher biological activity was expected to lead to an increased VS consumption (Zhang et al., 2018; Hao et al., 2018; González et al., 2019). Nevertheless, VS consumption ratios were in general terms markedly lower (4.6% and 8.3% for SS-WW and SS-DE mixtures) than the values found in co-biodrying literature (Ma et al., 2016; Hao et al., 2018; Zhang et al., 2018). 11.4% and 13.9% of the specific VS provided by the sludge and co-substrate was consumed in the trial mixtures supplemented with WW and DE, respectively. Conversely to what we were expecting, the use of co-substrates did not significantly lead to enhanced VS consumption as the specific VS consumption ratio values were lower than those found for conventional sludges (Section 5.1.1).

Both mixed end-products obtained from co-biodrying trials achieved the MC and LHV setpoints defined in the previous section (< 40% MC and LHV >4MJ kg^{-1}) (Table 5-8). The mixed products showed higher quality than the most of the products reported in co-biodrying literature in terms of MC (45-68% MC) (Hao et al., 2018; Zhang et al., 2018; Ma et al., 2019; Wang et al., 2020).

Table 5-8. Quality assessment parameters in the sieved and mixed BMFs from co-biodrying trials assessed.

	SS-WW		SS-DE	
	Sieved	Mixture	Sieved	Mixture
MC (%, w.b.)	51.5	39.4	40.7	34.3
VS (%, d.b.)	55.9 ± 0.4	64.30 ± 0.01	63.3 ± 0.2	62.4 ± 0.0
HHV (MJ kg^{-1}TS) (% increase)	12.94 ± 0.08 (116%)	14.56 ± 0.08 (107%)	15.63 ± 0.01 (140%)	15.3 ± 0.1 (112%)
LHV (MJ kg^{-1}) (% increase)	4.49 ± 0.04 (149%)	7.20 ± 0.05 (118%)	7.62 ± 0.01 (252%)	8.4 ± 0.1 (138%)

Even, the sieved BMF obtained from the SS-DE co-treatment reached an acceptable MC value (around 40%). Conversely, the MC values of the sieved BMF obtained from the SS-WW co-treatment were over 50% and a bit far from the desirable results expected.

Altogether, in all cases (mixed and sieved), the LHV values obtained correspond to materials to high energetic potential, especially for those values higher than 8 MJ kg^{-1}, leading to promising expectations in the use of biodrying technologies applied to the considered co-substrates.

Compared to co-biodrying results found in literature, calorific values of both mixed BMFs were above the values reported before for some co-biodrying trials (Zhang et al., 2018; González et al., 2019). Hao et al., (2018) reported higher LHV (10.16 ± 0.06 MJ kg^{-1}). However, since the highly energetic co-substrate (spent coffee ground with a HHV around 22 MJ kg^{-1}) supplementation proportion was 25% in that case, the calorific value of the sludge-based product was probably increased by the co-substrate used. Moreover, their working volume (17L) was not as representative as in the current work.

5.3 ENHANCEMENT OF SLUDGE BIODRYING PROCESS BY IMPROVED AERATION STRATEGIES AND INNOVATIVE INDEXES TO ASSESS THE PROCESS PERFORMANCE

Among the different sludges studied in previous sections, cellulosic sludge (CS) was considered the best candidate for further optimisation of the biodrying process through the evaluation of different aeration strategies.

As described in Chapter 1, effective biodrying performance depends on several mixture and process control parameters (Velis et al., 2009). Assuming suitable initial conditions (e.g., organic content of raw materials and matrix structure and porosity), water evaporation in the biodrying process depends mainly on two interrelated operational parameters: (1) airflow temperature (inlet and outlet) and (2) airflow rate. There are several studies assessing the effect of different aeration strategies and rates in the biodrying performance of sludge (Zhao et al., 2010; Sadaka et al., 2012, Sharara & Ahn 2012; Cai et al., 2013; Winkler et al., 2013; Huiliñir & Villegas, 2014; Zhang et al., 2015). Aeration is also recognised to be the main operational cost during biodrying processes due to its high electricity demand (Psaltis and Komilis, 2019) and therefore it should be carefully considered.

Additionally, obtaining high-quality end products is critical for an energetically self-sustaining process and achieving, in the best cases, considerable net energy production. However, sludge biodrying literature is mainly focused on exploratory purposes and only a few of them specify the heating value of the products obtained (Huiliñir & Villegas 2014; Hao et al., 2018; Zhang et al., 2018). In addition and to our best knowledge, none of the previous studies presents a thorough analysis of biodrying of sewage sludge, considering energy consumption and production aspects and moreover, there is not any previous methodology described to quantitatively assess the efficiency of the biodrying process in terms of net energy production and quality of end products.

Therefore, this section aims to go beyond the current state of the art i) assessing different phase-adaptive aeration strategies in terms of biodrying performance efficiency by using ii) two new efficiency indexes that consider not only biodrying efficiency in terms of moisture removal and VS consumption but also including energetic consumption and product quality in the evaluation.

5.3.1 SPECIFIC EXPERIMENTAL OPERATION

Briefly, CS was collected from the Wastewater Treatment Plant (WWTP) of Geestmerambacht, the Netherlands where Cellvation® cellulose recovery technology was implemented (additional information can be found in Chapter 3). Pruning waste (PW) was used as bulking agent. It was obtained from the MSW composting plant of the Parc Ambiental de Bufalvent located in Manresa (Barcelona), Spain. Sludge and bulking agent were mixed manually adjusting their initial moisture content to 50-60% (Villegas and Huiliñir, 2015) with a mixing ratio of 1:2.5 (v:v) of CS to pruning waste. Table 5-9 shows the initial physic-chemical characteristics of raw materials and initial mixtures in each case assessed.

Three different aeration strategies were adopted for cellulosic sludge biodrying: (1) to reach and maintain the highest temperature of bulk material and the longest duration of thermophilic phase (Strategy 1 or S1); (2) setting high airflows, tripling the values of S1 airflows (Strategy 2 or S2); and (3) a combined strategy where S1 airflows were maintained until the thermophilic phase was over (below 45ºC), followed by S2 airflows until the end of the process (Strategy 3 or S3). For S1, optimal aeration levels typically used during composting process were chosen. For the second strategy, the significantly higher aeration levels were expected to facilitate the extraction of the evaporated water and ultimately to improve water removal (Navaee-Ardeh et al., 2006). Finally, S3 aimed to combine the advantages of the two previous strategies. Figure 5-4 shows the specific aeration rates defined for each temperature range specified in the control algorithm (<35ºC, 35-45ºC, 45-55ºC, 55-70ºC and > 70ºC).

Table 5-9. Chemical characteristics of cellulosic sludge, bulking agent and initial mixtures.

Parameter	Strategy 1			Strategy 2			Strategy 3		
	CS	B.A	Mixture	C.S	B.A	Mixture	C.S	B.A	Mixture
Total solids (%, wb)	30.3 ± 0.6	80.3 ± 0.2	48.1	23.6 ± 0.2	81.86 ± 0.03	41.8	46 ± 1	84.44 ± 0.00	50.7
Volatile solids (%, d.b.)	92.7 ± 0.1	85.8 ± 1	88.6	88.5 ± 0.7	87.03 ± 0.01	87.6	93.1 ± 0.1	95.98 ± 0.00	94.6
NH_4^+-N (g N kg^{-1}TS)	2.3 ± 0.1	n.a	-	2.5 ± 0.2	n.a	-	1.94 ± 0.04	n.a	-
TKN (g N kg^{-1}TS)	11.8 ± 0.5	10.8 ± 0.6	11.4	2.68 ± 0.05	13.4 ± 0.6	8.5	10.7 ± 0.3	8.7 ± 0.3	9.75
C/N	42.9	43.4	42.5	180.5	35.5	56.3	47.6	60.7	53.0
HHV (MJ kg^{-1}TS)	19.0 ± 0.1	18.1 ± 0.4	18.5	18.1 ± 0.1	17.5 ± 0.6	17.51	18.7 ± 0.5	19.03 ± 0.1	18.85
LHV (MJ kg^{-1})	3.73 ± 0.09	14.0 ± 0.4	7.4	2.14 ± 0.03	15.9 ± 0.6	5.44	6.8 ± 0.2	14.8 ± 0.1	8.91
pH	6.89 ± 0.06	8.20 ± 0.05	-	4.71 ± 0.02	8.15 ± 0.05	-	5.24 ± 0.01	7.16 ± 0.03	-
Conductivity (mS cm^{-1})	0.542 ± 0.01	1.1 ± 0.1	-	1.57 ± 0.02	1.7 ± 0.1	-	1.6 ± 0.03	0.6 ± 0.1	-
Density (kg m^{-3})	850	179	422	653	210	346	471	130	362
DRI 24h max (gO_2 kg^{-1}VS-CS h^{-1})	2.8 ± 0,2	0.5 ± 0.2	-	2.3 ± 0.3	0.6 ± 0.1	-	3.0 ± 0.2	0.34 ± 0.08	-
AT_4(gO_2 kg^{-1}VS-CS)	191.5 ± 17.5	42 ± 5	-	176.3 ± 8.6	49 ± 7	-	212 ± 28	29 ± 6	-

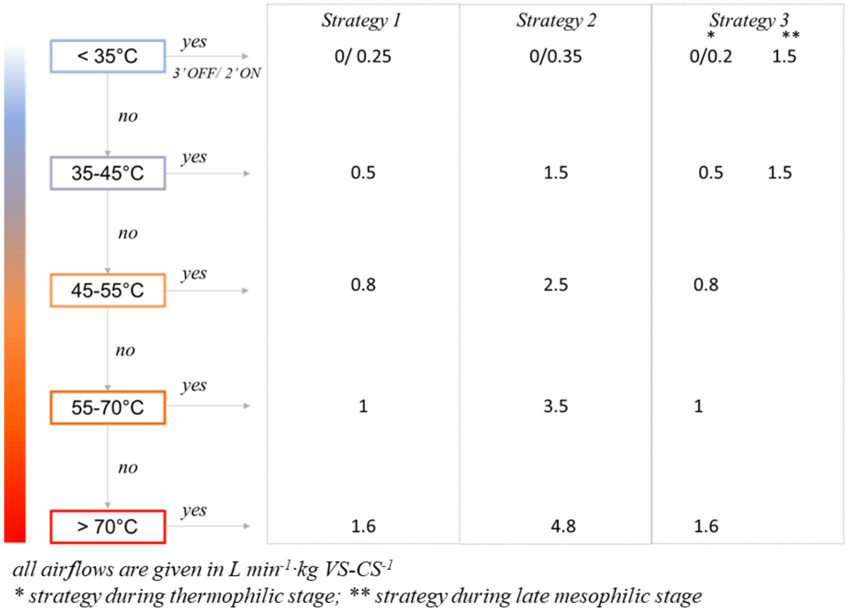


Figure 5-4. Specific airflow rates selected for each temperature range in each of the aeration strategies assessed.

5.3.2 PROCESS EVOLUTION: TEMPERATURE, MOISTURE CONTENT AND AIRFLOW RATES

Temperature profiles, moisture and airflow evolution obtained during all three trials are shown in Figure 5-5.

In general, temperature profiles are comparable to those found in the literature regarding sewage sludge biodrying (Zhao et al., 2010). Maximum temperatures achieved were equivalent for Strategy 1 and 3 (72ºC and 73.4ºC, respectively), and temperature profiles remained roughly similar until the process entered a late mesophilic stage, when the aeration rate clearly differed. As expected, the temperature profile with Strategy 2 was different, reaching 55ºC after 24h and 63.5ºC after 4 days.

Thermophilic temperatures were maintained for 5.3, 4.1 and 4.9 days with Strategies 1, 2 and 3, respectively, which were comparatively longer than in trials described in section 5.1 and literature (Zhao et al., 2010; Vilegas and Huiliñir, 2014).

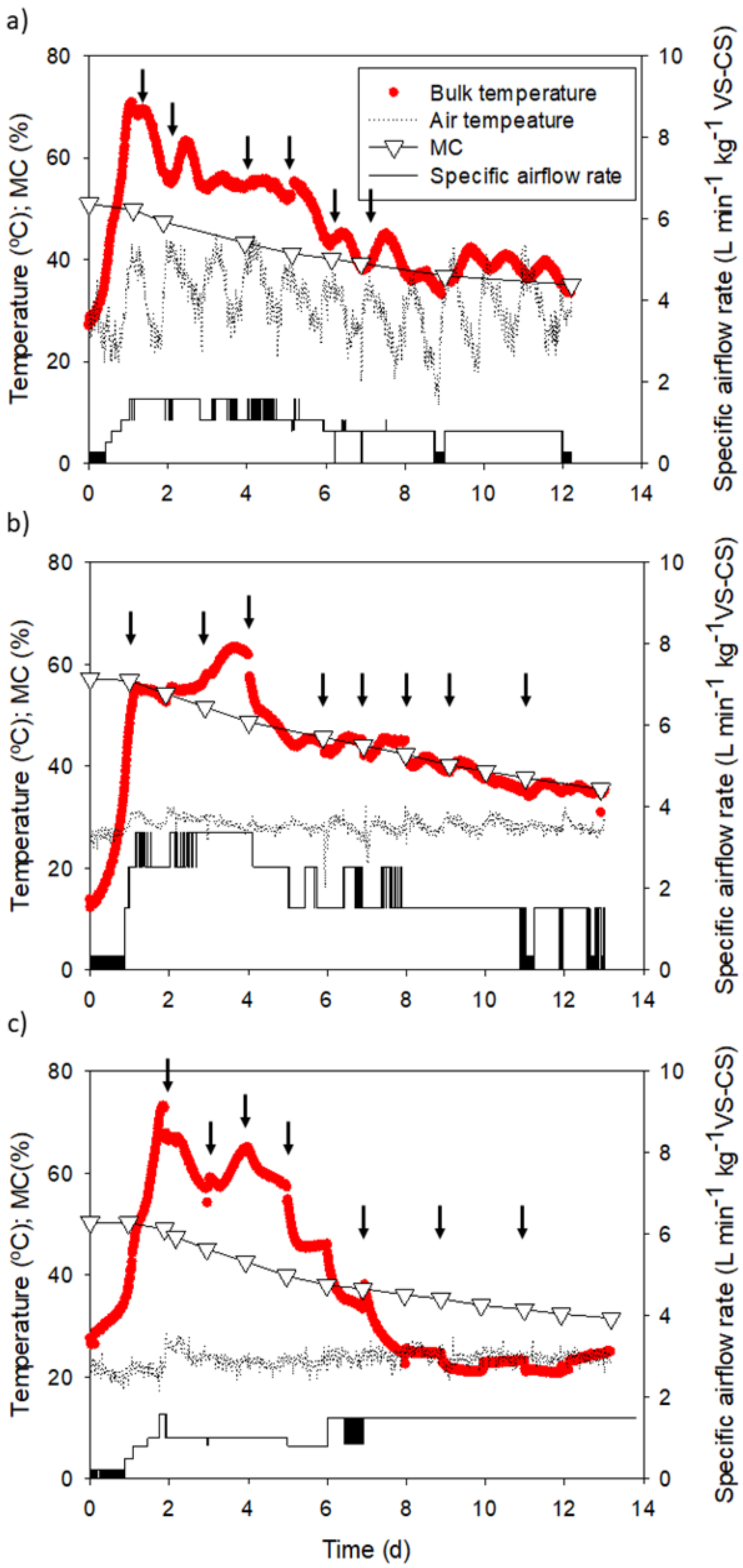


Figure 5-5. Temperature, airflow rate and MC profiles during experimental biodrying trials implementing strategies 1 (a), 2 (b) and 3(c). Arrows indicate whenever mixture was turned.

As shown in Figure 5-5, the airflow rates supplied were considerably different for the three strategies. The high airflow rates used in Strategy 2 (up to 3.5 L min^{-1} kg^{-1}VS-CS) probably led to a delayed temperature peak although they were not high enough to impair the biodrying process. Temperature profiles of Strategies 1 and 3 were equivalent until day six, when biogenic temperature generation when applying S3 was not able to counterbalance the heat loss caused by the high aeration rate supplied. Accordingly, with Strategy 3, it could be assumed that only convective drying occurred after day eight.

Considering the results shown in Table 5-10, the maximum moisture removal ratio was obtained when applying Strategy 2. MC removal ratios obtained were 55.0%, 62.4% and 57.5% for Strategies 1, 2 and 3 respectively. When comparing these results using a fixed VS mass consumption of 1.19 kg VS from Strategy 3, which was the minimum value obtained among the three strategies, the moisture removed by applying Strategy 2 would still be 38% and 11% higher than Strategies 1 and 3, respectively, demonstrating the high efficiency of Strategy 2. Moisture removal ratios obtained were in general in the high range of what was previously reported in the literature for conventional sewage sludges and similar scales, which usually achieved 17-68% moisture removal ratios (Zhao et al., 2010; Villegas and Huiliñir, 2014). However, results obtained in the current work are remarkable since they are almost comparable to best-performing trials reported in the literature, while they were achieved in significantly shorter experimentation periods (maximum 14d in the current work vs. 20d in Zhao et al., 2010) leading to higher VS conservation.

Due to the high temperatures achieved when applying Strategies 1 and 3 during the thermophilic stage together with its longer duration, this stage presented the highest MC removals. On the contrary, for Strategy 2, the MC removal ratios were balanced among the thermophilic and mesophilic-cooling stages. The difference between the moisture removal with Strategies 2 and 3 could be the unbalance in the biological heat production during the mesophilic stage which was limited when applying Strategy 3. Conversely, the VS conservation during the thermophilic stage of Strategy 2 probably allowed its later consumption maintaining mesophilic temperatures higher than those found with Strategy 3.

Table 5-10. Air supplied and overall mass balances in the different stages of cellulosic sludge biodrying trials operated with the different control strategies.

		Duration	Air supply		Weight loss	Water removal	VS consumption
		Days	Total m^3	Av. $m^3 kg^{-1} VS\text{-}CS\ d^{-1}$	kg (%)	kg (%)	kg (%)
S1	TOTAL	12.2	128.5	1.4	14.6	11.8 (55.0%)	2.8 (14.9%)
	TIP	0.6	1.4	0.3	1.2 (8.2%)	1.1 (5.1%)	0.1 (0.6%)
	THERMOPHILIC ST.	5.4	76.6	1.9	10.4 (71.2%)	8.4 (39.0%)	2.0 (10.8%)
	MESOPHILIC ST.	6.3	50.7	1.1	3.0 (20.6%)	2.3 (10.9%)	0.7 (3.5)
S2	TOTAL	13.0	207.4	2.7	13.6	12.3 (62.4%)	1.26 (10.0%)
	TIP	1.0	2.2	0.4	0.4 (2.9%)	0.4 (1.8%)	0.04 (0.3%)
	THERMOPHILIC ST.	4.1	101.8	4.2	7.3 (53.7%)	6.8 (34.2%)	0.5 (4.2%)
	MESOPHILIC ST.	8.0	103.4	2.2	5.9 (43.4%)	5.2 (26.3%)	0.7 (5.4%)
S3	TOTAL	13.2	211.6	1.7	12.2	11.0 (57.5%)	1.19 (6.9%)
	TIP	1.1	2.5	0.2	0.2 (1.6%)	0.2 (0.8%)	0.05 (0.3%)
	THERMOPHILIC ST.	4.9	64.6	1.4	8.8 (72.1%)	8.0 (41.6%)	0.8 (4.8%)
	MESOPHILIC ST.	7.2	144.4	2.1	3.2 (26.2%)	2.9 (15.1%)	0.3 (1.8%)

*The mass balances were done according to bulk mixtures, to be consistent with other authors. **TIP is referred to Temperature Increasing Phase

Cumulative oxygen consumption profiles were used to estimate VS consumption ratios in each stage of the processes (Table 5-10).

Considering BA biodegradation negligible (Ponsá et al., 2011), maximum VS consumption occurred when applying Strategy 1, while the lowest was achieved with Strategy 3. Again, when comparing the results with a fixed value of moisture removed (11 kg of water corresponding to Strategy 3), there would be a 57% and 52% lower VS consumption for Strategies 2 and 3, respectively, than in the case of Strategy 1. As expected, maximum VS biodegradation occurred during the thermophilic stage. The low VS consumption obtained when applying high airflow rates (along all stages of Strategy 2 and mesophilic-cooling stage of Strategy 3) reinforces what other studies previously found about high airflow rates limiting biological activity and degradation of organic matter (Huiliñir and Villegas, 2014; Zhang et al., 2020). Specifically, some authors reported significant cellulose and hemicellulose biodegradation during SS biodrying, contributing to the 20-30% of the total VS reduction (Zhang et al., 2020), in particular when low aeration rates were used (0.9 L min^{-1} $kg^{-1}VS$) due to enhanced mesophilic and thermophilic bacterial community (Ma et al., 2019; Zhang et al., 2020). It is worthwhile to highlight the potential effectiveness of all three strategies implemented in terms of VS conservation, as even the highest VS consumption found for Strategy 1 (14.9% of VS of the bulk mixture) was found to be lower than previous studies (normally between 15 and 40% of VS consumption) (Zhao et al., 2010; Villegas and Huiliñir, 2014).

5.3.3 EVALUATION OF PROCESS YIELDS

The most effective biodrying performance would consist on (i) maximising moisture removal, while (ii) limiting VS consumption, and (iii) minimising energy consumption. The Biodrying Index (BI) is usually reported in the literature as a quantitative performance efficiency index that interrelates the first two of the mentioned key parameters. Additionally, a new index integrating all three parameters mentioned is defined in this thesis: the Energetic Biodrying Index (EBI).

Process monitoring, by means of BI and EBI, for the three aeration strategies are shown in Figure 5-6 a and b, respectively.

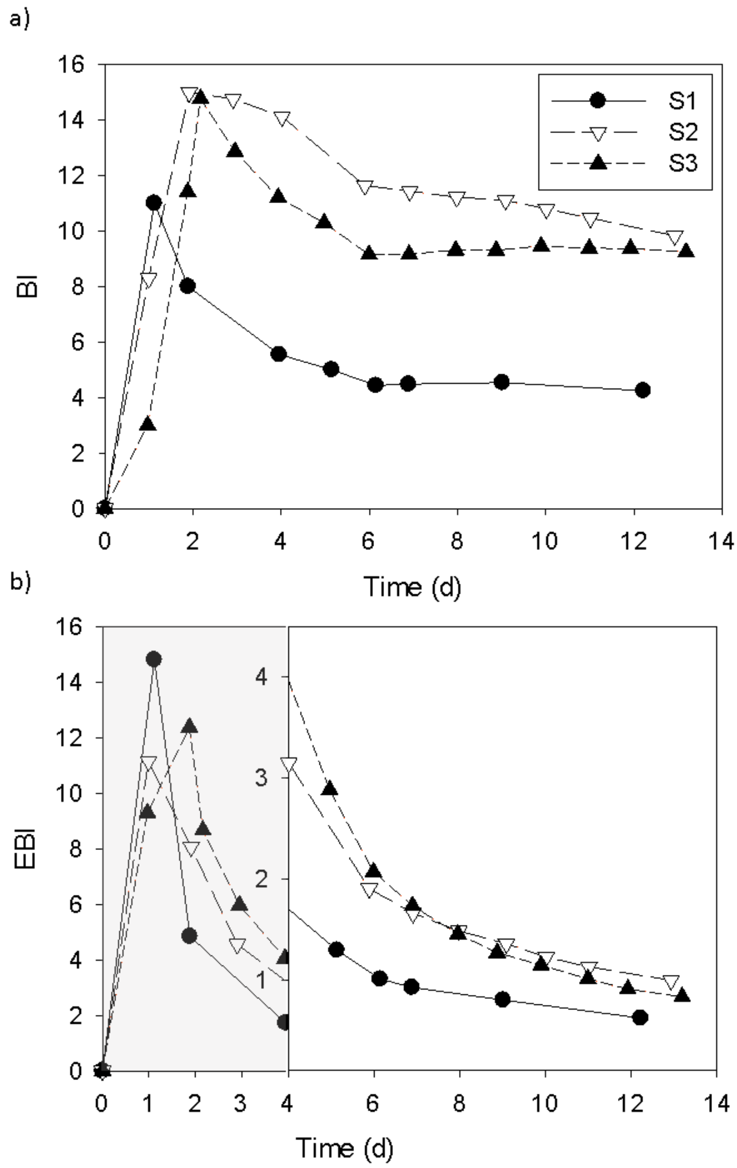


Figure 5-6. Daily comparative biodrying performance efficiency indexes: (a) Biodrying Index and (b) Energetic Biodrying Index, the grey area indicates roughly thermophilic stages during trials. Axes for thermophilic and the rest of the process of EBI profiles differ as they were adjusted to the values obtained in each phase.

When comparing the three strategies assessed, the best BI was obtained when applying Strategy 2 (9.8 kgH_2O kg^{-1}VS), followed closely by Strategy 3 (9.2 kgH_2O kg^{-1}VS), and finally by Strategy 1 (4.3 kgH_2O kg^{-1}VS). The lowest BI obtained for Strategy 1 was related to its higher VS consumption. The limitation of organic carbon mineralization is key to improving biodrying performance since it would affect the BMF quality. On the contrary, the best BI obtained by Strategy 2 is related to the high MC removal ratio and the moderate VS consumption. Accordingly, some authors also reported that airflow rate had more effect in water removal than in VS consumption (Zhao et al., 2010; Huiliñir and Vilegas, 2015; Zhang et al., 2018). Table 5-11 shows the comparative results obtained in the current work and meaningful studies in the literature.

All the strategies tested, especially Strategies 2 and 3, obtained satisfactory results in terms of process efficiency, due to high MC removal ratios, but more particularly due to the reduced VS consumption reported for similar wastes and scales (Zhao et al., 2010; Huiliñir and Villegas 2014).

However, some of the authors reported higher BI values (up to 20 kg H_2O $kg^{-1}VS$) (Villegas and Huiliñir, 2015), compared to those presented in the current study, although the difference was probably linked to lower VS consumption due to low temperature profiles achieved in that work which did not reach the thermophilic range of temperatures. The better drying performance of Strategy 2 compared to Strategy 3 during the late mesophilic stage is probably due to the difference in bulk temperature during that stage. Although airflow rates were equivalent, the below-mesophilic temperatures found in Strategy 3 clearly hampered the drying efficiency compared to Strategy 2. The depletion of most biodegradable VS during the first half of the Strategy 3 trial seemed to have reduced the biogenic heat production in later stages, leading to low bulk temperatures. Thus, during the late mesophilic stage, although high airflow rates can result in good MC removal ratios, a minimum bulk temperature around 35-40ºC seems to be necessary for an improved drying efficiency.

Figure 5-6b shows the EBI profile along the three biodrying strategies assessed, presenting clear differences among them, mainly related to their different aeration strategies. This EBI is proposed as an innovative and quantitative method to assess the efficiency of the biodrying process. Comparativelly to the BI, the EBI includes also an energy-consumption parameter. Considering the aeration to be a major contributor for the energy consumption associated to the biodrying technology, the EBI permits going beyond the information provided by the conventional BI enabling the effective comparison between different strategies assessed through the aeration that would be necessary to draw concrete conclusions. Overall, considering the energy consumption during the process and the EBI, the most efficient strategy was found to be Strategy 2 (0.99 kgH_2O $kg^{-1}VS$ kWh^{-1}), followed closely by Strategy 3 (0.85 kgH_2O $kg^{-1}VS$ kWh^{-1}).

Table 5-11. Comparison of overall biodrying efficiencies between the current study and other studies for similar high moisture organic wastes.

Reference	Raw material	Co-substrate (Y/N; which)	Scale	Specific aeration ($L\ min^{-1}\ kg^{-1}VS$)	Initial MC (%)	Final MC (%)	MC removal ratio (%)	VS consumption ratio (%)	BI $kgH_2O\ kg^{-1}VS$	EBI $kgH_2O\ kg^{-1}VS\ kWh^{-1}$
This study	CS	N	Bench (100L)	S1	51.9	35.1	55.0	14.9	4.3	0.62
				S2	57.8	32.5	62.4	10.0	9.8	0.99
				S3	51.8	31.5	57.5	6.9	9.2	0.85
González et al., 2019	SS	Y (DE)	Bench (100L)	Variable	54.6	35.9	58.8	14.5	5.7*	
Hao et al., 2018	Dewatered sewage sludge	Y (Spent Coffee Ground)	Lab (28.3L)	1.37	68.3-71.6	46.2	79.7	43.5	4.37	
Zhang et al., 2018	Dewatered sewage sludge	Y (MSW)	Lab (19.44L)	0.49-0.56	70	45.1-68.3	45.1-78.6	35.1-46.7	3.3-4.6	
Villegas and Huiliñir, 2014	SS	N	bench (64L)	1.05-3.14	58	51-52.5	16.9-24	5-14.3	16-20	
Huiliñir and Villegas, 2014	PPS	N	Lab (9L)	0.51-5.26	64.4-65.2	62-45	20-58	0-18	2.5-12.7*	
Winkler et al., 2013	Dewatered sewage sludge	N	Industrial (1900 m^3)	Variable	75	27.4	90.5	26	11.1*	183.6*
Cai et al., 2012	Sewage sludge	N	Pilot ($1.6m^3$)	Variable	66.1	54.7	46.1			
Sharara et al., 2012	CM	N	Bench (147 L)	0.05-1.5	55.9	28-35	70.7-79.1	26.3-41.9	2.6-3.2*	24.7-346.7*
Sadaka and Ahn, 2012	CM	N	Pilot (0.9 m^3)	0.65	59	30	59	8.1	15.5*	0.126*
	SM				60	28	58	5.8	19.8*	0.08*
	PM				61	40	53	5.9	19.0*	0.11*
Tambone et al., 2011	residual MSW	N	Industrial	Variable	32.7	17.8	65.5	29	2.26*	
Shao et al., 2010	MSW	N	Bench (150L)	1.4	73	48.3	79.9	37.3	7.02*	
Zhao et al., 2010	Dewatered sewage sludge	N	Bench (81L)	3.1-6.1	67.8	30.5-41.9	57.5-68.2	31.0-36.7	5.9-6.1*	
Frei et al., 2004	PPS	N	Pilot (1m3)		52.5-75.5	34.3-59.5	47-53.5	5.5-18	5.9-21.7*	

* refer to the values estimated from literature. CM refers to cattle manure, SM refers to swine manure, PM refers to poultry manure and MSW refers to municipal solid waste.

Conversely and although it had the lowest overall energy consumption, Strategy 1 obtained the lowest EBI value (0.62 kgH_2O kg^{-1}VS kWh^{-1}), particularly due to the high VS consumption, which did not particularly improve moisture removal efficiency. Energy consumption data in biodrying studies are scarce and only a few of them present some data. Sharara et al., (2012), determined energy consumption values around 1 kWh kg^{-1}mix, when treating livestock waste and using equivalent airflows as with Strategy1. Nevertheless, energy consumption data per water removed are more favourable, achieving 0.4- 0.9 kWh kg^{-1}H_2O in the current study compared to the 2.2-2.5 kWh kg^{-1}H_2O consumed by Sharara et al., (2012).

5.3.4 QUALITY ASSESSMENT OF FINAL PRODUCTS

Both mixed and sieved BMFs from the three aeration strategies were assessed and results are presented in Table 5-12.

Table 5-12. Quality assessment parameters of cellulosic sludge biodrying end-products.

Parameter	Strategy 1		Strategy 2		Strategy 3	
	Sieved	Mixture	Sieved	Mixture	Sieved	Mixture
MC (%, w.b.)	57.3	35.1	51.4	35.5	43.3	31.5
VS (%, d.b.)	88.7	88.7	85.5	84.7	91.9	94.2
HHV (MJ kg^{-1} TS)	17.1 ± 0.05	17.1 ± 0.1	17.2 ± 0.1	16.9 ± 0.3	16.9 ± 0.1	17.71 ± 0.00
HHV % lost from initial	9.9	4.2	4.9	3.4	10	6.1
LHV (MJ kg^{-1})	5.4 ± 0.03	9.5 ± 0.2	6.57 ± 0.06	9.4 ± 0.2	7.88 ± 0.07	10.6 ± 0.00
LHV % gained from initial	46.1	27.0	206.9	53.5	60.8	30.5
Specific production ratio* (kg TS product kg^{-1} TS-CS fed)	0.65	-	0.81	-	0.87	-
EP/EC** (kWh kWh^{-1})	1.8	-	2.1	-	3.1	-
BPI	15.7	-	25.6	-	35.1	-

*Specific production ratio refers to the overall dry mass of BMF obtained per dry mass of CS fed.

**EP/EC refers to the calculated amount of energy produced by BMF combustion per amount of energy consumed in the biodrying process, both in kWh.

Moisture content in all mixed BMFs was below the targeted 40% while the values obtained were indeed significantly lower than those reported in other studies treating sludges or Solid Recovered Fuels (SRF) (Shao et al., 2010; Cai et al., 2012; Villegas and Huiliñir, 2014; Zhang et al., 2020). Besides, all the produced mixed BMFs, presented high LHV values that in all cases were above the minimum 4MJ kg^{-1} set before as acceptable criteria for BMF.

Specifically, the mixed BMF produced using the three aeration strategies presented LHV values higher than 9MJ kg^{-1} LHVs, being these values equivalent to those presented in section 5.1 and higher than those found in the literature for conventional sewage sludge and pulp and paper mill sludge biodried products (5.5-7.5MJ kg^{-1} in the best cases) (Huiliñir and Villegas, 2014; Zhang et al., 2018; González et al., 2019). Additionally, the mixed product obtained with Strategy 3 reached 10MJ kg^{-1} which can be classified into group 4 according to the SRF quality standard (EN 15359).

Compared to conventionally used biomass fuels, the mixed BMF obtained from the trial using Strategy 3 has a calorific value comparable to wood chips (40%MC) or forest waste (47.6% MC) (Quiroga et al., 2010, Gobierno de Navarra, 2015).

As mentioned in previous sections, literature related to biodrying processes is normally reporting results of mixed BMF (including the biodried sludge and the bulking agent). Nevertheless, bulking materials (normally pruning waste or wood chips) may hide or dilute the real values corresponding to the waste streams that are being valorised as BMF. Sieved BMFs consistently presented higher MC and consequently, lower LHVs than mixed BMFs. Nevertheless, the best sieved BMF with the highest LHV was the one obtained with Strategy 3, which was comparable to previous mixed biodried products reported in the literature (Huiliñir and Villegas, 2014; Hao et al., 2018; González et al., 2019) and to some of our previous trials reported in Section 5.1.2.

As it was reported elsewhere, VS consumption minimization is a critical point in the biodrying process, since the calorific value in the end-product is directly dependent on the final VS content (Huiliñir and Villegas, 2014). However, according to the results obtained, it can be concluded that although LHV is affected by the VS content of the final material, the effect of MC of the sample in the LHV is clearly more important.

Additionally, the energy production per energy consumed (EP/EC) and the biodrying performance index (BPI) were presented in the current work as the most suitable indicators for the evaluation of the process by means of drying performance, end-product quality and net energy production. Considering the results presented in Table 5-12, nearly 2 to 3 kWh can be recovered from sieved BMFs per each kWh consumed in the process, demonstrating the energetic efficiency of the process in all three cases. Moreover, the new BPI proposed in this work could be used as an overall biodrying efficiency indicator, facilitating decision-making and efficiency comparisons. It considers all the main factors involved in biodrying performance, as well as product quality parameters, thus energy recovery potential of the end-products obtained. The best BPI was achieved when applying Strategy 3 (35.1) mainly due to its high specific production ratio. Comparatively, BPI values for Strategies 2 and 1 were 27% and 55% lower than Strategy 3 values, respectively.

In general terms, although Strategy 2 seemed to be the best performing strategy in terms of BI and EBI, when considering also end-product quality parameters and energy production potential, Strategy 3 was considered the best performing strategy.

Figure 5-7. Pelletised end-products obtained from cellulosic sludge biodrying.

5.4 ASSESSMENT OF GASEOUS EMISSIONS ASSOCIATED WITH BIODRYING PROCESS

Apart from the technical assessment of the best performing biodrying strategy, the environmental assessment associated with the process implementation, by means of gaseous emissions determination, becomes absolutely necessary to demonstrate that biodrying can be considered as a sustainable sludge management alternative. Bearing this in mind, ideally, the biodrying process treating sewage sludge should seek: (1) a low energy consumption and harmful gaseous emissions, allowing, in turn, (2) the production of a high-quality product ready to be used, leading to renewable resource recovery solutions.

Mainly direct gaseous emissions and indirect emissions through consumption of electricity are the parameters contributing to environmental threats associated with the biodrying process. Regarding direct gaseous emissions, first, CH_4 and N_2O emissions are key given their contribution to the global warming potential (GWP) of the process. Second, NH_3, H_2S and total Volatile Organic Compounds (tVOCs) are associated with direct environmental impacts such as terrestrial and aquatic eutrophication, photochemical ozone formation and global warming potential. Additionally, these indicators are recognised to play a crucial role in the social acceptance of this kind of technology, especially when full-scale plants are built, given the social worries caused to neighbourhoods near the treatment plants due to unpleasant odour formation and other possible gaseous emissions.

All in all, information regarding polluting gaseous emissions in biodrying is extremely scarce, in particular when treating sewage sludge. In fact, only our previous work on gaseous emissions during biodrying of mixed sludge deals with most relevant gaseous emissions profiles and cumulative emission factors during the process (González et al., 2019b). Accordingly, this thesis aims to provide meaningful data in this regard, assessing direct and indirect gaseous emissions monitored in relevant biodrying trials. Specifically, CH_4 and N_2O were monitored in the worst-case considered scenario, thus the CS biodrying trial applying Strategy 1, the one with comparatively lower air supply. Results obtained would be meaningful enough to evaluate the potential impact in (GWP) of the biodrying process in general terms. Also, indirect emissions through the monitoring of electricity consumption in every trial were assessed. Additionally, NH_3, tVOCs and H_2S, were monitored and shown for all the trials carried out with CS, PRS and PPS, so the effect of specific sludge characteristics in their emissions could be assessed. Finally, to better assess the social burden that this process could produce, units of odour emitted were also monitored in the worst-case scenario as well as characterisation of tVOCs emitted in the day with maximum emission rate (thermophilic peak). Therefore, the data provided in this section provide novel, significant and helpful qualitative and quantitative information related to sludge biodrying process.

5.4.1 POLLUTING GASEOUS EMISSION RATES AND PROFILES DURING BIODRYING: WORST-CASE SCENARIO

For the description of emission profiles, the worst-case scenario was recognised to be the most meaningful scenario. This worst-case scenario was the one applying Strategy 1 in CS biodrying. This trial was the one supplying the lowest aeration rates, increasing the probability to develop anaerobic conditions, compared to the rest of the strategies. Additionally, it was the strategy associated with the longest thermophilic range maintenance, and therefore expecting an enhancement of ammonia volatilisation.

Figure 5-8 shows the emission profiles of relevant gaseous compounds and odorant units during the biodrying trial applying Strategy 1. For comparative purposes, also temperature profile is shown. Emission rates are given in daily mg of pollutant emission per kg of TS of sludge treated.

Maximum emission rates of CH_4 and N_2O occurred during the first hours (0h for N_2O and 24h for CH_4). These maximum emission values are probably related to anaerobic conditions during dewatering and shipping of raw materials (Awasthi et al., 2016; Han et al., 2018b). After adjusting initial porosity by using a suitable bulking agent, CH_4 and N_2O stored in sludge were probably stripped out by forced aeration (Yuan et al., 2016; Han et al., 2018b; González et al., 2019). Due to the shipping/storing conditions, the emission peaks monitored for N_2O and CH_4 were 74% and 26% higher than in our previous work treating conventional mixed sludge (González et al., 2019). Once compounds were stripped out, their emission rates decreased to barely detectable levels. Yet, at day 6 a second very small peak of N_2O was detected coinciding with the change from thermophilic to mesophilic conditions as previously related to a restored nitrification activity (Ahn et al., 2011; Yuan et al., 2016; González et al., 2019).

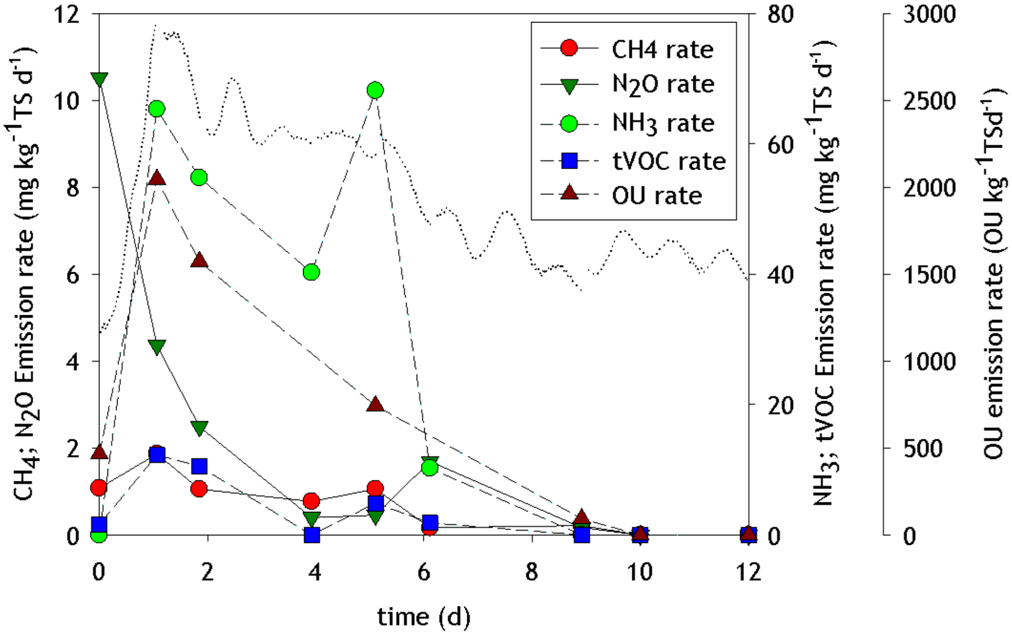


Figure 5-8. Daily emission rates of CH_4, N_2O, NH_3, tVOCs and odorous compounds during CS biodrying when implementing Strategy 1. The dotted line shows the corresponding temperature profile achieved as a reference.

NH_3 and tVOC profiles followed the same behaviour already detected in our previous work (González et al., 2019). Thus, maximum NH_3 emission peaks were related to thermophilic temperatures, while tVOCs were emitted mainly in the first days of operation (Maulini-Duran et al., 2013; González et al., 2019). Odour emissions achieved their peak together with the temperature, coinciding with NH_3 and tVOCs maximum daily emission rates. H_2S was not detected at any moment. NH_3 emission rates were rather high whenever thermophilic temperature ranges were maintained. A second NH_3 emission peak occurred on day 5, coinciding with the second biological activity peak found during the respirometric test. Considering the respirometry profile shown in Figure 5-9, two main biodegradation mechanisms can be identified: (i) the first activity peak was probably related to easily available organic matter consumption, most likeably short-chain Volatile Fatty Acids (VFAs) produced during CS shipping period. (ii) the second activity period could be related to more complex organic matter consumption that would release ammonium that would be eventually volatilised in form of ammonia facilitated by thermophilic temperatures.

In general terms, maximum emission rates of NH_3, tVOCs and odour emission rates were notably lower than our previous biodrying work with conventional sludge (González et al., 2019). The difference in NH_3 emission rate levels could be related to the composition of CS, which had comparatively a lower nitrogen content than conventional sludge.

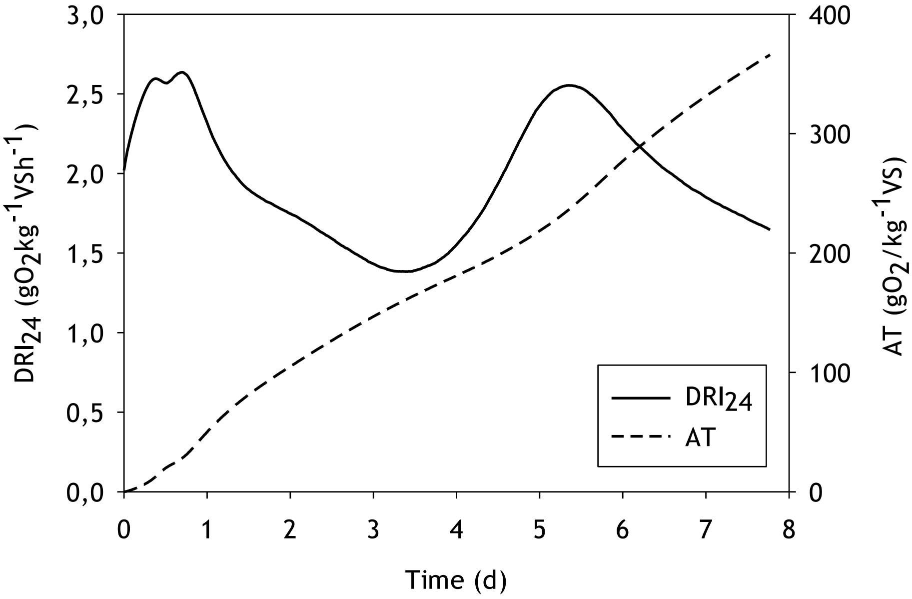


Figure 5-9. DRI24 and at profiles of CS treated when implementing Strategy 1.

5.4.2 CUMULATIVE EMISSION FACTORS OF POLLUTING GASEOUS COMPOUNDS DURING BIODRYING OF SELECTED SLUDGES

In cumulative terms, overall emission factors calculated for CH_4 and N_2O in CS biodrying applying aeration Strategy 1 were 0.007 kgCH_4 t^{-1}TS and 0.018 kgN_2O t^{-1}TS, respectively (Table 5-13). Even though it was considered the worst-case biodrying scenario, cellulosic sludge biodrying emits 74.5% less CH_4 and 70.2% less N_2O than conventional sewage sludge biodrying trials (González et al., 2019). Composting, which could be considered as an alternative to biodrying technologies for sludge valorisation, showed notably higher cumulative CH_4 and N_2O emission factors in a twin reactor set up for the equivalent process period of 14 days (up to 90% higher)(González et al., 2020). Two differential factors should be noted in that difference: on one hand the availability of nitrogen in those feedstocks and on the other hand, although Strategy 1 was considered to be the

Table 5-13. Cumulative polluting gaseous emission factors (per dry sludge mass) during various sludge biodrying trials and comparative relevant values in literature.

	CH_4	N_2O	direct CO_2 eq	indirect CO_2 eq	overall CO_2eq	NH_3	H_2S	tVOC	odours
	$kgCH_4\ t^{-1}TS$	$kgN_2O\ t^{-1}TS$	$kgCO_2eq\ t^{-1}TS$	$kgCO_2eq\ t^{-1}TS$	$kgCO_2eq\ t^{-1}TS$	$kgNH_3\ t^{-1}TS$	$kgH_2S\ t^{-1}TS$	$kgC\text{-}VOC\ t^{-1}TS$	$OU\ t^{-1}TS$
Biodrying of CS Strategy 1	0.007	0.018	0.025	0.23	0.26	0.3	0.000	0.036	8118
Biodrying of CS strategy 2				0.41		0.051	0.000	0.011	
Biodrying of CS Strategy 3				0.30		0.007	0.000	0.016	
Biodrying of PRS				0.49		0.36	0.0002	0.4	
Biodrying of PPS				0.20		0.004	0.000	0.004	
González et al., 2019 biodrying conventional sludge	0.045	0.10	28.22			1.23	0.002	0.14	3.10E+07
González et al., 2020 composting conventional sludge (d14)	0.11*	0.1*	45.9*			5.1*	0.17*	6.20*	2.61E+07*

*data calculated from raw data

worst-case scenario, aeration rate during maximum biological activity period was 13% higher in the biodrying trial of CS applying Strategy 1 compared to the composting trial.

Energy consumption during biodrying trials was monitored to estimate indirect CO_2eq emissions. Equivalent GWP was calculated using the value accepted for LCA studies based on PEF (0.28 gCO_2 kWh^{-1}) (Environmental Footprint (EF) secondary data sets version EF 2.0., European Commission, 2020d). Energy consumption was uniquely derived from the air compressor since energy consumption from monitoring equipment was negligible. Strategy 1 emitted indirectly 0.23 CO_2eq t^{-1}TS. Comparatively, Strategy 2 emitted almost the double indirect CO_2eq, while values found for Strategy 3 were in between. Comparatively, the biodrying trial of PRS was the highest emitting trial given the higher airflow rate required to guarantee an effective air supply throughout its too wet initial mixture. Compared to composting trials in twin reactors carried out in our facilities (González et al., 2020), biodrying of sewage sludge seems to be in general a less GHG emitting alternative in the equivalent process period (approximately 14days) (three orders of magnitude lower in the biodrying process), even though biodrying is higher energy-demanding technology. However, it should be clarified that the difference on the duration of performances of both technologies would be a major reason of this difference. Additionally, the enormous energy demand of conventional drying methods would certainly be associated with high GWP, probably markedly higher than biodrying.

The highest NH_3 emission factor was found when implementing Strategy 1, which emitted 83% and 98% more NH_3 than when using Strategies 2 and 3 (Table 5-13). High aeration rates have been previously related to a lowered ammonification rate (Yuan et al., 2016) and this could be the reason of the differences found. When comparing those NH_3 emission factors to our previous mixed sewage sludge biodrying work, they were notably higher in the last (González et al., 2019), probably due to the higher availability of nitrogen compounds in conventional secondary sludges. PRS and PPS showed CS equivalent NH_3 emission factors, being the differences mainly related to the nitrogen content in sludge and thermophilic temperature profile achieved during the biodrying process.

Again, in accordance with results found for GHG, the worst-case scenario in terms of aeration (biodrying trial of CS with Strategy 1) achieved the highest emission factor of tVOC, while Strategy 2 and 3 achieved, respectively, 69% and 56% lower values. tVOC emissions have been related to anoxic conditions (Maulini-Duran et al., 2013). Compared to CS, biodrying of PRS emitted notably more tVOCs, while the emission factors found for PPS biodrying were in the low range of the values found in CS biodrying. Considering the notably lower FAS value achieved in the bulk mixture of PRS compared to that of CS, it would be reasonable to suggest that anoxic zones would be developed more easily in the first mixture mentioned, leading to probably a more intense emission of VOCs. Accordingly, compared to our previous biodrying work (González et al., 2019), tVOC emissions were lower except for the PRS biodrying trial.

NH_3 and tVOCs are often related to odours emitted during biological processes. Compared to our previous work on the biodrying study of conventional mixed sludge (González et al., 2019), odour emissions with CS were significantly lower. Odour emission factor was also lower than in advanced composting of conventional mixed sludge with a twin reactor set-up (González et al., 2020) (Table 5-13). Finally, to assess and identify the source of potential social burdens associated with sludge biodrying, identification and quantification of potential undesired compounds was carried out during process performance.

Bearing this in mind, a characterisation of VOC families was carried out, identifying and quantifying these compounds in a sample taken during the period of highest emission rates. Figure 5-10 summarises the distribution of VOC families detected in that sample.

12 VOC families were identified in total, being terpenes, alkanes, carboxylic acids and ketones the most abundant. Siloxanes, furans and alcohols were found in a very low percentage (< 0.4%), while esters were not detected. Aromatic hydrocarbons, N compounds, S compounds and alkenes emissions were low, accounting for around 1% of the total VOC emissions each.

In our previous biodrying work, also terpenes were predominant compounds, as well as it happens in sludge composting processes (Maulini-Durán et al., 2013; González et al., 2019; 2020). Terpenes are known to be typical intermediates produced in composting processes during aerobic degradation (Shiavon et al., 2017) although they have also been related to the type of bulking agent used for the process and its mixing ratio (Maulini-Duran et al., 2013; González et al., 2019). Comparing to VOC characterization results from our previous biodrying and composting works, S compounds were higher (up to 10% of the VOCs emitted) in those studies treating conventional mixed sludge (González et al., 2019; 2020). The low relative abundance of S compounds of this study, together with the absence of H_2S could indicate a low organic sulphide content in cellulosic sludge, Thus, elemental analysis of CS would confirm that statement.

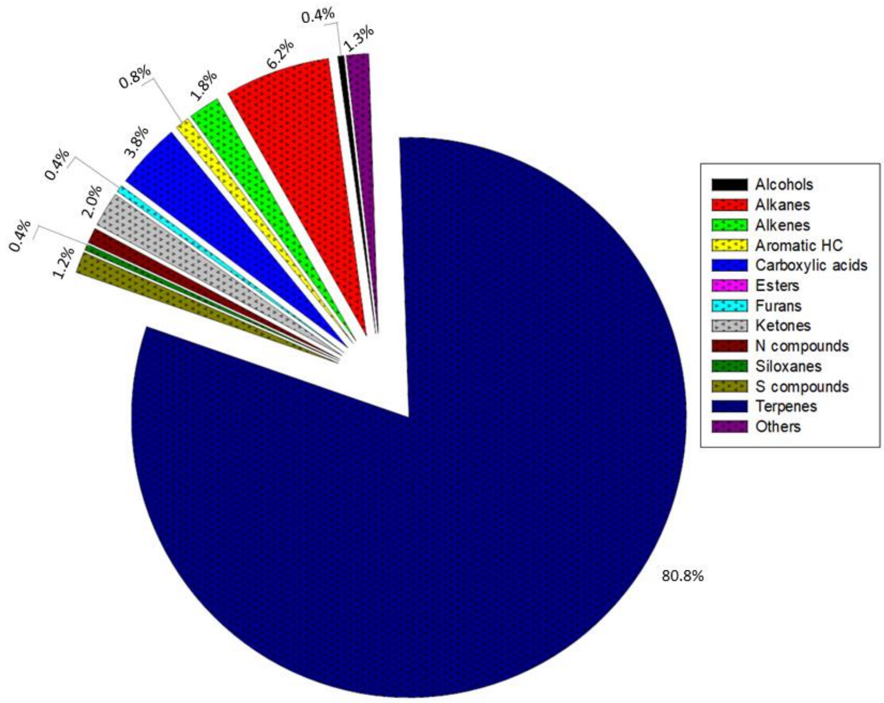


Figure 5-10. tVOC characterization during biodrying of cellulosic sludge implementing strategy 1 on emission peak day.

Considering the polluting gaseous emissions, NH_3 and tVOC emissions are recognised, in particular, to be key weaknesses of the biodrying process. In general terms, considering the results obtained in this section, CS and PPS have been demonstrated to have attractive characteristics for their treatment through biodrying, not only form a technical performance point of view, but from environmental point of view as well.

5.5 TECHNO-ECONOMIC ASSESSMENT OF BIODRYING PROCESS

Apart from the technical feasibility and environmental assessment, a techno-economic analysis is required for a complete comprehensive assessment of any technological solution. In this context, the development of a thorough economic model through Life Cycle Costing methodology will provide reliable claims for stakeholders that might lead to, eventually, attract investment for technological transference and scale-up, as in fact, has already happened. For the economic assessment, cash inflows and outflows are considered. Thus, cash outflows or costs are given into three categories based on life cycle phases: (a) project initiation and construction, (b) operation and maintenance, and (c) end-life costs. Project initiation and construction costs include capital expenditures (CAPEX) for treatment infrastructure and equipment, including construction costs and other external costs associated with construction. Operation and maintenance costs (OPEX) include consumption of chemicals and other consumables, maintenance of equipments, licensing fees, administration, and training and labor requirements. Clearly, energy requirements and waste disposal, if any, represent a significant operational cost. End- life costs are the costs associated with the decommissioning of the installation after the completion of the assumed end-life of the infrastructure. Cash inflows to consider are derived from high added-value products selling, in this case, biomass fuel (BMF), and the avoided sludge disposal costs.

Bearing this objective in mind, experimental data obtained were used to define and build an economic model to evaluate the performance and economics in a hypothetic plant in which biodrying would be implemented to produce the BMF from cellulosic sludge. Two case scenarios were assessed through the same model. In the first scenario, an isolated biodrying plant with a treatment capacity equivalent to the annual cellulosic sludge production rate of WWTP of Geestmerambacht was modelled and break-point was calculated by applying the NPV=0 approach, obtaining the minimum population equivalent served in the WWTP able to achieve a positive economic scenario (Imeni et al., 2019). After that, an expansion of the system was assessed where biodrying technology was coupled with the application of Cellvation® fine sieve in the first stage of wastewater treatment followed by the subsequent conventional wastewater treatment steps already implemented. For the second scenario considered the implementation of biodrying to treat the primary sludge produced in the existing Almendralejo WWTP. To do so, economic model was adapted to the experimental results obtained and afterwards break-even point was calculated for this scenario.

5.5.1 SYSTEM BOUNDARIES AND SCENARIO DESCRIPTION

The system in study includes an input of cellulosic sludge and bulking agent. In the first scenario, the WWTP of Geestmerambacht, which serves to 262,000 PE and treats yearly 15,614,342 m^3, was modelled. Thus, WWTP Geestmerambacht is a large scale WWTP that produces yearly more than 8000 metric tons of dewatered sludge. The first approach only considered the implementation of the biodrying technology while afterwards the expansion of the system including also the implementation of Cellvation® fine sieve technology substituting conventional primary settler treatment in the whole WWTP train. Figure 5-11a describes the boundaries of the system modelled. Grey area refers to the first approach, considering only the biodrying technology implementation, while the orange area refers to the expansion of the system, considering the WWTP as a whole system. In the second scenario, considering the still low implementation level of Cellvation-like fine sieves as primary wastewater treatment, biodrying technology was proposed to treat the most equivalent conventional municipal sludge, namely primary sludge. Therefore, in line with the experimental trials assessed, WWTP of Almendralejo was modelled. Almendralejo serves 41,888 PE and it was estimated to produce yearly 3440 tonnes of primary sludge. Figure 5-11b describes the boundaries of the system modelled to valorise primary sludge. Product selling was assessed by estimating market price of the sieved end-product obtained, according to its estimated calorific value and following the conventional biomass market price in Spain, which is given per energy provided (Avebiom, 2019). Product selling was selected as main parameter to determine cash inflows from biodried product commercialisation instead of considering energy recovery in order to facilitate and homogenize biodrying and composting models.

Specifically, in the first scenario, according to the Salsnes finescreen filter implemented the WWTP in study (Geestmerambacht, The Netherlands), 1.2E-04 metric tonnes of cellulose can be recovered per each cubic meter of wastewater treated. 17% of recovered cellulose was assumed not to meet the minimum quality parameters from and it is therefore rejected. Accordingly, this last estimated amount of rejected CS was used for the LCC calculation. In the expanded scenario, apart from the valorisation of rejected CS, recovered cellulose was sold as secondary raw material. The detailed description of the expanded scenario can be consulted in deliverable D4.5 entitled "Socio-economic assessment including life cycle costing (LCC) and cost benefit analysis (CBA) reports" (SMART Plant G.A 690323, 2020). In the second scenario, all the primary sludge produced was assumed to be biodried.

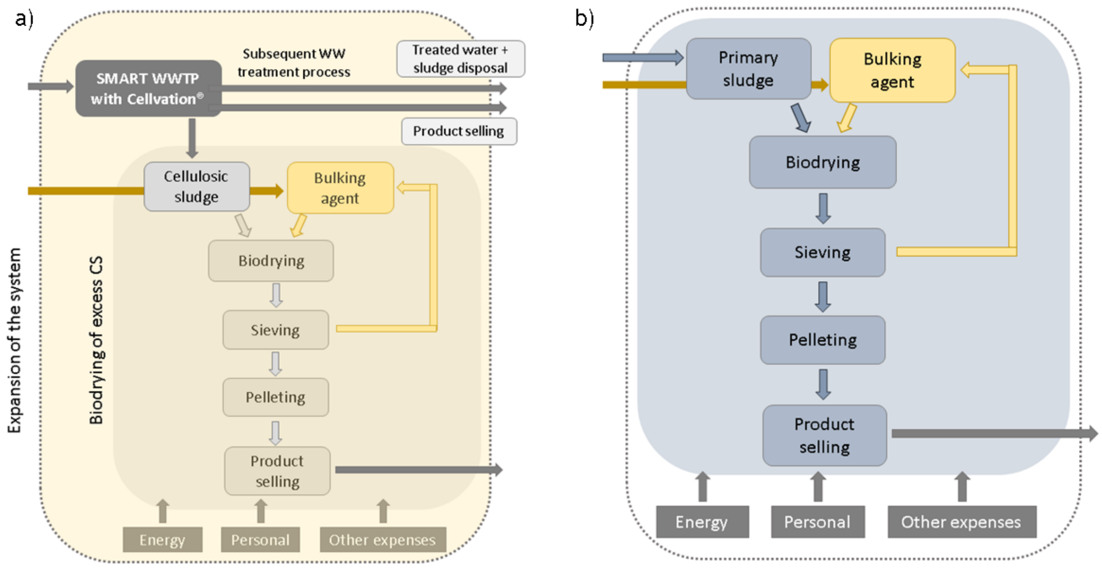


Figure 5-11. Biodrying system boundaries considered for LCC models considering a) biodrying technology to valorise CS separately marked in grey area and the expansion of the system including Cellvation® technology implemented in the first stage of wastewater treatment or b) biodrying technology to valorise primary sludge in an existing WWTP.

Main physic-chemical characteristics of cellulosic sludge and primary sludge assumed in this section are shown in Appendix III. Similarly, biodrying process performance assumptions were done according to experimentally obtained data in each case and then used for the estimated calculation of mass balances and economic parameters at full scale.

Figures 5-12a and b show the mass balances achieved in each base case.

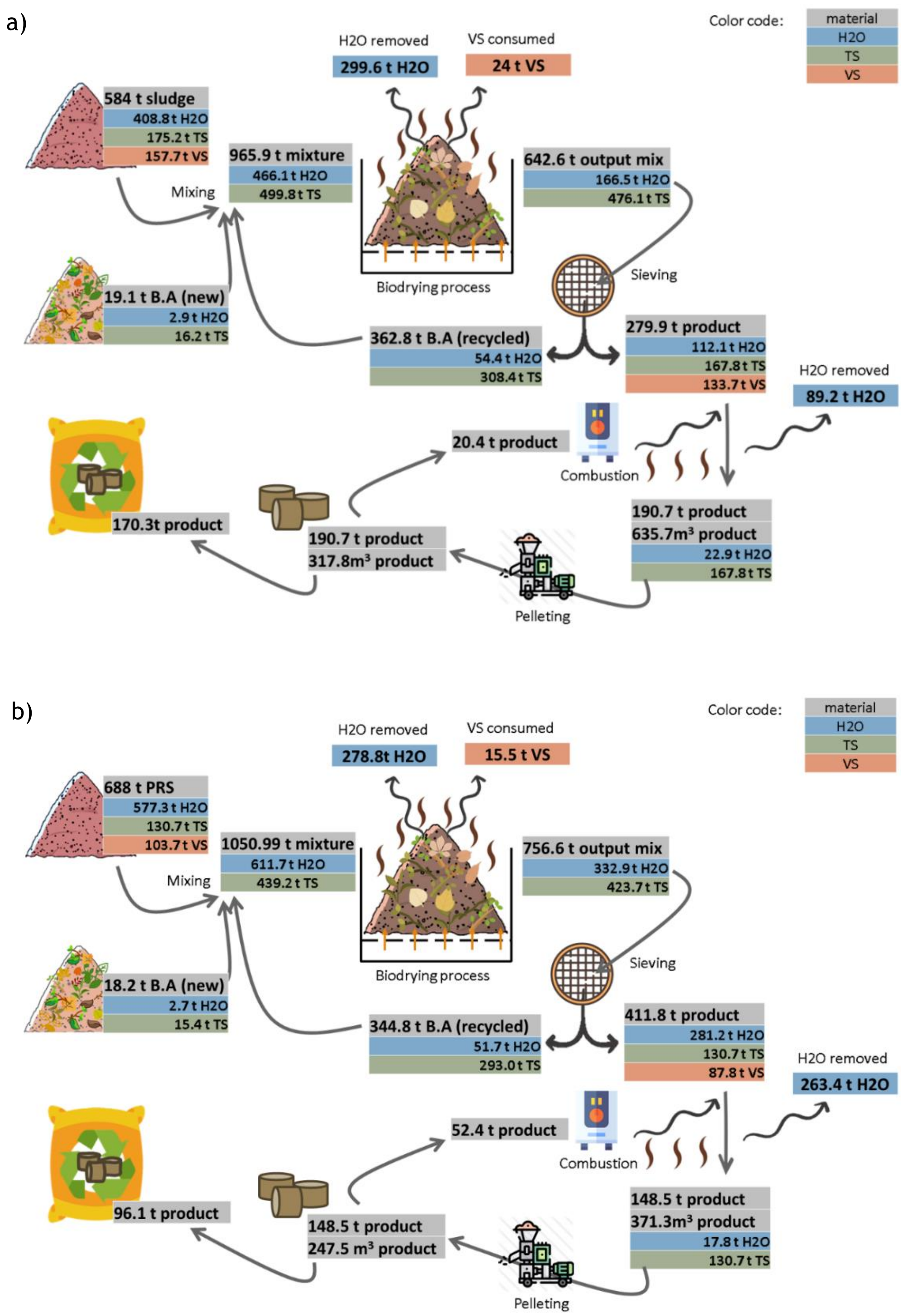


Figure 5-12. Detailed mass balance of BMF production through biodrying process of a) CS for the base case WWTP and b) PSR for the model adjusted for conventionally existing WWTPs.

Given the industrial scale nature of the simulated scenario, higher thermal inertia, longer thermophilic stage and enhanced moisture removal can be expected. Accordingly, full-scale estimated results, present equivalent values to those determined at pilot scale in Section 5.3 (Winkler et al., 2013), since it is assumed that at full scale, at least, these results could be indubitably obtained.

Also, since pelleting was considered as post-treatment, calories required to remove excess moisture from the output product mass were calculated and assumed to be removed by the combustion of part of the pellets produced in a biomass boiler. Products obtained were considered to achieve a biomass boiler performance equivalent to pellets produced from forestry wastes or simply wood, thus they were assumed to achieve a self-sustaining process making them suitable for the European market.

The MC of the final sieved product in the CS scenario was 40.1%. In order to reduce the moisture content of the product to the targeted 12% to make it suitable to pelletise, excess water was estimated to be removed by the combustion of the 10% of the pellets produced, being the rest available for their selling. In terms of final product, pelleting reduced the volume of product to almost the half of the volume, multiplying almost by two the energy transport capacity of each truck.

5.5.2 FIRST CASE SCENARIO: TECHNO-ECONOMIC ANALYSIS OF BMF PRODUCTION THROUGH BIODRYING TECHNOLOGY USING CELLULOSIC SLUDGE AS FEEDSTOCK

Estimation of CAPEX and OPEX costs were done according to real budgets from real engineering providers and following the orientation given by real WWTP consulted (WWTP of Manresa). According to the infrastructure needs modelled, overall CAPEX of 59.845 € was estimated, while yearly OPEX costs were estimated to be 29.232 €. Overall CAPEX, OPEX, benefits and parameters for the economic evaluation of the hypothesised biodrying plant are given in Table 5-14.

According to the economic model, main investment costs were associated to the biomass boiler required to adjust the moisture content of the product to pelletise it, accounting for 37% of overall CAPEX. Civil works and the fixed cost of engineering services resulted to be also significant investment cost reaching respectively to 28% and 18% of overall CAPEX.

Considering the OPEX results obtained from LCC model, principal costs are derived from pelleting costs (40%) followed by mixing truck rental (21%). Although, the amount of cellulosic sludge input is clearly not high enough to require the hiring of an operator for the management of the plant, a partially working operator was assumed for those costs. Thus, considering the sludge input, a partially working operator was estimated. That operator would be hired by the WWTP and would work 4 hours weekly in the management of biodrying plant, accounting for 15% of yearly operational costs.

Table 5-14. Main economic parameters obtained from base case cellulosic sludge biodrying LCC model.

Parameter	Value
Lifespan (y)	25
CAPEX (€)	59,845 €
OPEX (€/y)	29,232 €
Incomes (€/y)	27,838 €
Benefits (€/y)	-1,394 €
NPV	-92,418 €
IRR (3 years)	n.a
IRR (6 years)	n.a

n.a refers to not available data

Energy costs, mainly from electricity used for aeration but also from diesel consumption for turning the piles, accounted for almost 8% of the overall OPEX (estimated results are detailed in Appendix VI). The overall estimated energy consumption along the year would be 9,452 kWh, corresponding to 16.2 kwh per metric ton of CS treated. Comparatively, full-scale conventional sludge drying technologies, such as convective and conductive drying consume significantly more energy (between 700 and 1,400kWh t^{-1}sludge) (Bennamoun et al., 2013). Furthermore, considering the energy consumption and production potential from the BMF produced, a considerable yearly net energy production of more than 633 MWh could be achieved. The rest of the categories considered accounted for the 5-7% of the overall yearly OPEX.

Incomes were estimated from the biodried product amount obtained. First, the energy required to remove the excess moisture of the product was estimated and then the price of this energy was calculated comparatively from two different sources: use of part of the pellets produced and use of natural gas. Accordingly, 11% of the pellets produced would be needed to be burned that would lead to a scenario 77% cheaper than using natural gas. Therefore, 16,157 € would be yearly obtained from selling the rest of the pellets produced. The fair market price estimated for the product would be 94.3 € t^{-1} of product, which was the triple of the price paid for refuse derived fuel (Fernández-González et al., 2017). Besides, according to Avebiom, the Spanish association of the energetic valorisation of the biomass, the biodried product pellet price was estimated to be 36-40% lower than wood pellets while the heating value would be only 20% lower (Avebiom, 2021). Avoided costs from sludge disposal were also considered for the calculation of benefits for each scenario, assuming a sludge disposal cost of 20 € per each metric ton of sludge external management avoided (value provided by the WWTP consulted, Manresa WWTP), which was in the low range of what was reported in the literature (Piao et al., 2016; Capodaglio & Olsson, 2020). It is important to highlight that the value considered for disposal cost did not include transportation cost, which would increase more the yearly avoided costs.

As a result, incomes will be lower than OPEX producing consequently, yearly 1,394 €of economic expenses. Taking into account the economic scenario estimated, recoup of initial investment would not be possible in the lifespan considered (25 years), achieving a NPV of -92,418 €.

Once the base case was analysed, a break-even analysis was carried out to find out which would be the minimum plant capacity (PC) leading to an economically feasible scenario. To do so, the zero-profit scenario was calculated by applying NPV=0 principle. (Abdallah et al., 2018; Imeni et al., 2019).

To this aim, a variable cellulosic sludge input scenario was designed according to the population size served by the WWTP.

Same economic parameters as before were estimated for each of them. Distribution of the estimated CAPEX and OPEX categories are shown in Figure 5-13.

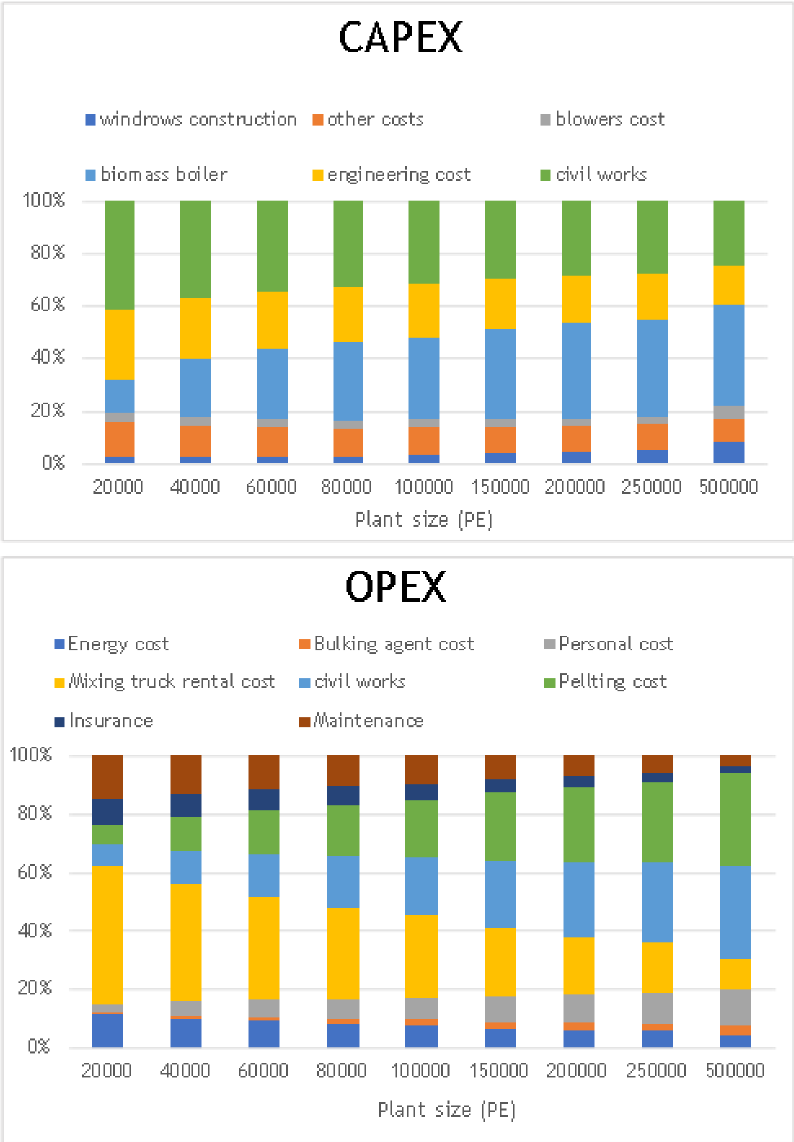


Figure 5-13. Distribution of CAPEX and OPEX categories according to the plant sizes studied.

Civil works were the main cost associated to the smallest scale plants while the percentage contributing to the overall CAPEX were gradually reduced together with plant size. Conversely, biomass boiler cost accounted for more than 35% of the overall CAPEX costs in plant sizes larger than 150,000 PE. Regarding OPEX, main yearly OPEX were associated to mixing truck rental, accounting for more than the 40% of the OPEX in plants smaller than 60,000 PE and being gradually reduced in larger plants. Personal costs were also a significant operational cost although in this case the percentage of its contribution correspondingly increased with plant size.

Figure 5-14 shows the regression curves of CAPEX, OPEX and benefits according to the plant size modelled. CAPEX followed a hyperbolic trend roughly stabilising in the WWTPs with maximum treatment capacity. OPEX and benefits fitted a linear regression model.

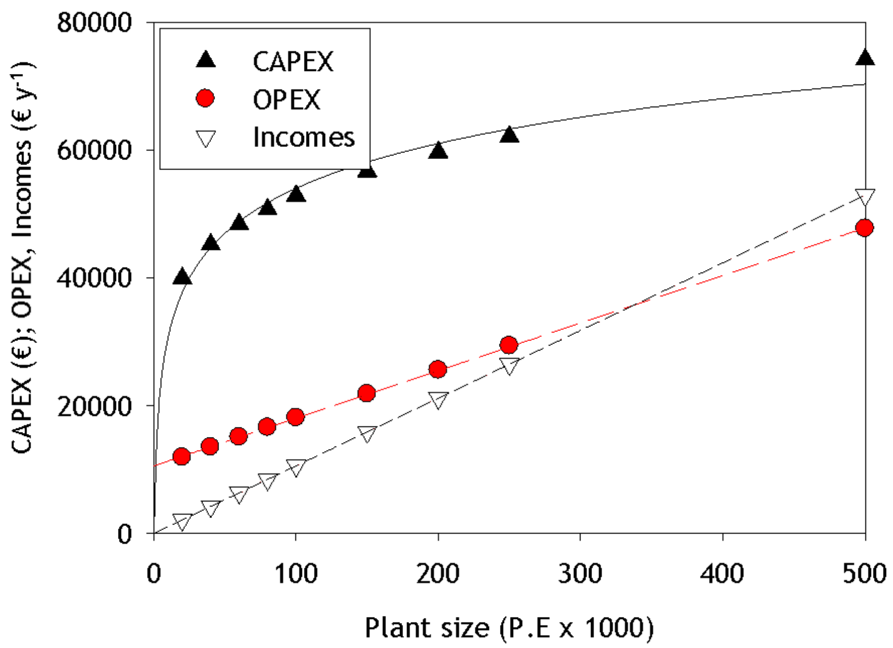


Figure 5-14. WWTP treatment capacity dependent economic parameters: black triangles indicate CAPEX, white triangles indicate OPEX and red circles indicate cash incomes.

By definition, NPV is calculated as the difference between the overall benefits of a certain period of time (t) and the initial investment costs, considering a certain discount rate (r) which reflects the decline in value of costs in that certain period of time. Figure 5-15 shows the estimated NPV values dependent on the plant capacity (PC) which fitted a linear model.

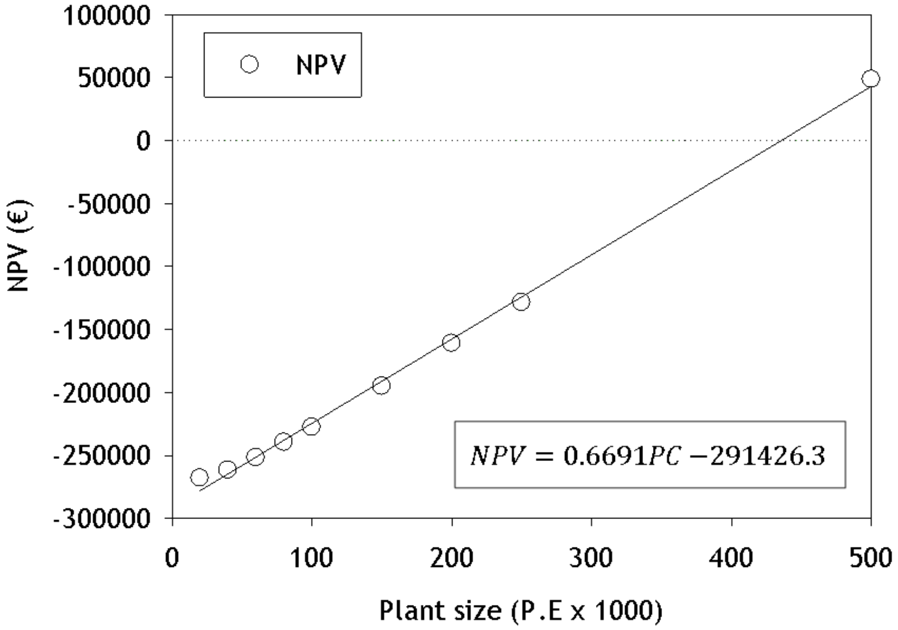


Figure 5-15. WWTP treatment capacity dependent NPV.

Assuming NPV equal to zero the minimum plant size capacity for an economically feasible biodrying scenario was calculated: 435,550 PE. Although a detailed economic model was built, reliability of predictions is limited and affected by a number of external factors. First, from cash inflow point of view, policy constrains might arise for biomass fuel recovered from waste, in this case, sludge, as they are not specifically considered in the European renewable energy strategy. Moreover, thoroughly analysing the current market for this kind of product would be critical for a more realistic economic assessment. Given the current priorities of Europe considering green deals fundamental pillars of the European economy, including the promotion of renewable energies, a well-developed waste derived fuel market could be expected for the coming years. To this aim, though, specific policies and regulations should be defined including the minimum characteristics and quality of such renewable energy sources derived from waste. Hence, the validation and promotion of this type of bio-fuels seems to greatly depend on the efforts that European Commission is making in this context. Second, the increasing price of sludge management could be the key to guarantee the economic viability of the project. The gradual banning of sludge landfilling and increased limitation for agricultural application of certain sludges might increase its associated management costs and, in such situation, biodrying could rise as a feasible alternative, either technically, environmentally and economically.

The scenarios studied in this section assume availability of CS, thus an additional expanded scenario was included and assessed in which biodrying would be coupled with Cellvation® system, the technological solution producing CS. To assess the entire range of the conventionally found plant sizes, three different plant treatment capacities were studied, namely, small, medium and big. LCC results were estimated for the technological solutions proposed (Cellvation® alone and Cellvation® coupled with biodrying) and compared upon the base case LCC of WWTP. The results described in this section are available in Deliverable 4.5 of SMART-Plant entitled "Socio-economic assessment, incl. LCC and CBA reports" (SMART Plant G.A 690323, 2020).

The implementation of SMART Plant solutions, either Cellvation® technology individually or coupled with biodrying would allow the increase of investment costs around 10%, while OPEX would be reduced around 24% in the lifespan considered (Table 5-15). Overall, the solution proposed would allow savings of 19-22% compared to a conventional WWTP achieving the most beneficial scenario in the biggest capacity plant size.

Table 5-15. Comparison of economic parameters of base case WWTP, WWTP with Cellvation® technology and the same WWTP with coupled biodrying technology implementation for the valorisation of excess CS.

Scenario descriptor	small	medium	big
PE	50,000	100,000	250,000
YEARS	25	25	25
End of life costs (€/PE)	30	25	23
Base CAPEX – TOTAL (€/PE)	300	250	230
Delta CAPEX w Cellvation® (€/PE)	330	275	253
Delta CAPEX w Cellvation® + biodrying(€/PE)	330.3	275.1	253.1

Base OPEX - TOTAL (€/PE/y)	450	400	400
Delta OPEX w Cellvation® SMARTech (€/PE)	337.5	300	300
Delta OPEX w Cellvation® SMARTech + biodrying (€/PE)	341.6	301.6	298.8
Cellvation® SMARTech Benefits (€/PE)	65	65	65
Cellvation® SMARTech + biodrying Benefits (€/PE)	66.9	66.7	66.7
LCC Results w/n Cellvation® SMARTech	780	675	653
LCC Results w Cellvation ® SMARTech	633	535	511
LCC Results w Cellvation® SMARTech coupled with biodrying	635	535	508
SAVINGS w Cellvation® SMARTech	18.9%	20.7%	21.7%
SAVINGS w Cellvation® SMARTech coupled with biodrying	18.6%	20.7%	22.2%

5.5.3 SECOND CASE SCENARIO: TECHNO-ECONOMIC ANALYSIS OF BMF PRODUCTION THROUGH BIODRYING TECHNOLOGY USING PRIMARY SLUDGE AS FEEDSTOCK

Bearing in mind that although the implementation of fine sieves is gradually increasing, their application is still not widespread. Consequently, finding an appropriate feedstock in the conventional existing WWTPs would increase the impact of the solution proposed in this chapter, opening a window to the full-scale application of the biodrying technology. However, full-scale implementation of a technological solution depends highly on its economic feasibility. Thus, economic model developed was adapted to the conventional sludge with highest potential, primary sludge. The WWTP of reference in this case was WWTP of Almendralejo (Badajoz, Spain) which serves to 41,888 PE. According to the estimations done from the yearly generation data provided by the WWTP, Almendralejo would produce annually 2872 m^3 of dewatered PRS. Main physic-chemical characteristics of PRS and operational parameters considered in the economic model are based on the experimental results obtained and they are detailed in Annex III.

The model estimated two biodrying windrows of 262m^3 each. This scenario would be associated to a CAPEX of 129,753 € and yearly OPEX of 89,059 €. From the categories studied in CAPEX, windrows construction and biomass boiler are the most remarkable (23% and 31% of overall CAPEX, respectively). Pelleting costs (44% of overall OPEX), personal costs (26% of OPEX) and energy cost (13% of overall OPEX) are the most remarkable categories in OPEX. The selling price of a sieved product with a 12% of MC and a LHV of 15.6 MJ kg^{-1} would be 108.1€ per metric tonne and its selling would be linked to yearly 39,455 € in incomes. That scenario would lead yearly 2,665 € of overall benefits and leading to a NPV of -67,490 € in the lifespan considered (25 years), thus recovery of the investment done would not be possible in this scenario. Again, plant size dependent model was constructed to estimate the minimum plant size for an economically feasible scenario. Thus, CAPEX, OPEX and Incomes were calculated according to the variable sludge inflow in each plant size. Figure 5-16 shows the distribution of CAPEX and OPEX in the categories considered.

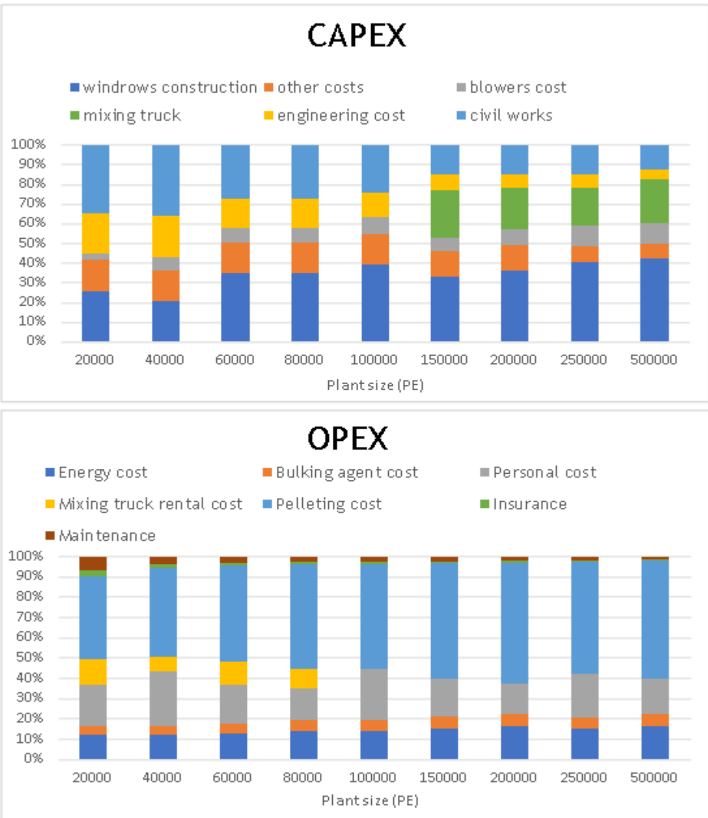


Figure 5-16. Distribution of CAPEX and OPEX categories according to the plant sizes studied.

Biomass boiler purchase, civil works and windrows construction were the main CAPEX categories in the smallest scale plants below 100,000 PE, while the costs strictly associated to infrastructure costs such as windrows construction and other costs for auxiliary materials were the main cost in plants with higher size. Regarding OPEX, pelleting costs were always the main operation associated cost accounting at least for 40% of overall OPEX costs. Personal costs were also significant in every case study assessed (14-27% of overall OPEX) while energy costs accounted always for 12-15% of the OPEX regardless of plant size.

Figure 5-17 and Figure 5-18 show the regression fits of CAPEX, OPEX and Incomes economic categories and NPV regression fit, respectively.

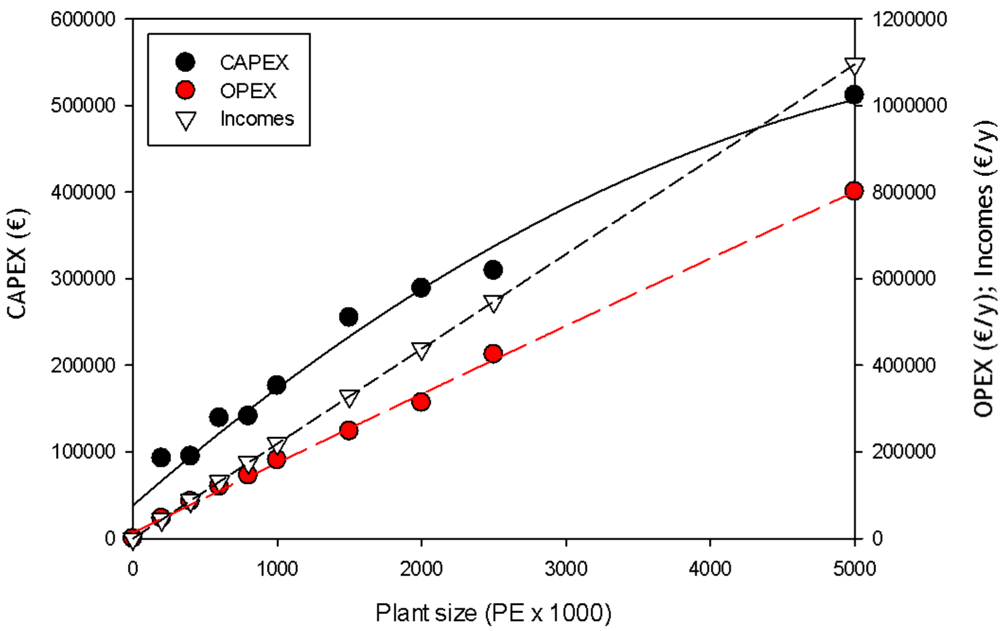


Figure 5-17. Plant size dependent economic parameters for a biodrying plant to valorise primary sludge: black triangles indicate CAPEX, white triangles indicate OPEX and red circles indicate cash incomes.

According to the regression calculated, the minimum plant size for an economically feasible biodrying plant to valorise PRS would be of 40,066 PE.

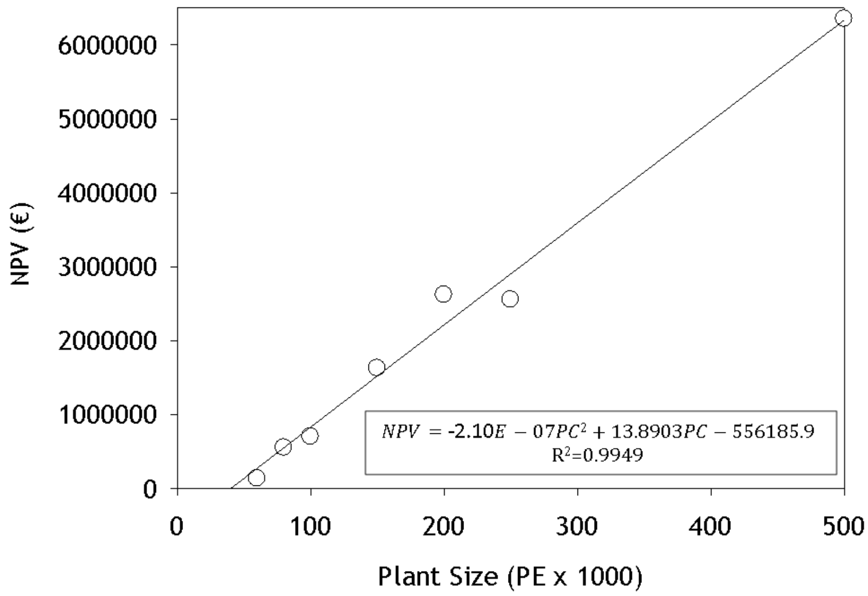


Figure 5-18. Plant size dependent NPV regression fit

5.6 MAIN STRENGHTS AND WEAKNESSES OF THE BIODRYING OF CS OR PRS AS A SOLUTION TO RECOVER ENERGY

As the final outcome of the research conducted and presented along this chapter, this section summarises the main strengths (S), weaknesses (W), opportunities (O) and threats (T) that the solution proposed presents (Figure 5-19).

Sludges, when efficiently managed, show potential as energy source. Biodrying technology can be regarded as an option for the mentioned management, obtaining as a result a sustainable and renewable energy source, contributing to the European Circular Economy Action Plan and the strategies launched in the framework of climate action. This solution would promote particularly:(i) recovering energy from sludge, a renewable and carbon neutral source, (ii) promote the energy and heat self- sufficiency of Europe opening a window to move away from imported natural resources, (iii) the zero-waste approach and end-of waste status for sludge , (iv) relies on locally produced feedstock and can be implemented on site, avoiding the transportation of the waste and (v) lower carbon footprint than fuel-based energy sources, and than conventional thermal sludge treatment processes.

Figure 5-19. SWOT analysis of the biodrying technology applied to CS and PRS.

Additionally, and although it was out of the scope of the present thesis, removing cellulosic material as a first step of WWTP might have benefits for the subsequent wastewater treatment stages in terms of efficiency or treatment costs, as it was stated in previous literature. Nevertheless, the weaknesses that this solution presents and should be addressed are the (i) the variability of feedstocks and particularly initial moisture content that can substantially affect the efficiency of the process and (ii) the requirement of post- treatment steps, being pelleting step the most remarkable. The first difficulty can be addressed by optimising first, the performance of Cellvation® technology or dewatering unit in the case of primary sludge and second, the sludge to bulking agent mixing ratio. Also, recirculating part of the dried product to decrease the moisture content of initial mixture can contribute to obtaining a product with higher quality. For the second difficulty and considering the current trend towards bioenergy, pelleting technologies will probably be more affordable, both in terms of investment cost and cost of operation.

In contrast, the threats that this technology faces are mainly related to standardisation issues and lack of administrative support to effectively access to the European market. Consequently, in the current context, the only available market niche in which BMF would fit would be in cement kilns. Additionally, market accessibility of BMF is limited due to the competitive prices of the conventional wood-based biomass.

Therefore, assuming the overcoming of the abovementioned limitations, this technological solution would help aligning the water sector with the European economy action plan by boosting the end-of waste status of sewage sludge, reducing the dependency on non-renewable energy sources and creating good quality jobs for European citizens. Besides, in terms of potential business-model, the alternative proposed here can be taken as commercial strategy as it presents good branding opportunities.

6. ASSESSMENT OF P RICH BIO-BASED FERTILISER PRODUCTION FROM SLUDGE THROUGH ADVANCED COMPOSTING PROCESSES

6. ASSESSMENT OF P-RICH BIO-BASED FERTILISER PRODUCTION FROM SLUDGE TRHOUGH ADVANCED COMPOSTING

Composting process of sewage sludges is extensively implemented technology allowing the sanitisation and stabilisation of such feedstocks and obtaining an organic amendment. However, the process presents some weaknesses that were mentioned in Chapter 1. Such weaknesses are related either with the technology performance or either with product quality, which are intended to be overcome through advanced composting. First, energy and land requirements are the main operative requirements which are usually key contributors of production costs of composts (Kulikowska and Gusiatin, 2015). Second, composts from conventional sewage sludges usually present low concentrations of nutrients compared to mineral fertilisers (Kominko et al., 2019; Rehman et al., 2020), even though they are claimed to improve physic-chemical and microbiological properties of soils (Sánchez et al., 2017; Grigatti et al., 2019). Third, the overall acceptance of composts produced from sludge might be rather limited due to potential presence of toxic elements such as heavy metals or pollutants of emerging concern. Consequently, compost in general, and compost from sludge in particular, present a low market price (Alibardi et al., 2020). Finally, due to the low nutrient content and linked to the limitations described above, the exportation and transportation costs of sludge-based compost cannot be supported for medium or long distances.

With all these considerations, the experimentation of advanced composting carried out in this chapter is based on: i) choosing the appropriate feedstock to end up with a nutrient rich bio-based fertiliser, and ii) improving the aeration system to reduce operational costs and improve the environmental and economic performance of the process. More specifically, the enhanced nutrients content of SCENA sludge, valorised through a dynamic composting process with an advanced oxygen demand (OUR based) dependent aeration system, presents the potential to overcome all the aforementioned limitations. On one hand, due to its nature, this sludge presents attractive characteristics in terms of high content of nutrients, particularly phosphorus, and organic matter content as well as low content of inorganic toxic pollutants. Given its specific characteristics this sludge will be referred to as P rich sludge (PS) from now on. This type of feedstock has the potential to be converted into a high quality and safe new-generation fertilising product (the so-called BBF) that presents potential to lead to higher benefits than conventional compost from sludge. On the other hand, the OUR based composting process presents the potential to maximise biological degradation of organic matter, providing the strictly required aeration while shortening the operational period and therefore reducing the production costs (Puyuelo et al., 2010). Finally, the intense biodegradation process during composting is probably able to reduce the content of emerging organic pollutants in the products obtained, as it was demonstrated for conventional composting processes (Ezzariai et al., 2018; Biel- Maeso et al., 2019; Zhang et al, 2019). Overcoming all these barriers would demonstrate that certain sludge derived products could be accepted in current regulation on the market of EU fertilising products (EC 2019/1009), despite their current ban as feedstocks in the mentioned regulation. Furthermore, given its high content in nutrients and organic matter, the bio-based fertiliser (BBF) obtained from the solution suggested could potentially contribute to nutrient recycling and promote the restoration of degraded soils due to intensive agriculture. BBF could partially substitute the fertilising products available in the market focusing, specifically, in those containing phosphorus from mineral origin which are majorly imported to Europe. Thus, BBF concept fits perfectly in the concept of new-generation sustainable fertilising products and overall, this solution would contribute to promote the circularity and safety of European agriculture, food systems and water sectors, in line with the current European bioeconomy strategy.

Considering the abovementioned limitations, this chapter aims to demonstrate the feasibility of producing bio-based fertilisers compliant with current market and regulatory demands from selected feedstocks. Additionally, the solution proposed seek to enhance both environmental and economic feasibility of composting process by lowering polluting gaseous emissions and increasing its economic competitiveness. To do so, nutrients content, stability and concentration of potential hazardous contaminants were measured in the BBF. Besides, a pot-based bioassay was performed to assess its agronomic quality. Moreover, and to get a comprehensive picture of the solution proposed here, most critical polluting gaseous emissions were monitored during advanced composting process and an economic model of the full-scale-adapted advanced composting process was developed to assess two potential and relevant case studies. Finally, this chapter concludes by describing the main strengths and weaknesses of the advanced composting of PS, highlighting the opportunities that are built from the solution proposed and identifying the challenges that should be tackled to bring both the technology and BBFs obtained to the European market.

Considering the significance of the results described in this chapter in the framework of the European circular economy and how the solution proposed here meets the current farm-to fork strategy contributing to the end of waste status of sewage sludges, we aim to make a relevant contribution to the current state of the art of new-generation fertilising products from the findings of this chapter. The scientific paper is currently under preparation and it is expected to be submitted in the first quarter of the 2022.

6.1 PHOSPHORUS RICH SLUDGE CHARACTERISTICS AND SET UP PARAMETERS FOR COMPOSTING PROCESSES

A set of composting trials of P rich sludge (PS) was carried out, performing in all the cases an active degradation stage of 15-20 days followed by a curing stage of 50-70 days. The active degradation stage was performed using the pilot plant described in Chapter 4 and the OUR-based control algorithm for the air supply (Puyuelo et al., 2010). During curing stage, aeration consisted on turning of the bulk mixture once a week. Raw PS characteristics are summarised in Table 6-1.

In terms of nutrients content, PS presented rather high nitrogen and phosphorus contents compared to conventional dewatered sludges: 3-5% (in dry basis) for nitrogen (Grigatti et al., 2019; Laura et al., 2020) and 1-2.3% (d.b.) for phosphorus (Tyagi et al., 2013; Tarayre et al., 2016).

It is worth mentioning that the improvement of SCENA performance during the project seemed to have led to increased phosphorus contents in PS (up to 4% d.b.). Regarding K content of PS, the values determined were in line with literature (Laura et al., 2020). Concerning biological parameters, PS showed moderate biological activity with a DRI_{24} below 5 gO2 $kg^{-1}VS\ h^{-1}$ (Barrena et al., 2011).

Table 6-1. Range of physic-chemical characteristics of PS assessed through advanced composting.

Parameter	DM	VS	TKN	C/N	Phosphorous	Potassium	pH	EC	DRI_{24h}	AT_4
Unit	%, w.b.	%, d.b.	%, d.b.		%TP, d.b.	%TK, d.b.		mSc m^{-1}	g O2kg^{-1} VS h^{-1}	gO2kg^{-1} VS
PS	17.1 - 18.7	72.0-74.9	6.9-8.1	5.0-5.7	3.0 - 4.2	0.3- 0.6	8.0-8.3	0.7-0.9	2.5- 4.5	157-243
INITIAL MIXTURE	31-40	83-85		12.9-19.9						

w.b. refers to wet basis and d.b.refers to dry basis

Concerning heavy metals, PS was considered to be of excellent quality as it is shown in Table 6-2 which indicates the content of heavy metals in PS and relevant regulatory limits for agricultural use of sludge in Europe.

Table 6-2. Heavy metals content in one sample of PS and relevant European regulatory limits for its application into soil.

Metal	PS	[1]	[2]
Mg (mg kg $^{-1}$TS)	5390 ± 309		
Cr (mg kg $^{-1}$TS)	8.1 ± 0.3		600-800
Ni (mg kg $^{-1}$TS)	6.9 ± 0.4	300-400	100-200
Cu (mg kg $^{-1}$TS)	54 ± 5	1000-1750	600-800
Zn (mg kg $^{-1}$TS)	226 ± 18	2500-4000	1500-2000
Cd (mg kg $^{-1}$TS)	0.24 ± 0.02	20-40	2-5
Pb (mg kg $^{-1}$TS)	9.5 ± 0.7	750-1200	200-500
Hg (mg kg $^{-1}$TS)	0.44 ± 0.09	16-25	2-5

[1] Directive 86/278/EEC; [2] Working document on Sludge Directive, 3rd. draft, prevision for 2015 and 2025 (European Commission, 2000)

The content of heavy metals in PS were far below the maximum limits established in the current sludge directive 86/278/EEC (European Commission, 1986) and also below the values established in the more stringent last version of the working document of the forthcoming revised sludge directive (European Commission, 2010). In fact, heavy metals contents were in compliance with the most stringent worldwide regulatory limits found such as Austria, The Netherlands, Denmark, Canada or Japan (Colón et al., 2017; Collivignarelli et al., 2021).

In all composting processes, pruning waste was used as bulking agent to achieve values of initial moisture close to 60%, known to be optimum for the process (Haug, 1993). Therefore, sludge to pruning mixing ratio was set to be around 1:3, having in every batch at least 25 kg of PS. Figure 6-1 shows an example of the PS treated through advanced composting (a) and. the initial mixture prepared (b).

Figure 6-1. PS treated through advanced composting (a) and mixture of PS and pruning waste (b).

6.2 PERFORMANCE OF THE ADVANCED COMPOSTING OF PHOSPHORUS RICH SLUDGE

Regarding composting performance, both active degradation phase and curing phase were satisfactorily carried out in all the cases. Among the monitored and controlled parameters during composting processes of PS, temperature in the middle of the bulk mixture, airflow and oxygen consumption are the main ones used for the assessment of the composting performance. Figure 6-2. shows a representative profile of active degradation phase in advanced composting. Different trials (6) were performed to have a complete assessment of advanced composting process and to demonstrate the reproducibility of the process.

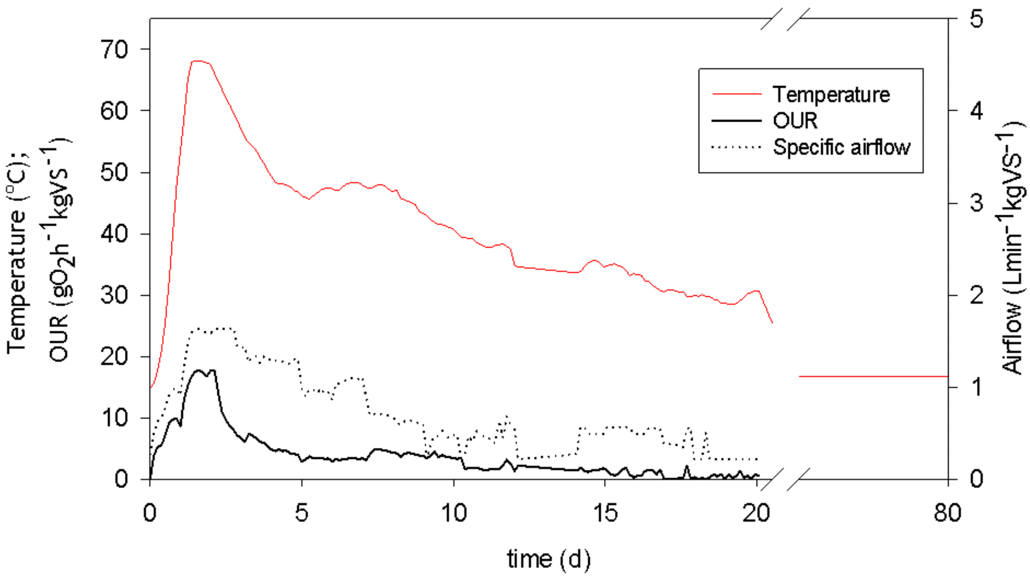


Figure 6-2. An example of advanced composting performance of PS during active degradation phase: evolution of bulk temperature (red line), OUR (black line) and specific airflow rate supplied (discontinuous line).

In average terms, composting start-up phase occurred successfully, arriving to a maximum temperature above 70ºC after approximately 24h. In all the cases, thermophilic phase lasted between 5 and 7 days, followed by mesophilic temperature ranges until the end of the processes. After active degradation phase, curing stage of the bio-based fertiliser was carried out. During the curing stage temperature of bulk mixture was monitored once per week whereas biological stability was monitored each month. Thus, in total, trials lasted between 70 and 90 days

Energy consumption was uniquely due to the air supply, since energy consumption of monitoring system was negligible. As expected, composting consumed less energy than biodrying using the same system and reactors, mainly due to a more intensive aeration strategy in biodrying processes. Specific energy consumption during trials was in the range of 0.12- 0.17 kWh kg^{-1}VS (Table 6-3), in line with what was expected from OUR based system according to previous works, which was claimed to be energetically efficient (Puyuelo et al., 2010) and lower than conventional aeration strategies (Puyuelo et al., 2014; Han et al., 2018a; Rincón et al., 2019; Wang et al., 2019).

As described in Chapter 3, the RIE-Energy Consumption Index or simply RIE-EC, is suggested as an appropriate composting performance assessment index which relates overall energy consumption with stability of the material achieved during the composting process (Colón et al., 2012; Puyuelo et al., 2014).

Table 6-3. Energy consumption and energetic efficiency results of advanced composting trials of PS.

Energy consumption	Specific energy consumption		RIE-EC
kWh	kWh kg^{-1} PS	kWh kg^{-1}VS-PS	kWh ($gO_2 kg^{-1}$VS-PS $h^{-1})^{-1}$
2.09- 4.78	0.14-0.17	0.97-1.31	0.85-1.91

The RIE-EC results obtained in the present study were highly variable (0.85- 1.91 kWh (gO_2 kg^{-1}VS-PS$h^{-1})^{-1}$) mainly due to the variability of biological activity of raw samples.

Considering the values along the composting process, the overall nitrogen loss was between 50% and 73% of initial nitrogen content. The maximum nitrogen loss occurred in the experiment with the lowest C/N ratio (C/N ratio of 13). For the subsequent trials, C/N ratio of mixtures was better-adjusted to close to 20, lowering the overall nitrogen loss to 50-57%. The values found in literature are highly variable, finding nitrogen loss values between 17 and 63% (Jiang et al., 2014; Lim et al., 2017; Wang et al., 2017). Nitrogen loss was substantial, particularly in first active degradation phase, probably intensified by the thermophilic temperatures and rather basic initial pH in PS. Nitrogen loss involves an important weakness of the composting process, not only leads to a lower content of nitrogen in the end-product but also implies a critical secondary pollution source in form of ammonia, associated to an increasingly concerning environmental impact (Wang et al., 2018a).

6.3 ASSESSMENT OF END-PRODUCT QUALITY

As the main objective of this chapter was to demonstrate the production of a high quality and safe bio-based fertiliser using PS as suitable feedstock through advanced composting, a comprehensive assessment of the quality parameters of the BBF was performed, including: agronomic quality of the product through nutrient content, stability and agronomic trial in pot; and safety assessment by measuring toxic pollutants such as heavy metals and organic pollutants of emerging concern. All this work was done in the framework of the SMART-Plant project and thanks to the collaborations with University of Verona (agronomic), University of Rome (determination of organic pollutants of emerging concern, in particular PPCP and pesticides) and Autonomous University of Barcelona (determination of heavy metals and some complex organic pollutants such as PAH)

6.3.1 AGRONOMIC QUALITY OF BIO-BASED FERTILISER

The physic-chemical characteristics of the end-products obtained are presented in Table 6-4. All products presented high VS contents always higher than 50% (d.b.) and therefore always with an estimated minimum TOC content above 30%. Hereupon, it is important to remark that, bearing in mind the current challenges related to climate change mitigation, application of organic carbon-rich bio-based fertilisers in croplands have been recognised to be an excellent way of carbon sequestration (Marks et al., 2021; Tiefenbacher et al., 2021). Total nitrogen content values of all end-products were above 3% (d.b.). Total phosphorus content in the end-products was in the range of 3.5-5.4% (d.b.) which gave an estimated P_2O_5 content of 7.9-12.3%, much higher than previous reported results that were in the range of 0.9 -2.2% of P, d.b. (Grigatti et al., 2019; Rékási et al., 2019). Manure based composts and organic amendments also showed lower phosphorus content (0.6-1%P, d.b.) (Mackay et al., 2017; Viaene et al., 2017; Puyuelo et al., 2019). Regarding potassium, values found in the BBF were more variable (between 0.7 and 1.3%, d.b., which equals 0.8 and 1.6% of K_2O), yet comparable to some composted products obtained from conventional sewage sludge or manure. Comparatively, BBF obtained in this study presented N and P contents equivalent to various organo-mineral fertilisers reported in literature which were supplemented with phosphate rock (Kominko et al., 2019; 2021; Rehman and Qayyum 2020). In the mentioned works, additives or co-substrates such as urea, diammonium phosphate, phosphoric acid or phosphate rock were added and consequently increasing phosphorus contents of organo-mineral fertilisers. Additionally, as a comparison to other wastewater based recovered products, struvite, hydroxyapatite and other phosphoric minerals consistently presented 20-60% higher P_2O_5 contents than the current BBF (Santos et al., 2021). However, BBF and its production show potentially some benefits over chemical precipitation, adsorption and thermochemical solutions for phosphorus recovery. Although biological P recovery might imply lower recovered phosphorus concentrations in the products, the high energy, chemical and adsorbent requirements of the alternatives mentioned are typically linked to higher operational and environmental costs (Santos et al., 2021). Also, additional pre and post steps are usually required (previous incineration or conditioning and post purification steps, e.g.) to achieve a marketable product.

Considering their nutrient content and the requirements in the new European fertilising products regulation, the BBF produced could be categorised as solid organic fertiliser (category PFC 1(A)(I)) or the so-called bio-based fertiliser. Table 6-4 details the minimum content of nutrients required for the targeted category considering at least one primary macronutrient: nitrogen, phosphorus or potassium. Additionally, the category defined for compost (CMC3) is also considered in the table as baseline.

The BBFs obtained has the characteristics to be considered as a solid organic fertiliser, presenting potential to partially substitute mineral fertilisers, demonstrating, thus, its added value as a fertilising product far beyond its baseline category as compost (CMC3).

DRI and AT_4 values demonstrated that end-products were properly stabilised in all cases. Adani et al. (2004) suggested a DRI value of maximum 1 $gO_2\ kg^{-1}VSh^{-1}$ to indicate medium stability products, while a mature product would have an OUR of 0.5 $gO_2\ kg^{-1}VSh^{-1}$. The degree of stability achieved indicates that the microbial activity of the organic fertiliser is low enough to guarantee no further degradation once the BBF is applied in agricultural soil (Wichuk & McCartney 2010; Komilis et al., 2011; Saveyn and Eder, 2014). In practical terms, the European fertilising products regulation (2019/1009) did not establish any stability criteria for the category of solid organic fertiliser (PFC 1(A)(I)) although it did for compost category (CMC3).

Table 6-4. Physic-chemical characteristics of bio-based fertiliser (BBF) obtained through advanced composting of PS.

Parameter	DM	VS	TKN	P_2O_5	K_2O	pH	EC	DRI_{24h}	AT_4	Rottegrade category
	%, w.b.	%, d.b.	%, f.b.	%, f.b.	%, f.b.		$mScm^{-1}$	g $O2kg^{-1}$ VS h^{-1}	$gO2kg^{-1}$ VS	
BBF	61.2-68.3	57.5-65.4	3.4-4.2	7.9-12.3	0.8-1.1	5.8-6.6	2.9-3.9	0.4-0.9	35-58	V
PFC 1(A)(I)* primary ingredient			>2.5	>2	>2					
CMC3*								<0.4		>III

* Categories PFC 1(A)(I) and CMC3 in 2019/1009 European fertilising products market regulation

The stability criteria coincided with what was suggested in End-of- Waste criteria for biodegradable waste subjected to biological treatment (Saveyn and Eder, 2014), being a stable product demonstrated either by a respiration index below 0.4 $gO_2h^{-1}kg^{-1}VS$ or self-heating test value of minimum level III by Rottegrade test. Additionally, cumulative AT_4 index was also suggested to be sensitive to correctly categorise composts by their stability (Ponsá et al., 2010), in this case at a fixed temperature of 37ºC, and it is in fact used in Austria and Germany, being required in German regulation before biowaste landfilling (through SAPROMAT test) (Barrena et al., 2006). Finally, American society of testing and materials proposed a different definition of AT_4 index and considered stable a compost with a AT_4 value 35-50 g O_2 $Kg^{-1}VS$. Roughly, the criterion mentioned demonstrate the satisfactory stability of the BBF produced.

Considering all the results obtained, it could be stated that high quality and stable bio- based fertiliser can be obtained from PS with an efficient performance and rather short active degradation period (maximum 20 days) by implementing OUR based aeration strategy. However, rather than the comparison with other products obtained in literature, there is no established quantitative methodology to assess the quality of a fertilising product. With the aim of standardising the measurement of compost quality, some attempts have been made to find a suitable compost quality index (Saha et al., 2010; Puyuelo et al., 2019). This work suggests an equivalent methodology for bio-based fertilisers that could contribute to quantitatively assess the quality of this kind of new products developed in the framework of European circular economy. Accordingly, nutrient content (NPK), organic carbon content and respirometric activity of BBF are used for the calculation of the fertilising index (FI). The most restrictive scoring system suggested by the mentioned authors was used to calculate the FI of the bio-based fertilisers obtained in the present work. Appendix II shows the scoring system used for the calculation of the FI, which was always between 3.9 and 4.1 out of 5.

Figure 6-3. BBF produced as mixture (left) and after sieving to 6mm (right).

Considering that FI values categorised above 3.5, the products obtained can be considered as high-quality BBF. Therefore, it can be stated that the high quality of the BBF claimed before was indeed quantitatively demonstrated.

Finally, it was regarded as a fundamental point understanding the effects of BBF in crop yields and P-bioavailability compared to currently used fertilisers. In this regard, to assess the effective use of the BBF, an agronomic bioassay lasting 8 weeks was carried out in pot tests of maize plants. For the tests performed in pots, the application of 25 mg of phosphorus per kilogram of soil was fixed, reaching the recommended P_2O_5 dose for maize plants (220 kgP_2O_5/ha) (Perelli, 2009). Negative (C-) and positive (TSP) controls were used to evaluate the effect of the use of the assessed BBF in growth parameters, nutritional status of plants and effective use of the product as P source.

Briefly, growth parameters of the BBF application, measured through the length of stem and leaf numbers, were comparable to the positive control (TSP) (Figure 6-4).

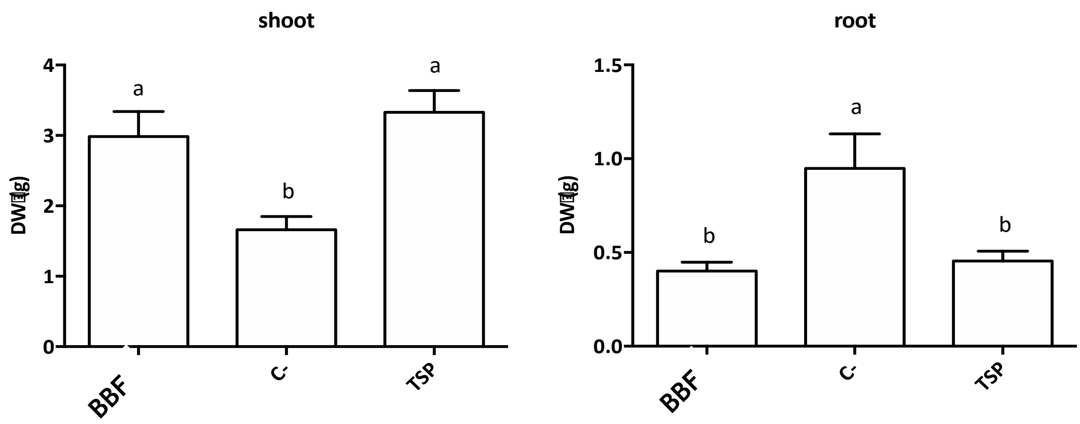


Figure 6-4. Dry weight of shoot and root tissues sampled at the end of experiment for maize plants treated with Bio- based fertiliser (BBF), negative (C-) and positive (TSP) controls. Data are expressed as mean ± s.e. (n=5); different letters indicate significant differences between groups (one-way Anova with Turkey's post hoc test, $p < 0.01$).

After 8 weeks of test there was no significant difference between BBF and TSP treatments, although negative control (no P source; C-) showed comparatively shorter stem length and lower leaf numbers. Regarding nutritional status of plants, no difference was found in SPAD index value among BBF, TSP and C- treatments. However, both fresh and dry shoot and root weights in BBF application were comparable to TSP positive control, and were significantly higher compared to C- treatment.

Consequently, plants with C- treatment seemed to show symptoms of P deficiency as they comparatively showed a more developed root apparatus (Hawkesford et al., 2012). Considering the root weight of the plants with BBF treatment, it can be stated that this product would be a good source of P for maize plants, achieving a fertilising performance comparable to mineral fertiliser. Moreover, from the multi-elemental analysis performed on shoot and roots, accumulated K and P levels were significantly higher in BBF treatment than the negative control and equivalent to the positive one (Figure 6-5).

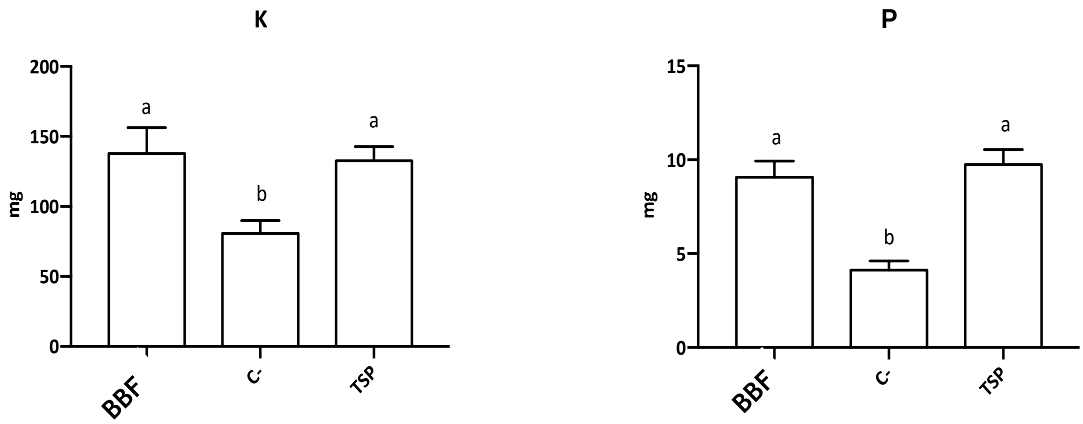


Figure 6-5. Total quantity of K and P accumulated per plant. Bio-based fertiliser (BBF), negative (C-) and positive (TSP) controls. Data are expressed as mean ± s.e. (n=5); different letters indicate significant differences between groups (one-way Anova with Turkey's post hoc test, p< 0.05).

6.3.2 SAFETY ASSESSMENT OF BIO- BASED FERTILISER: HEAVY METALS, EMERGING ORGANIC POLLUTANTS AND PESTICIDES

Within the wastewater treatment, pollutants tend to concentrate in treatment resultant sludges. In fact, sewage sludges have been excluded from the feedstocks permitted in the regulation on the market of EU fertilising products (EC 2019/1009) due to the extremely high heterogeneity of sludges as they are unable to guarantee the safety of the products obtained from them. In the work developed, though, safe sludge derived products are claimed, provided that adequate feedstocks are selected from the heterogeneity of sludges produced in the water sector. Classically, heavy metals are a major concern and their limits have been stablished in several regulations. However, other emerging micropollutants such as antibiotics, pathogens, pesticides and other organic complex compounds generate nowadays an increasing concern. This kind of emerging pollutants can be also retained in sludge given their extensive use and the limited elimination potential of such compounds in conventional wastewater treatment plants (Khadra et al., 2019). Currently no regulatory limits for the safe use of fertilisers in agricultural soil are established for the majority of the organic compounds mentioned. However, considering the one health umbrella, they might be a potential source of contamination through the soils and waters in which sludges or treated waters are discharged, posing therefore a decisive risk, either for human health or environment.

Hence, for the assessment of safety parameters of the BBF obtained, a fairly complete list of pollutants was measured in a representative sample of the BBF and used to evaluate the compliance of safety parameters. The list of the parameters analysed included the legally required heavy metals and a wide range of emerging organic pollutants such as pesticides, pharmaceuticals and personal care products (PPCP). Specifically, polycyclic aromatic compounds (PAH, 16 contaminants in total), chloroalkanes, pesticides (108 compounds) and PPCPs included in the EU watch list of 2018 (Commission Decision 2018/840, European Commission, 2018) were targeted in the analyses.

The BBF presented heavy metals contents bellow the limits established in relevant European regulations (Table 6-5), even below the most restringing European organic farming regulation 2092/91 and highest quality compost category in the Spanish Royal Decree regarding compost, except for zinc and mercury (EEC, 1991; BOE, 2013).

It is worth mentioning that compared to sludge-based organo-mineral fertilisers described in literature and defined as safe and ready-for market fertilising products, the BBF of this work presented consistently lower heavy metal contents (Kominko et al., 2021).

Table 6-5. Heavy metals content in a representative sample of BBF and relevant European and Spanish regulatory limits for its application into soil.

		Limits in European regulation				Limits in Spanish regulation	
Heavy metal	BBF	[1]	[2]	[3]	[4]	[5]	[6]
Mg	6959 ± 103						
Cr	24.4 ± 1.4	100		100	70	70	250
Ni	16.0 ± 0.9	50	50	50	25	25	90
Cu	63.2 ± 1.2	200	300	100	70	70	300
Zn	290 ± 3	600	800	300	200	200	500
Cd	0.32 ± 0.01	1.5	1.5	1	0.7	0.7	2
Pb	15.8 ± 0.4	120	120	100	45	45	150
Hg	0.482 ± 0.231	1	1	1	0.4	0.4	1.5

*All values are given in mg kg^{-1}TS. [1] End-of- waste criteria on biodegradable waste subject to biological treatment (Saveyn and Eder, 2014); [2]EU- Regulation 2019/1009 on the market of EU fertilising products, category organic fertiliser (FPC1) (EU, 2019); [3]EU- Commission Decision for Eco-label to growing media (European Commission, 2015); [4] EU Council Regulation No 2092/91 on organic production of agricultural products and indications referring thereto on agricultural products and foodstuffs (EEC, 1991); [5] and [6]Spain- RD 506/2013 sobre fertilizantes y afines, Spanish regulation. Categories A1 and B1, respectively (BOE, 2013).

Regarding emerging pollutants, only few of the targeted pollutants were detected in the BBF. All the PAH compounds analysed were below detection limit of 0.010 mg kg^{-1}TS, whereas naphthalene and phenanthrene were detected (0.014 and 0.020 mg kg^{-1}TS, respectively) in raw PS (Table 6-6). The overall PAH content of PS was in the low range of what was found in literature for conventional sludges used to formulated organo-mineral fertilisers (Kominko et al., 2021). PAH removal through composting could be achieved up to 100% (Biel-Maeso et al., 2019; Guo et al., 2020), being PAH removal positively correlated with temperature, pH, TOC, total nitrogen and C/N ratio.

Table 6-6. Emerging pollutants content in a representative sample of PS and the BBF obtained.

Compound type	Compound	PS	BBF	Quantification limit	Unit
PAH	Naphtalene	0.014 ± 0.002	BQL	0.010	mg kg^{-1}TS
	Phenanthrene	0.020	BQL		
Chlonoalkanes (C10-C13)		0.025 ± 0.000	BQL	0.010	mg kg^{-1}TS
Pesticides	Imidacloprid	BQL	6 ± 1	6.5	µg kg^{-1}TS
PPCP	Clarythromycin	50 ± 8	31 ± 8	1.7	µg kg^{-1}TS
	Azythromycin	506 ± 18	342 ± 19	4.6	
	Ciprofloxacyn	597 ± 6	217 ± 8	26.5	
	Estrone	22 ± 11	BQL	4.2	

BQL: Below quantification limit. Quantification limit was 0.010 mg kg^{-1}TSfor PAH compounds and chloroalkanes, while it was compound dependent in pesticides and PPCPs (Benedetti et al., 2020).

Similarly, chroloalkanes were only detected in raw PS (0.036 ± 0.004 mg kg^{-1}TS) (Table 6-6). Those results suggest that advanced composting would be effective for the degradation of this kind of compounds. Regarding pesticides, only imidacloprid was detected in the BBF (6E-03 mg kg^{-1}TS) (Table 6-6). However, this value showed high uncertainty as the value was close to the limit of quantitation. This compound is frequently found in sewage samples due to its widespread use in plant cultivation. It was not detected in raw PS and that could mean a concentration effect as well as rather limited biodegradation during composting process. Otherwise, its origin could also be attributed to the pruning waste used as bulking agent for advanced composting process rather than the raw PS itself. The change of the bulking agent may clarify its origin in future works.

Finally, from the 15 PPCP substances analysed (selected from the EU watch list of 2018), only 3, which were belonging to antibiotics class, were detected in the BBF: clazithromycin, acythromycin and ciprofloxacin, while estrone was also detected in raw PS. In general, the concentration of the three of them in PS was among the range of what literature established for other raw primary and secondary sludges (Lillenberg et al., 2010; Mailler et al., 2016; Khadra et al., 2019). The three compounds were reduced in the BBF obtained through the advanced composting process, being this reduction limited in the case of clacitromycin and azythromycin (38% and 32.4%, respectively) and more substantial in the case of ciprofloxacin (reduction of 63.7%) (Table 6-6).

Coinciding with previous literature, composting seems to be an effective method for the removal of several residual organic compounds. For instance, Biel- Maeso et al. (2019) reported removal efficiencies between 87% and 100% for a number of emerging contaminants including PPCP, PAH and PCBs. Composting has been demonstrated to be also very effective to remove compounds belonging to tetracyclines, sulphonamides and macrolides families (70-99% removal rates) (Ezzariai et al., 2018; Zhang et al., 2019). However, ciprofloxacin, which belongs to fluoroquinolone family, showed variable biodegradation potential through composting in previous literature and it has been recognised to remain in the final product after sewage sludge composting (Lillenberg et al., 2010; Cheng et al., 2019; Khadra et al., 2019).

Finally, following the standardisation attempt mentioned in the previous section, safety index (SI) is presented in this section as a comprehensive measurement of the safety degree of a BBF. SI was calculated following the scoring method suggested by Saha et al. (2010) although more strict criteria were added regarding heavy metals following scoring values suggested by Puyuelo et al. (2019) and according to the FPC1 category in the European fertilising products regulation (2019/1009). The whole scoring system is detailed in Appendix II. The bio-based fertiliser obtained from PS presented a SI value of 3.5, meaning that can be considered as a safe product.

Apart from the mentioned parameters, guaranteeing the correct sanitisation of the organic waste is critical for the safe use and selling of the product obtained. Specifically, the presence of relevant pathogens should be assessed. For instance, according to the European fertilising products regulation (2019/1009) low appearance or total absence of Escherichia coli and Salmonella sp. should be guaranteed in organic fertilisers. Additionally, some European countries regulations specify the limits of other biological parameters in composts such as helminth eggs or viable seeds (Cesaro et al., 2015). However, the mentioned sanitisation parameters were not evaluated in the present work. Some temperature criteria are established in the 2019/1009 regulation to assume the correct sanitisation of the composting mixture in the CMC3 category. Although PS presented very good characteristics for its composting, the scale of the pilot plant (100L) did not seem to be enough for the correct achievement and maintenance of thermal inertia to guarantee such sanitisation. Thus, those sanitisation results were assumed not to be representative of the composting process. Furthermore, PS certainly gathers the biological characteristics to guarantee the required sanitisation once advanced composting would be implemented at full scale, leading probably also to an enhanced biodegradation of certain organic pollutants.

Considering the overall safety results obtained in the bio- based fertiliser, it can be stated that the BBF can comply with safety requirements established in the regulation on the market of EU fertilising products (2019/1009). Furthermore, results suggest that safe sludge derived fertilising products can be obtained when using specifically selected or produced sludges as feedstock. Nevertheless, sewage sludges are not accepted in the feedstock list permitted to produce fertilising products. In fact, this kind of limitation was also claimed for other fertilising products recovered from wastewaters, such as struvite (Santos et al., 2021).

6.4 THE USE OF EXHAUST MESOLITES AS ADDITIVE TO REDUCE NITROGEN LOSS

The main weakness found in advanced composting process was related to the nitrogen loss which is in turn associated to secondary pollution produced in process and reduction of the agronomic quality of the final product. Nitrogen loss during composting occurs via ammonia and in a lesser extent via nitrous oxide (Lim et al., 2017; Wang et al., 2017; 2018b). In recent years some attempts for its reduction have been addressed through the adjustment of physic-chemical parameters, aeration strategy and the addition of different chemical, microbial and mineral additives in literature. In this regard, the use of zeolites has been reported as an effective solution for the limitation of NH_3 volatilisation in composting of sludge (Villasenor et al., 2011; Awasthi et al., 2016).

Zeolites are microporous structures with large negatively charged specific surface area (Villaseñor et al., 2011; Awasthi et al., 2016; Lim et al., 2017). Therefore, zeolites have been demonstrated to be capable of effectively adsorbing the NH_4^+ (Lim et al., 2017; Wang et al., 2017; 2018b) retaining part of the nitrogen that otherwise would be lost by volatilisation.

The aim of adding zeolite to the composting mixture was to evaluate the extent in which nitrogen loss could be reduced by this supplementation. To do so, a composting trial was designed supplementing PS with exhaust mesolite (a synthetically produced zeolite) from cation exchange column operated by Cranfield University in the framework of SMART Plant project. This cation exchange column, consisted in a demonstration ion exchange plant (IEX) ($10 m^3 d^{-1}$) used as a tertiary treatment for wastewater that would allow nutrient recovery (Huang et al., 2020; Guida et al., 2021). Nutrients in saturated ionic exchange media were extracted through regenerant solutions and when recovery yield was not satisfactory enough, exchange media was replaced. The use of exhausted exchange media (mesolite) as additive for advanced composting process was expected to reduce nitrogen loss via ammonia volatilisation and increase the nutrient content in the BBF produced. The downstream valorisation of different residual flows produced in mainstream and sidestreams into an added value product, such as BBF, is in line with the overall aim of SMART Plant project. Moreover, the solution proposed boosts the global circular and recovery-based management of urban wastewater sector towards an environmentally sustainable system.

6.4.1 SPECIFIC EXPERIMENTAL SET-UP

Having this objective in mind, exhausted mesolite was added to PS in a 1:10 mass ratio (Wang et al., 2017) valorising in this case 21.8 kg of PS. PS-mesolite mixture was then mixed with pruning waste in a volumetric ratio of 1:2. Table 6-7 summarises the main physic-chemical parameters of the mesolites and initial mixture. Composting process was carried out using the system developed for that purpose in the framework of this thesis and described in Chapter 3.

Table 6-7. Main physic-chemical parameters measured in exhausted mesolites and initial mixture.

Parameter	DM (%, w.b.)	VS (%, d.b.)	TKN (%, d.b.)	C/N	Phosphorus (%TP, d.b.)	Potassium (%TK, d.b.)	pH	EC $mScm^{-1}$
Exhausted mesolite	65.78 ±0.01	10.48 ± 0.07	0.97	5.0-5.7	0.5	5.6	6.44 ± 0.03	0.43 ± 0.00
Initial mixture	61.1	79.4		20.1				

As a cation adsorbing media, mesolite was rich in potassium although nitrogen was not as high as expected. Figure 6-6 shows a picture of the used exhaust mesolite together with the initial PS-mesolite mixture and initial mixture assessed.

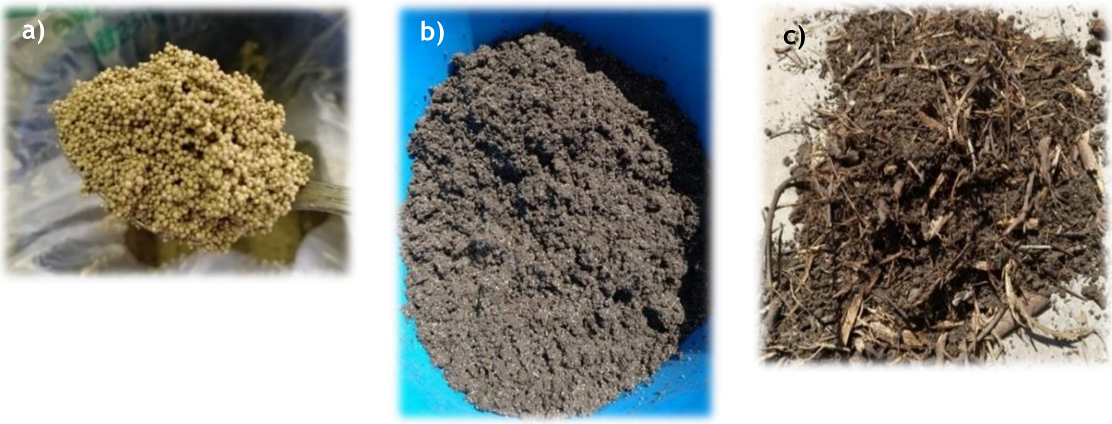


Figure 6-6. Mesolite (a) used as additive, PS-mesolite mixture (b) and initial mixture (c) in the trial assessing the use of spent zeolite to enhance nutritive characteristics of the BBF.

6.4.2 PROCESS PERFORMANCE AND PRODUCT QUALITY

Comparatively to the rest of composting assays performed, the profile achieved a comparatively lower temperature and biological activity (OUR) profile (Figure 6-7). The difference was certainly due to the inorganic zeolite addition in the same reactor volume to the detriment of organic matter mass available for its biodegradation and heat generation. In fact, in absolute terms, the organic matter provided by sludge resulted in almost the half compared to the previous experiments. Thermophilic range of temperatures was reached and maintained above 40ºC during almost the whole operational period. Density of initial mixture and thus, FAS (0.32 kg L^{-1} and 73.2%, respectively) of initial mixture would indicate that probably bulk mixture porosity was rather high and that would facilitate the excessive heat loss.

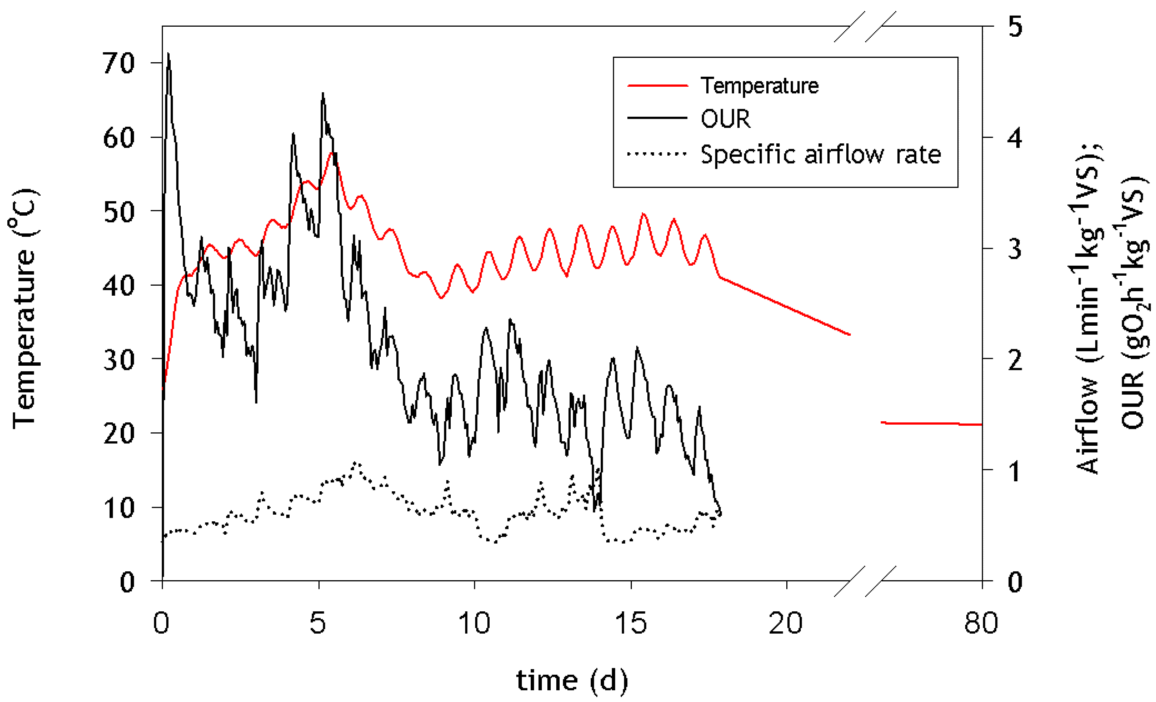


Figure 6-7. Advanced composting performance of PS with mesolite: evolution of bulk temperature (red line), OUR (black line) and specific airflow rate supplied (discontinuous line).

Zeolite addition was demonstrated to enhance biodegradation of organic matter (Wang et al., 2017) by increasing the porosity of mixtures, improving in turn the mineralisation of soluble organic matter. However, biodegradation of organic matter in the present batch was remarkably lower than the rest of the trials assessed (around 40% vs. 58-59%). Additionally, biological activity profile was comparatively lower than the previous PS composting trials treating solely PS. Maximum OUR value of 4.86 gO_2 $kg^{-1}VSh^{-1}$ was achieved, being considerably lower than the 8-16 gO_2 $kg^{-1}VSh^{-1}$ achieved in the rest of the composting trials. Thus, according to the OUR profile achieved, enhancement of biodegradation due to mesolites addition can't be affirmed.

Accordingly, to the lower OUR profile achieved, either overall and daily energy consumption were low compared to the rest of composting trials assessed (Table 6-8).

More remarkably, specific energy consumption was reduced by 28.6-41.2% compared to previous trials while RIE-EC was reduced up to 58%. In accordance to the previously mentioned, since the enhanced biodegradation due to mesolites addition is not clear, lower energy consumption seemed to be more related to lower air demand linked to lower biological activity.

Table 6-8. Energy consumption and energetic efficiency results of advanced composting trials of PS amended with synthetic zeolite.

Parameter	Energy consumption	Specific energy consumption		RIE-EC
	kWh	kWh kg^{-1}PS	kWh kg^{-1}VS-PS	kWh $(gO_2kg^{-1}VS\text{-}PS\ h^{-1})^{-1}$
Advanced composting of PS and mesolite	2.2	0.10	0.8	0.8
Advanced composting of PS	2.1-4.8	0.14-0.17	1.0-1.3	0.9-1.9

According to the mass balances obtained, less than 9%% of total nitrogen was lost along the composting process of PS mixed with mesolites, which resulted between 83 and 85% lower than previous trials with a well-adjusted C/N ratio. However, the extent in which addition of mesolites contributes to the reduction of nitrogen losses is not clear, since other facts, such as the lower temperature profile achieved, the overall lower airflow rate profile supplied and the acidic pH of mesolite, could also reduce the nitrogen loss. This kind of additives, such as zeolite and biochar have been lately used to effectively reduce pollutant gaseous emissions during sewage sludge and livestock manure composting processes. It seems that this kind of application is attracting interest within researchers as scientific publications assessing this effect notably increased within the last 5 years. In general terms, authors reported overall NH_3 volatilisation reduction of up to 20% during sewage sludge composting (Awasthi et al., 2016) and even up to 38% during pig manure composting (Lim et al., 2017). Some authors also achieved a drastic reduction (up to 60-80%) of N_2O with higher dosing of zeolites (Awasthi et al., 2016; Wang et al., 2017; 2018b), thus, limiting not only the loss of nitrogen through the zeolite amendment but also reducing the global warming potential of the composting process.

Apart from nitrogen, phosphorus and potassium contents were roughly stable from the beginning of the trial until its completion. Although there was clearly a dilution effect due to mesolites addition, phosphorus content of the BBF obtained was high enough to be categorised as, as organic fertilising product (category PFC 1(A)(I) in regulation 2019/1009) (Table 6-9).

Furthermore, the addition of zeolite improved the potassium content of the final product up to 4.5% (d.b.), higher than the potassium contents in previous BBF products in this chapter and composts obtained in literature, either after conventional composting or zeolite amended processes (Wang et al., 2017; Viaene et al., 2017).

Table 6-9. Physic-chemical characteristics of the BBF obtained through advanced composting or PS amended with spent synthetic zeolite addition.

Parameter	DM	VS	TKN	P2O5	K2O	pH	EC	DRI_{24h}	AT_4	Rottegrade category
	%, w.b.	%, d.b.	%, d.b.	%, d.b.	%, d.b.		mS/cm	g O_2 kg^{-1} VS h^{-1}	gO_2 kg^{-1} VS	
BBF from PS and mesolites	88.87 ±0.05	45.0 ± 0.5	3.1 ±0.1	5.2 ±0.3	11.9 ±0.2	5.76 ±0.04	7.5 ±0.1	0.73 ± 0.05	50.6 ± 0.6	V
BBF from PS	61.2-68.3	57.5-65.4	3.4-4.2	7.9-12.3	0.8-1.1	5.8-6.6	2.9-3.9	0.4-0.9	35-58	V

In fact, the calculated fertilising index for this product was of 3.5 and consequently categorising this material as a very good quality BBF. Wang et al., 2017 claimed that the addition of zeolite would also improve the quality of manure-based compost in terms of maturity indexes (pH; C/N ratio and germination index).

Figure 6-8. End product obtained from PS composting process amended with mesolite addition.

Moreover, according to the mentioned authors, the addition of zeolite was reported to significantly reduce the water-soluble fraction of copper and zinc due probably to the decreased mobility of such heavy metals, also supported by other authors (Zorpas & Loizidou, 2008; Villaseñor et al., 2011). Future work would consider including the specific analysis of toxic polluting compounds as well as bioassays assessing the mobility and toxicity of relevant compounds in the BBF.

6.5 MONITORING OF GASEOUS EMISSIONS ASSOCIATED TO ADVANCED COMPOSTING

Apart from the technical assessment of advanced composting, the appropriateness of the process proposed as an attractive sludge management and valorisation solution should be considered from a holistic approach, thus, considering also its environmental performance.

To this aim, gaseous emissions were daily monitored during the active degradation phase of composting processes. A plot showing the emission patterns of a representative advanced composting process is shown in Appendix V Gaseous emission patterns in advanced composting of PS coincided roughly with the literature found for sludge in similar scales (Maulini-Durán et al., 2013; Yuan et al., 2016; González et al., 2020). Emission peaks for GHG were detected in the first days of operation, probably associated to the anaerobic conditions during shipping of sludge samples and stripping effect after composting mixture porosity was adjusted and forced air supplied (González et al., 2020). Cumulative CH_4 and N_2O emission factors together with overall GHG emission factors in CO_2eq are shown in Table 6-10.

	Unit	advanced composting of PS	González et al., 2020	Maulini- Durán et al., 2013	Yuan et al., 2016
System description			identical set up	bench scale (50L, OUR based control)	bench scale (60L)
CH_4	kgCH_4 t^{-1}TS	0.26	0.14	0.06-0.07	
N_2O	kgN_2O t^{-1}TS	0.03	0.85	0.01-0.07	0.09-0.16
direct GWP	kgCO_2eq t^{-1}TS	18.38	230.00	5.02-23.24	83.3-130.7
Indirect GWP	kgCO_2eq t^{-1}TS	0.18			
Overall GWP	kgCO_2eq t^{-1}TS	18.56			
NH_3	kgNH_3 t^{-1}TS	18.08	5.13	1.3-4.1	0.57-1.4
H_2S	kgH_2S t^{-1}TS	0.10	0.17		
tVOC	kgC-VOC t^{-1}TS	3.26	6.20	0.6-1	
Odour emission	OU t^{-1}TS	3.55E+04	2.68E+07		

Values are given in kg of pollutant per metric ton of PS treated and with comparative purposes, relevant values in literature are also provided.

Table 6-10. Cumulative polluting gaseous emission factors (per dry sludge mass) during advanced composting and comparative values in literature.

Cumulative CH_4 results were higher than in our previous work treating conventional sewage sludge supplemented with diatomaceous earth (González et al., 2020) and also than the values reported in other works implementing also the OUR based control system for sewage sludge composting (Maulini-Duran et al., 2013). Conversely, cumulative N_2O emissions were lower than both mentioned works. In previous works, it was demonstrated that OUR based system was able to improve the overall GHG emissions when treating OFMSW (Puyuelo et al., 2014) and the results obtained for sludge in comparison to literature seem to be in accordance with that statement (Yuan et al., 2016; Han et al., 2018b).

Calculating the equivalence of the measured gases in global warming potential, advanced composting of PS emits 18.38 kg of CO_2eq per metric ton of dry sludge treated. This value was markedly lower (up to 40 times lower) compared to conventional sewage sludge composting, which highlights the global warming mitigation capability of advanced composting (Yuan et al., 2016; Han et al., 2018b). Additionally, analogously to biodrying trials, estimated CO_2 equivalents associated to energy consumption were calculated according to the estimation factors suggested in Product Environmental Footprint (PEF) calculation (Environmental Footprint secondary data sets version EF 2.0, 2020) and shown in Table 6-10.

Emission peaks of NH_3, tVOCs and odours were related to high temperatures and airflow rates. Both maximum emission rate of NH_3 and its cumulative emission factor were significantly higher (between 58 and 83%) than equivalent literature found (Maulini-Durán et al., 2013; Yuan et al., 2016), including our previous work with exactly the same pilot set up and same control algorithm (González et al., 2020). Associated to its side stream treatment origin, nitrogen content of PS was exceptionally high. Also, its pH was rather basic and its C/N ratio was half of what Maulini-Durán et al. (2013) reported. Accordingly, intense ammonia volatilisation was indeed expected in first days of thermophilic phase while the subsequent biodegradation of proteinaceous material probably contributed also to rather high NH_3 emission rates.

VOC emission values found in literature are highly variable, probably related to variability on feedstocks and experimental set ups, mainly related to appropriate aeration (Maulini-Duran et al., 2013). Compared to our previous work with conventional sludge composting, tVOCs emissions were reduced when valorising PS (González et al., 2020) although other authors with equivalent aeration system reported lower values (Maulini-Durán et al., 2013).

H_2S emissions depend on the S content of the raw material and are normally generated under anaerobic conditions (Han et al., 2018a). H_2S was briefly detected in the first day of operation and it was never detected again. H_2S was probably produced during PS management and storing period under reductive environment and then striped out once forced aeration was supplied. Emission factor of H_2S was slightly lower to the values found in our previous work (González et al., 2020), probably associated to the chemical characteristics of sludges treated.

The last three compounds described are associated to odour formation during aerobic processes. Accordingly, their emission peaks coincided with the odour emission peak measured by dynamic olfactometry. Compared to our previous work, cumulative odour emissions were significantly reduced. No NH_3, tVOC or H_2S was detected along curing stage.

Regarding VOCs characterisation, 17 VOC families were detected and quantified in total and their relative abundance is shown in Figure 6-9.

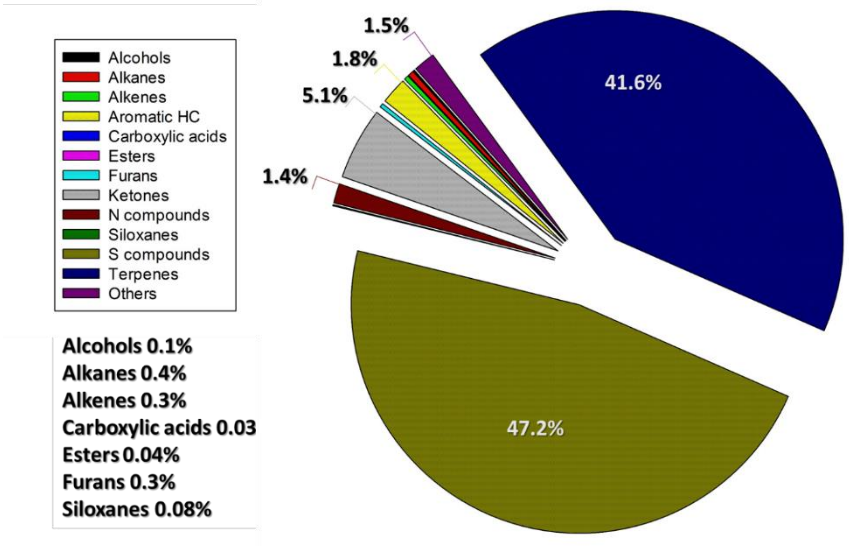


Figure 6-9. Main detected VOC families during maximum activity period of the advanced composting of PS.

Similar to what was found for biodrying, terpenes were one of the main emitted VOC families (41.6% of the total VOCs emitted). The selection of a more suitable and inert bulking agent could limit the emissions of this type of compounds. As expected due to their typical appearance in sewage sludge composting, the other main emitted COVs family is the sulphur containing compounds (47.2%) such as dimethyl sulphide (DMS) or dimethyl disulphide (DMDS) (González et al., 2020).

Sulphur compounds could pose a crucial role for the social acceptance of a composting plant due to their association to sensorial experiences and thus, they should be carefully considered (Maulini-Duran et al., 2013).

As a conclusion, the results obtained suggest the overall efficiency of the system developed in terms of GHG emissions. However, NH_3 emissions seem to be currently one of the main weaknesses of the advanced composting process of PS which should be targeted if the process proposed here will be regarded as a sustainable technological alternative to valorise sludge. The addition of complementary materials such as mesolites could be an attractive alternative that would be worth investigating more in deep.

6.6 TECHNO-ECONOMIC ASSESSMENT OF ADVANCED COMPOSTING

A comprehensive assessment of an innovative technological solution ideally requires also the demonstration of its economic feasibility so it would attract investors. Since this demonstration can't be made empirically, the estimation of economic parameters through specifically developed economic models through LCC methodology is very helpful. Thus, considering experimental data obtained though the implementation of advanced composting, an economic model was developed in the framework of this thesis. Two case scenarios were assessed through the same model. A first scenario was defined to evaluate the economics in a hypothetic plant in which advanced composting would be implemented to produce a bio- based fertiliser from PS production as sidestream technological solution (SCENA). This scenario assumes advanced composting as an independent treatment, uncoupled from the sludge production step. Additionally, in this first approach, the break-even point of the system was estimated in which minimum population equivalent served in a WWTP for an economically feasible advanced composting solution was calculated (Imeni et al., 2019). Finally, this first scenario was completed considering the expansion of the process where sidestream SCENA technology implemented in the WWPT would be coupled with advanced composting of resulting PS. This approach was targeted within the framework of SMART Plant project in which our group collaborated with our partners from Verona University and Innovation Excellence Hub InnoEXC GmbH (INNOEXC). Within this scenario 3 sub-scenarios were studied assessing small, medium and big size WWTPs. Second, the adaptation of the economic model to a more spread PS produced in mainstream EBPR system allows increasing the impact of the solution proposed as a whole. Thus, this second scenario studied the economic performance of implementing an advanced composting system to valorise the PS produced in conventional EBPR systems.

6.6.1 SISTEM BOUNDARIES AND SCENARIO DESCRIPTION

The system in study included an input of PS and bulking agent. Information regarding specific PS production in the first scenario was provided by the WWTP in study (WWTP of Carbonera). This WWTP serves to 40,000 PE and treats yearly 3,650,000 m^3 of wastewater producing yearly approximately 29.41 m^3 of PS after the implementation of SCENA system as a sidestream wastewater treatment solution. The second scenario studied was adapted to a real WWTP with EBPR implemented in its mainstream treatment of wastewater: WWTP of El Prat (Barcelona, Spain). System boundaries for advanced composting LCC model for both case scenarios are shown in Figure 6-10.

Advanced composting process was simulated in the economic model through mass balances according to experimental data obtained. Main physic-chemical characteristics of P-rich sludge assumed in the model and operational parameters of advanced composting assumed for the mass balance were set according to experimental data obtained and detailed in annex III. End-product quality and its market price was calculated from mass balances and according to nutrient substitution values given elsewhere (Sharara et al., 2018) and they were afterwards used to calculate economic parameters.

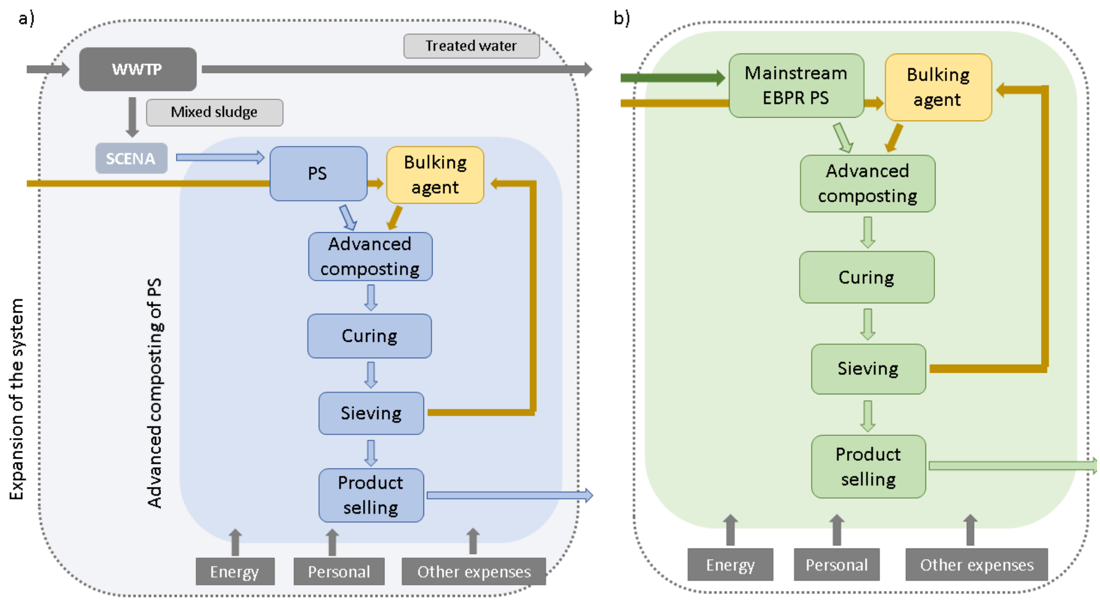


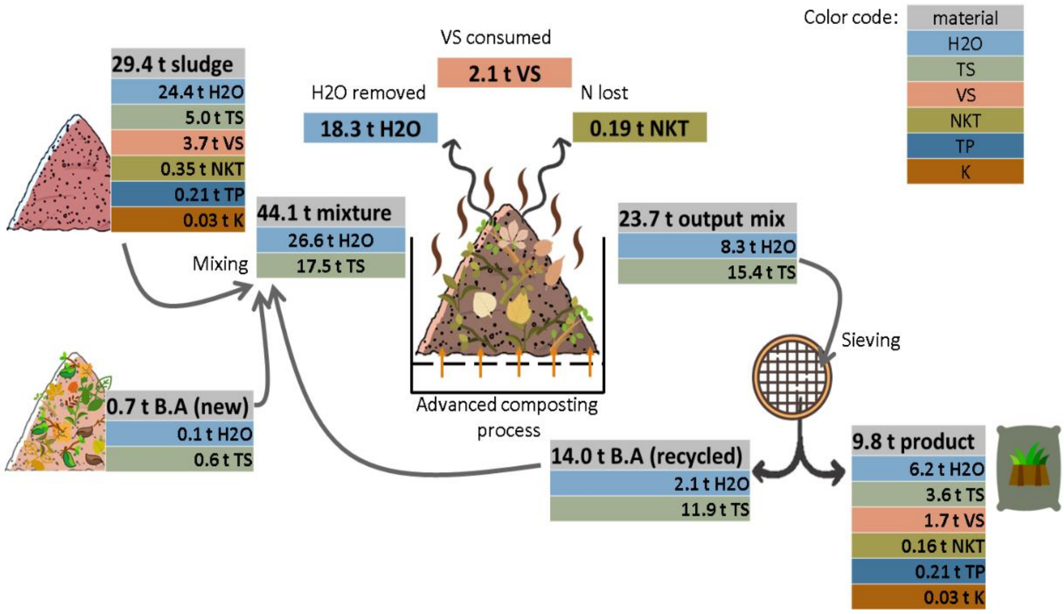
Figure 6-10. Advanced composting system boundaries considered for LCC model adapted to a) advanced composting of PS produced in SCENA system, including the expansion of the system and b) advanced composting of PS obtained from conventional mainstream EBPR system.

In the first case scenario, SCENA system in the modelled Carbonera plant (40,000 PE) would produce yearly 23.5 metric tons of PS which would be transformed into 7.8 metric tons of BBF yearly. The nutrients content of the product obtained in the modelled scenario (4.6 % NKT, 5.9 % TP and 0.8 % K, all in dry basis) would be enough to meet the requirements given for organic fertilising products (FPC1) in the European fertilising market regulation 1009/2019.

In the second case scenario, the yearly PS input of 61,500 metric tons would produce 12,946 metric tons of BBF with a nutrient content of 3.2% N, 5.0% TP and 0.9% K, all in dry basis. The detailed mass balances of both scenarios are detailed in Figure 6-11.

The fair market price of the bio- based fertiliser was calculated considering the substitution capability of the product given the main NPK nutrients (Sharara et al., 2018) and would be of 48.11 €/t. The value was in the high range of the compost prices in Spanish market.

a)



b)

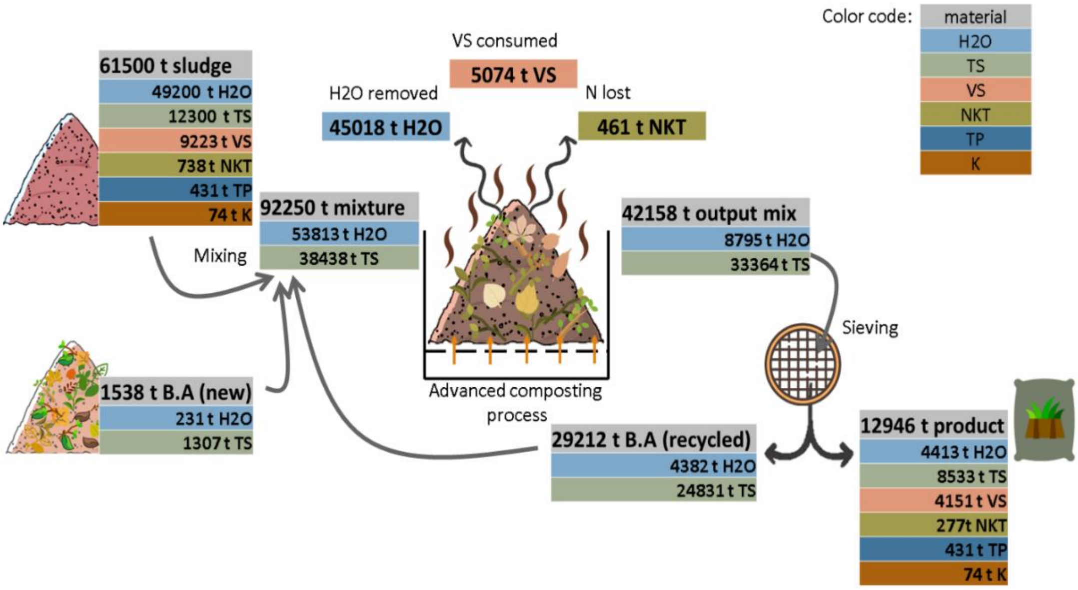


Figure 6-11. Mass balance of the both case scenarios assessing a) the coupling of SCENA technology and advanced composting and b) advanced composting of PS obtained in conventional mainstream EBPR system.

To give some numbers, the market prices for medium and low-quality composts (B and C categories) in Spain can vary considerably, reaching up to 40€ per metric ton of compost, although in some other cases compost is given by free (Alvarez de la Puente, 2007).

However, considering its characteristics and technology level in its production, the target for the product obtained could be high quality organic fertilising products market or even as ingredient for the formulation of tailored fertilisers. A selling price of up to 140€ t^{-1} have been estimated for complex formulated organic fertilising products (Kominko et al., 2019) making that option attractive for investors.

The specific design parameters estimated for the composting plant modelled are given in Appendix III.

6.6.2 FIRST CASE SCENARIO: TECHNO-ECONOMIC ANALYSIS OF BBF PRODUCTION FROM SCENA P-RICH SLUDGE THROUGH ADVANCED COMPOSTING

CAPEX and OPEX, incomes and overall benefits estimated for this scenario are given in Table 6-11, while CAPEX and OPEX distribution in the categories considered are given in Figure 6-13. Given the small size of the windrow (7.2 m^3) the overall CAPEX estimated was rather low (12,648€).

The major investment cost was associated to engineering services hired for project design, which were estimated to account for almost the 55.6% of the overall costs. The rest of the construction and equipment costs were always in the same range, regardless of the category accounting for between 8 and 13% of the total costs. Main costs associated to operation and maintenance of the composting plant would be derived from the mixing truck and energy bills, accounting for 44% and 25% of the yearly OPEX, respectively. Although the sludge input probably would not require the daily use of mixing truck, its rental would be considered as a fixed cost which would be in the range of what other authors reported (Ruggieri et al., 2009). It is worth mentioning that the calculation of personnel hiring costs were done according to the sludge input, estimating the number of working hours required for its management, which in this specific case is low.

Table 6-11. Main economic parameters obtained from the first case scenario through the economic model of advanced composting of PS.

Parameter	Value
Lifespan (y)	25
CAPEX (€)	12,592 €
OPEX (€/y)	6,837 €
Incomes (€/y)	1,059 €
Benefits (€/y)	-5,778 €
NPV	-147,587 €
IRR (5 years)	NA
IRR (10 years)	NA

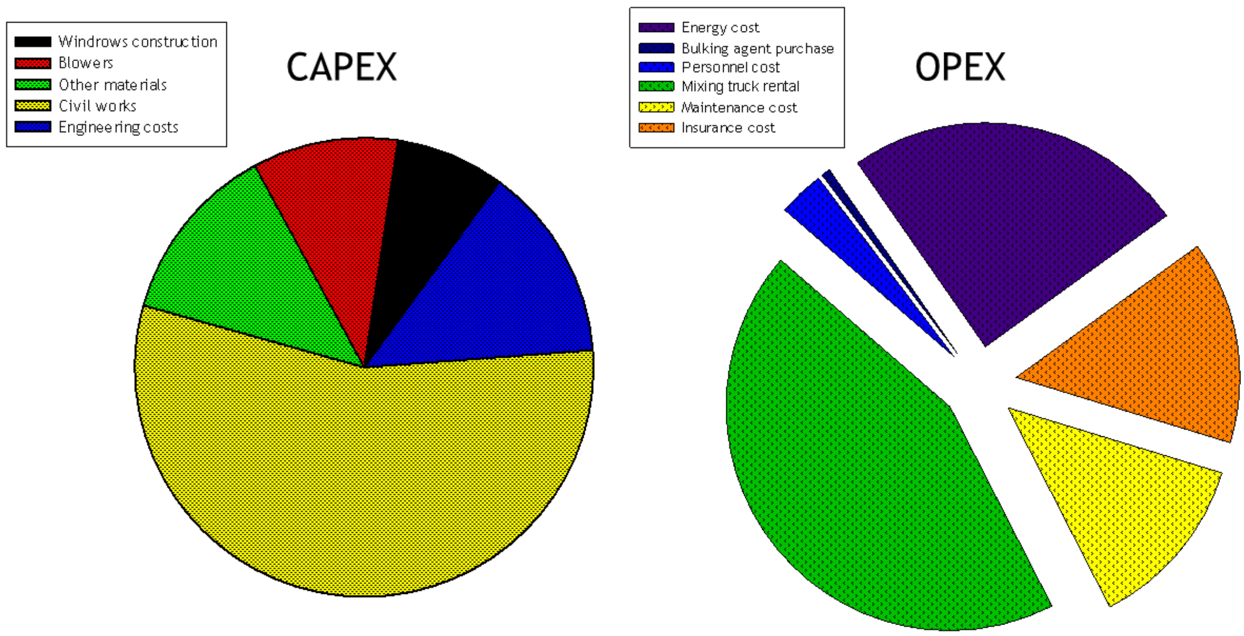


Figure 6-12. Estimated distribution of CAPEX and OPEX in the base case economic model developed.

At this point, as a benchmark comparison, it is worth to contrast the specific energy consumption of the system compared to other technological solutions aiming the recovery of phosphorus from wastewater, either directly, from sewage sludge or sewage sludge ashes. According to the values reviewed by Santos et al., (2021) phosphorus extraction from sludge ash would be associated to the highest energy requirements (over 20 kWh $PE^{-1}y^{-1}$), making both environmentally and economically more feasible the option of phosphorus extraction directly from sewage sludge (below 10 kWh $PE^{-1}y^{-1}$) (Santos et al., 2021). According to the model developed in the present work, the energy requirements associated to advanced composting would be of 0.27 kWh $PE^{-1}y^{-1}$ far below than the solutions based on chemical and thermochemical processes.

Under these circumstances, cash outflow would exceed cash inflow, leading to important economic losses in the lifespan of the project modelled. Not even increasing the market price of the product according to the organic fertilisers market would make the project economically feasible (Kominko et al., 2019).

To better assess the economic feasibility of the technological solution proposed, economic feasibility of PS advanced composting was evaluated for different treatment capacity plants to find the minimum plant size that would present an economically feasible scenario. The model was adapted according to the increase in the sludge volume treated, associated to the increase in the capacity of WWTP.

In the Figure below (Figure 6-14) the main economic parameters are shown under different WWTP capacity sizes. All the three economic indicators increased linearly concomitantly to the plant capacity size.

Concerning again to energy consumption, smallest scale composting plants would require 0.2-0.4 kWh PE^{-1} y^{-1}, whereas middle and biggest scale plants (over 100,000 PE) would require 0.07-0.1 kW PE^{-1} y^{-1} energy. These values are again significantly lower than the values reported for other technologies aiming the recovery of fertilising products such as struvites, hydroxyapatite and other phosphoric minerals (Santos et al., 2021).

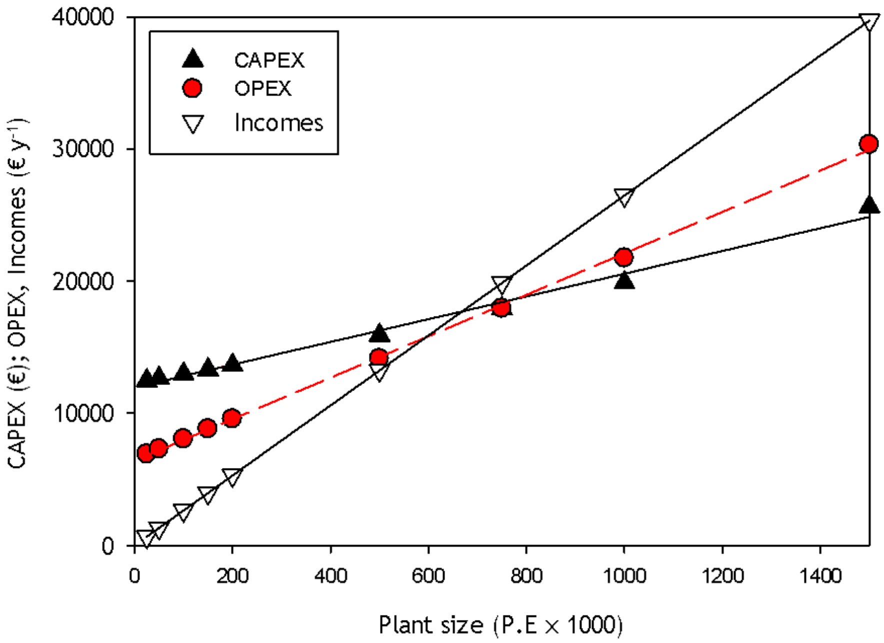


Figure 6-13. WWTP treatment capacity dependent economic parameters for advanced composting plant treating PS: black triangles indicate CAPEX, white triangles indicate OPEX and red circles indicate cash incomes.

Overall, considering the fitting models of the three economic parameters presented, there would exist a certain plant size in which economic performance could be feasible. According to the regression model based on NPV (Figure 6-15), the estimated break-even point was of 664,424 P.E served by the WWTP. Thus, this would be the minimum plant size (PS) able to overcome all the project associated costs, yet it would not be able to make any profit.

This scenario would be associated to a payback period of 24 years although it would not be able to achieve a positive IRR in 25 years. It should be noted though, that the economic feasibility of the project would depend directly on the real market of the BBF and the willingness of end-users to pay its estimated market price. Also, the avoided costs associated to sludge disposal greatly affect the economic feasibility of the project, particularly considering the gradual increase of sludge management costs.

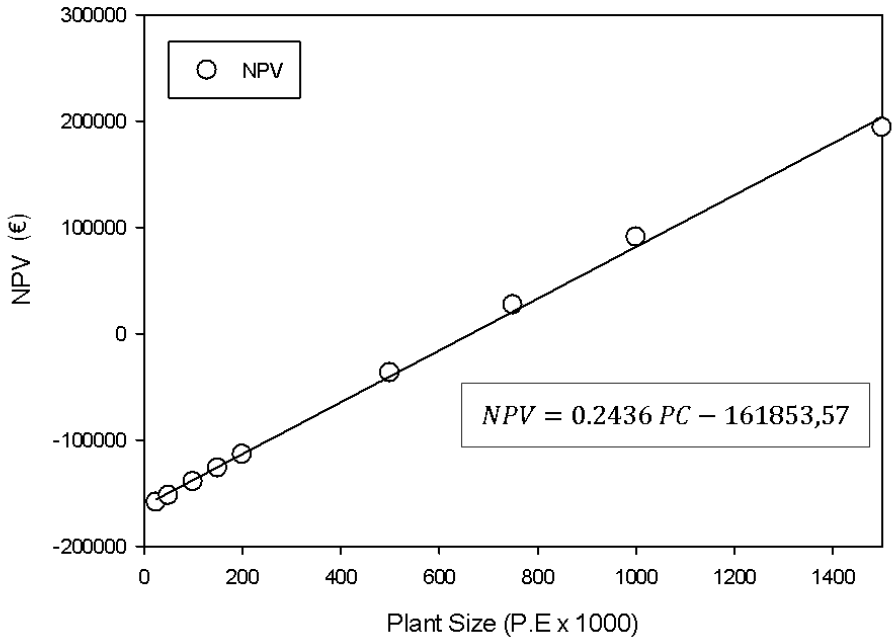


Figure 6-14. WWTP treatment capacity dependent NPV with its fitting linear regression model for advanced composting plant to valorise PS.

It is worth mentioning that both sludge treatment costs and BBF production costs were extremely high in small and medium capacity WWTPs, while the costs for higher capacity plants seem to be more realistic (Figure 6-15). More specifically, a wide range of disposal cost values can be found in literature which are mainly country dependent or treatment dependent. Some authors reported specific range of cost values depending on the sludge disposal method, finding the lowest disposal costs, 25-210€ per ton of sludge in land application while incineration had associated higher costs (38-438€ ton^{-1}) (Piao et al., 2016; Capodaglio & Olsson, 2020).

Usually, in conventional composting plants two are the economic parameter that are daily used and can be taken as benchmark: sludge treatment cost, which is given in € per ton of sludge treated in plant (in wet basis) and compost production cost which is in this case given in € per ton of product obtained. Thus, Figure 6-15 shows how sludge treatment costs and BBF production costs correlate with the capacity of the plant aiming to facilitate the comparison to the values that can be typically found in compost-related economic studies. A plant capacity size between above 200M P.E would be able to equal its sludge treatment cost to the most spread 30-70€ per ton of sludge treated. Analogously, compost production costs in literature for conventional sludge composting comprise 14-44 € per ton of compost produced (Ruggieri et al., 2009; Song & Lee, 2010). Even WWTPs with highest capacity would not be able to achieve comparable production costs, achieving a minimum cost of 83€ per ton of BBF produced, that in general terms would be the minimum BBF production cost achievable.

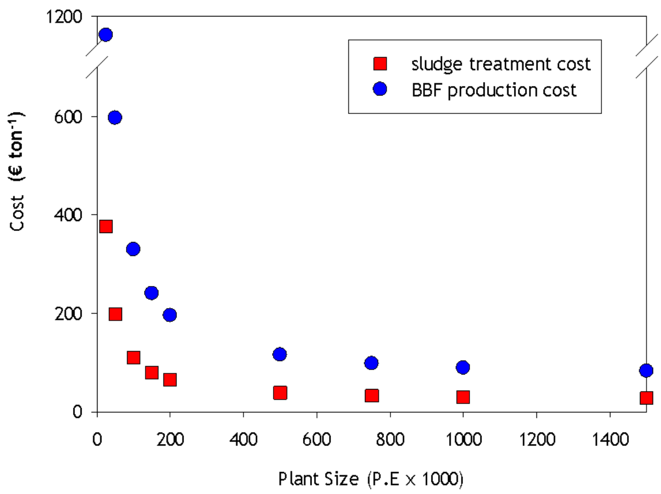


Figure 6-15. PS treatment costs and BBF production costs in different WWTP treatment capacity sizes (values given are referred in wet basis).

Additionally, and considering phosphorus recovery for wastewater into new-generation fertilisers, it is worth to compare the relative price of recovered phosphorus to phosphoric minerals obtained by chemical precipitation, adsorption or thermochemical processes. Thus, advanced composting of PS would permit a price associated to the recovered phosphorus of 0.5-7.0 € kg^{-1}P. These values are, in general, below the prices reported for phosphorus recovered in form of struvite or calcium phosphate (2-10€ kg^{-1}P) (Santos et al., 2021).

Finally, the savings in life cycle costs when implementing SMART Plant solutions, were then calculated and in the cases in which result would be positive, the SMART Plant solutions implemented would have a positive impact in the lifetime of the WWTP.

The implementation of SMART Plant solutions, either SCENA technology individually or coupled with advanced composting, would allow the increase of investment costs around 10%, while OPEX would be reduced around 25% in the lifespan considered (Table 6-12).

Table 6-12. Comparison of economic parameters of base case SMART WWTP and the same WWTP with coupled SMART Downstream B advanced composting technology implementation.

Scenario descriptor	small	medium	big
PE	50,000	100,000	250,000
YEARS	25	25	25
End of life costs (€/PE)	30	25	23
Base CAPEX – TOTAL (€/PE)	300	250	230
Delta CAPEX w SMARTech SCENA (€/PE)	330	275	253
Delta CAPEX w SMARTech SCENA + advanced composting (€/PE)	330.3	275.1	253.1
Base OPEX - TOTAL (€/PE/y)	450	400	400
Delta OPEX w SMARTech SCENA (€/PE)	338	300	300
Delta OPEX w SMARTech SCENA + advanced composting (€/PE)	340.1	300.4	299.3
SMARTech SCENA Benefits (€/PE)	0	0	0
SMARTech SCENA + advanced composting Benefits (€/PE)	0.3	0.2	0.2
LCC Results w/n SMARTech	780	675	653
LCC Results w SMARTech SCENA	698	600	576
LCC Results w SMARTech SCENA coupled with advanced composting	700	600	575
SAVINGS w SMARTech	10.5%	11.1%	11.8%
SAVINGS w SMARTech coupled with advanced composting	10.2%	11.1%	11.9%

Overall, the solution proposed would allow savings of up to 11.9% compared to a conventional WWTP. Highest treatment capacity plant would achieve the most beneficial results although the small capacity plants would also achieve considerable of savings (10.2%). At this point it is worth repeating and clarifying that the expansion of the system was done in the framework of the SMART Plant project.

Thus, the criteria for the methodology to assess the coupling of SCENA and advanced technology was made as a consortium decision. However, considering the overall economic results it would be interesting to assess a centralised advanced composting economic scenario as it could possibly lead to a better economic performance, even considering the uncertainties related to the inclusion of sludge transportation costs. Other feasible scenario would be the one in which conventional composting plants would be adapted towards advanced composting scenario in which high quality feedstocks selected in origin would be separately managed and treated to obtain a BBF with high added value, improving the overall economic performance of the plant.

6.6.3 SECOND CASE SCENARIO: TECHNO-ECONOMIC ANALYSIS OF BBF PRODUCTION FROM EBPR CONVENTIONAL SLUDGE THROUGH ADVANCED COMPOSTING

Conversely to the studied SCENA technology to obtain sludges with high quality, the implementation of mainstream EBPR systems is more conventional. Thus, coupling advanced composting to conventional EBPR system to produce organic fertilisers with added value is claimed to be a highly promising solution towards nutrient recovery. Compared to SCENA system, specific sludge production ratio is far higher in conventional mainstream EBPR systems and physic-chemical characteristics of sludge can be considered comparable. Considering the abovementioned, the economic scenario predicted was expected to be more positive than the one assessed in the first case scenario. To test this hypothesis, the economic model developed was adapted to the data provided by WWTP of El Prat Which has an EBPR system implemented in its mainstream. WWTP has a treatment capacity of 2 million population equivalent and a yearly EBPR sludge generation of 76,875 m^3, which is rich in phosphorus (3.5% in dry basis).

Considering this approach, a composting plant with 34 windrows of 150m^3 associated to an investment cost of 1,614,894 € was modelled with yearly operational costs of 969,872 €. A yearly BBF production of 12,946 metric tons containing 3.2%N, 5.0%P and 0.9%K in dry basis would present a fair market price of 70€ per metric ton of BBF. Product selling would achieve 1,175,059 € yearly net benefits. In a lifespan of 25 years, the NPV associated would be 25,839,746 €. This scenario shows a payback scenario of 1 year with very high IRR already in 3 years (Table 6-13).

Table 6-13 Economic parameters of second case scenario of advanced composting: valorisation of conventional EBPR sludge

Parameter	Value
CAPEX (€)	1,614,894 €
OPEX (€ y-1)	969,872 €
benefits	1,175,059 €
NPV in 25y lifetime	25,839,746 €
IRR 3 years	188%
IRR 5 years	243%
Payback period	1 year

Figure 6-16 shows the regression models fitting the CAPEX, OPEX and Incomes for the WWTPs with increasing treatment capacities analysed. Considering these results, a WWTP above 26,000 P.E size would be associated to economic profit, achieving positive IRR within 5 years in plants above 100,000 P.E and within 10 years in plants above 50,000 P.E.

Also, it is worth mentioning that under these circumstances and in plants above 150,000 P.E, sludge treatment costs would equal the fees paid for sludge management that currently are of 20€ t^{-1} (wb) approximately. Thus, considering the environmental benefits of suitably treating sewage sludge, modification of WWTPs above that size would be positive even in a non-profit scenario.

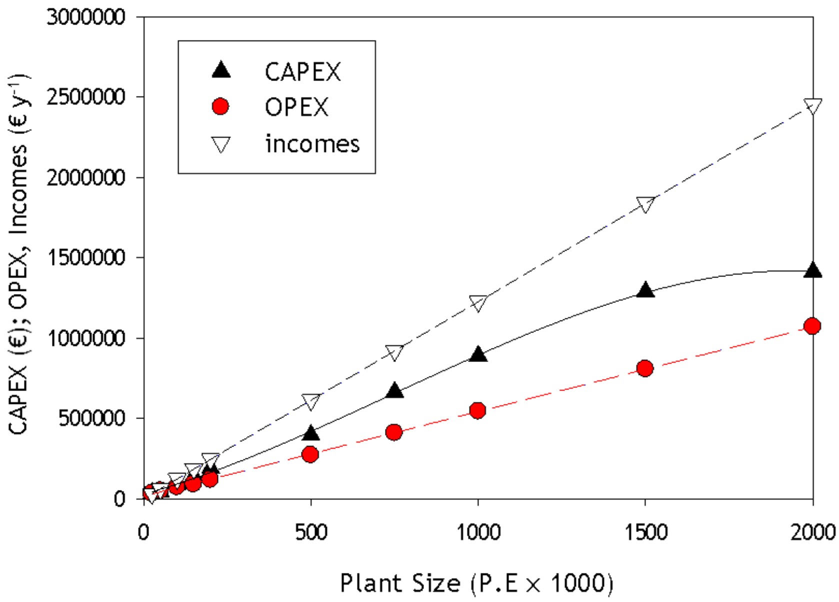


Figure 6-16. WWTP treatment capacity dependent economic parameters for advanced composting for conventional EBPR sludge valorisation: black triangles indicate CAPEX, white triangles indicate OPEX and red circles indicate cash incomes.

If the BBF proposed here will be able to reach the market of fertilising product it should be able to compete with the conventionally used mineral fertilising products, particularly in terms of product price, quality and safety. Technical feasibility of both quality and safety parameters was already demonstrated in previous sections while the effect of the market price was assessed through a sensitivity analysis of the economic model. To do so, economic parameters were estimated by using 3 potential market prices: (i) fair market price calculated according to nutrients substitution potential of the BBF, (ii) medium and (iii) low BBF selling prices of 40€ and 20€ per metric ton of product, respectively. Selling prices for sensitivity analysis were selected according to the values given in literature and real market prices facilitated by several sludge composting plants in Spain (COGERSA S.A.U., Arazuri WWTP, Manresa WWTP).

Figure 6-17 shows the regression model of the NPV for a lifespan of 25 years modelled for the same scenarios according to the market prices of interest. A WWTP serving to 100,000 P.E would be able to reach a positive economic scenario even in the worst BBF price scenario, estimating in that specific case, a payback period of 6 years and an IRR of 42% within 10 years.

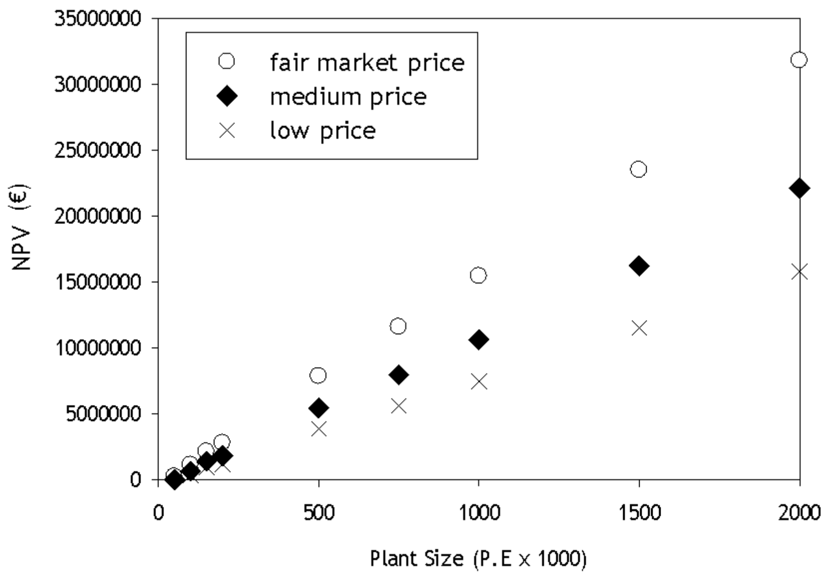


Figure 6-17. WWTP treatment capacity dependent NPV with its fitting linear regression model for advanced composting plant to produce BBF from EBPR sludge considering 3 market prices for BBF: a fair market price according to nutrient substitution capacity of BBF in the fertilising products market (70.6€/t); a medium selling price for organic fertiliser (40€/t); and a low selling price equivalent to the typical compost selling price (20€/t).

According to the breakeven analysis performed, plant sizes associated to zero-profit scenario would be 25,880 P.E, 37,057 P.E and 51,578 P.E in each product selling price case.

Considering the data provided by European Environmental Agency and the breakeven points calculated for the different market prices considered, around 40% of the WWTPs in Europe would be able to adapt their secondary wastewater treatment stages to an EBPR system (Waterbase UWWTD-v08) (EEA, 2020) and make economic profits, being thus able to serve to more than the 90% of the European population. Figure 6-18 shows the distribution of plant sizes in European Countries where 16 countries have at least 40% of their WWTP suitable to be adapted to a EBPR system followed by an advanced composting plant.

Overall, the adaptation of those plants would permit the yearly recovery of 154,886 tons of phosphorus in Europe. According to the 2,7 million tons of phosphate demand estimated by Fertilizers Europe, this scenario would be able to meet 17.6% of the phosphorus demand in Europe.

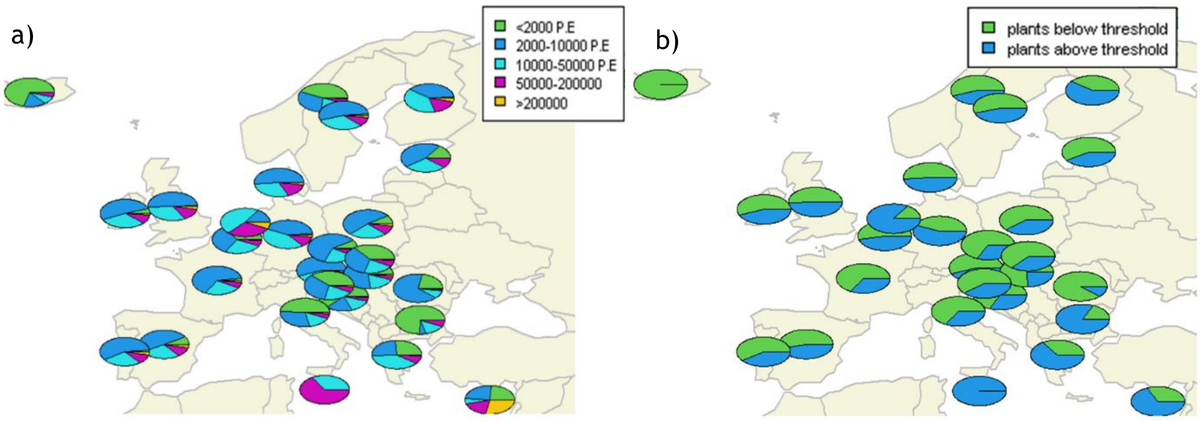


Figure 6-18. Wastewater plant size distribution in European Countries (a) and the plants that would be able to adapt their secondary stage by an EBPR system coupled with advanced composting leading to an economically feasible scenario (b).

6.7 MAIN STREGHTS AND WEAKNESSES OF THE ADVANCED COMPOSTING OF PS AS A SOLUTION TO RECOVER NUTRIENTS

This section aims to sum up the main findings, their potential impact in the current European context and some barriers that should be considered to effectively implement the solution proposed as an alternative for an efficient and safe nutrient recycling. To do so, the main strengths (S), weaknesses (W), opportunities (O) and threats (T) of the solution proposed are summarised (Figure 6-19).

Overall, permitting suitable sludges as feedstock for fertilising products would be coherent with the current European sustainability objectives, particularly aligning with the Circular Economy Action Plan and Farm to fork strategies. This solution would promote particularly:(i) nutrient recycling towards reduction of regional nutrient imbalances, (ii) restoration of degraded soils, (iii) contribution to a safe and efficient food production chain, (iv) promote the independency from imported natural resources, (v) the zero-waste approach and end-of waste status for sludge and (vi) reduction of secondary pollution through the reduction of nutrient loses.

Figure 6-19. SWOT analysis of advanced composting of PS.

Nevertheless, the weaknesses that this solution presents and should be addressed are the (i) intense ammonia emissions, (ii) variability of feedstocks and (iii) high production costs in small. capacity plants. All the three weaknesses have been assessed in this chapter. First, mitigation of ammonia emissions has been successfully achieved by the recycling of exhaust mesolites already used as adsorption media in tertiary wastewater treatment stage. Second, variability of feedstocks leading to unsuitable PS could be overcome by the monitoring of toxic pollutants in the influent wastewater. Also, and suggesting it as a long-term solution, eliminating potentially toxic industrial streams as influent in the WWTP could be helpful, although it would require major modifications on the sewer systems and infrastructures. Third, the reduction of production costs has been found feasible when the advanced composting technology is implemented in a conventional WWTP using EBPR as secondary treatment for wastewater. The sludges produced would allow the obtention of a high-quality product and the volume of PS produced in the mainstream would permit feasible production costs.

In contrast, the threats that this technology faces are mainly related to standardisation issues and lack of administrative support to effectively access to the European market. Current regulatory framework (2019/1009 regulation on European market of fertilising products) does not support nutrient recycling from sludge. Given also that the limitations for direct land application of sludges are increasing, it seems that the only option for nutrient recovery (phosphorus, potassium and other micronutrients, but not nitrogen or carbon) from sludge is through combustion ashes, that show other economic and environmental constrains. Additionally, lack of consumers' confidence and social acceptance of recovered BBF would limit their market accessibility, which is itself limited and highly competitive due to the price and variability of mineral fertilisers available. All in all, achieving the administrative support needed and aligning the sludge-related regulatory framework to the European circular economy action plan would help also to gain the confidence of potential BBF consumers opening a window to access to the market of fertilising products.

Therefore, assuming the overcoming of the abovementioned limitations, this technological solution would help aligning the water sector with the European economy action plan by boosting the end-of waste status of sewage sludge, contributing also to the effective nutrient balances in European regions and creating good quality jobs for European citizens. Besides, in terms of potential business-model, the alternative proposed here can be taken as commercial strategy as it presents good branding opportunities.

7. CONCLUSIONS AND FUTURE WORK

7. CONCLUSIONS

This thesis pretends to overcome the current physical-chemical, technological and regulatory barriers related with sustainable sewage sludge valorisation, in terms of both, energy and materials (mainly nutrients) recovery.

In general terms, we proposed the production of new-generation sustainable biomass fuels and bio-based fertilisers with high quality from selected sludges by applying and improving biodrying and advanced composting processes. The assessment of most suitable sludges in each case was demonstrated considering technical, quality, environmental and economic aspects. In the case of biodrying, this thesis proposes innovative process performance evaluation parameters while demonstrating the appropriateness and need of thoroughly considering the quality parameters of the products obtained. In addition, biodrying has been improved (mainly in terms of control strategies), high quality biomass fuel has been produced and we have definitely contributed to go beyond the current state of the art (specifically in the case of low porosity-high moisture feedstocks). In the case of composting, it has been demonstrated that suitable feedstocks can definitely lead to safe and high-quality bio-bases fertilising products that could perfectly fit with the EU regulations. Additionally, and still in the case of composting, it has been demonstrated that some strategies can be considered for significantly mitigating the polluting gas emissions (which is one of the main weaknesses of the process).

Overall, this thesis provides new information that might help to boost the transition of the water sector towards circular bioeconomy.

The main findings/conclusions drawn from the work carried out are listed below:

- A fully operative pilot plant (with 2 units: composting and biodrying) was developed in the framework of this thesis to assess biodrying and advanced composting processes of selected sludges. The pilot plant developed was able to monitor key process parameters (inlet and bulk temperatures, airflow supplied, oxygen content in outlet gases and energy consumption) and manipulate the control parameter (airflow rate) accordingly to the control algorithms implemented. Additionally, the appropriateness, robustness and replicability of the control algorithms developed was demonstrated.

- Successful completion of both processes was achieved obtaining in both cases high quality products with appropriate characteristics for their end-use.
- The feedstocks selected and the related technological solutions proposed contribute significantly to the alignment of the water sector with the European circular economy action plan by suggesting robust and easy-to implement technologies to recover energy and nutrients from a wide range of sewage sludges.
- Some more concrete conclusions referred to biomass fuel production through biodrying and bio-based fertilising products through composting are listed below:

BIODRYING PROCESS:

1. They were identified and selected the criteria to identify appropriate feedstocks, satisfactory biodrying performance and quality of end products.
1. Cellulosic sludge (CS), primary sludge (PRS) and pulp and paper mill sludge (PPS) presented appropriate characteristics for an effective biodrying process, demonstrated by their performance and products obtained that gathered promising quality parameters to be effectively used as alternative biomass fuel source.
2. It was demonstrated that biodrying process can be successfully applied to low-biodegradability sludges, such as secondary sludges, if an appropriate con-substrate is selected.
3. The biodrying process of CS was successfully assessed in terms of overall performance, energy requirement and product quality by comparing 3 temperature dependent airflow control strategies.
4. Considering all the parameters assessed, the S3 aeration strategy was the best performing strategy. This strategy maximised moisture removal by using high temperature in the thermophilic stage as the main driver followed by the enhanced moisture removal by supplying a significantly higher aeration rate during mesophilic phase.

5. High quality biomass fuel was obtained with quality parameters that can be considered equivalent to agro-forestry products that are currently in the market.
6. In general terms, it can be confirmed that the biomass fuel obtained from selected sewage sludges could compete with current marketable biomass fuels, in all referred terms: quality, environmental and economic feasibility.
7. As a concrete example, the net energy production, in the case of CS, would be of 2,42 kwh PE-1.
8. One of the most important contributions of this thesis to go beyond the state of the art in biodrying processes was: i) the definition of two new and highly meaningful performance indexes and ii) the demonstration of their appropriateness for the overall and comprehensive assessment of the biodrying process. These two indices are the so called: the Energetic Biodrying Index (EBI) and the Biodrying Performance Index (BPI).
9. The relevant polluting gaseous emissions were monitored during biodrying processes of selected sludges. Results obtained demonstrated the better environmental performance of biodrying process of sludge compared to composting process (considering the same process duration and equivalent to the maximum biodrying duration of 14 days) or other conventional drying processes. These results, furthermore, certainly contribute to filling the gaps in current state of the art of biodrying processes.
10. A LCC model was successfully developed to assess CS valorisation through biodrying.
11. The minimum plant size, estimated with the LCC model adapted for CS, shows a feasible economic scenario (break-event point) would be from 169.580 PE on. The scenario assessing the economic feasibility of coupling the Cellvation® technology with biodrying in a conventional WWTP resulted in positive economic performance.
12. When the biodrying technology is applied to valorise primary sludge, a medium size plant (33.855 PE) would be enough to achieve a positive economic performance opening a window for biodrying to be regarded as an effective sludge valorisation alternative for this feedstock.
13. Biodrying technologies can clearly contribute to the end-of waste status of sludge reducing, in turn, the dependency of WWTPs on non-renewable energy and heat resources.

14. The main barriers to enter the EU market for these new biomass fuels and compete with conventional biomass fuels, are the lack of standardisation and specific regulatory framework.

ADVANCED COMPOSTING PROCESS:

1. High-quality bio-based fertilisers (BBF) were successfully obtained from phosphorus rich sludge through advanced composting. The quality of the products was assessed in terms of agronomic and safety parameters which were demonstrated to comply with the requirements stablished in relevant regulations on fertilising products.
2. The dependence of high-quality products obtained with the quality and properties of selected feedstocks was demonstrated.
3. The safe use and application of BBF is considered the main challenge that will enable the effective application of these technologies. Consequently, the high quality and safety of BBF are the main target of the proposed solutions. In this thesis, we have demonstrated the absence or low concentration of the most relevant toxic pollutants.
4. Additonally, the agronomic quality of the BBF was demonstrated (i) first, by means of meaningfully high content of nutrients, and in particular phosphorus (up to 5.3% TP in dry basis) which was as high as in various organo-mineral fertilisers based on the use of phosphate rock. (ii) second, the effective use of the BBF was assessed by means of an agronomic bioassay. The results obtained demonstrate the effective growth of maize plants showing an agronomic performance equivalent to the plants fertiliser with mineral counterparts, being particularly significant the effective use of phosphorus from the BBF.
5. The use exhausted mesolites to reduce ammonia emission was demonstrated to be a successful strategy, reaching nitrogen loss reduction up to 85%.
6. A LCC model was successfully developed to assess P-rich sludge (PS) valorisation through advanced composting.
7. The minimum plant size, estimated with the LCC model adapted for PS produced through SCENA technology, shows a feasible economic scenario (break-event point) only from 664,424 PE on. However, the cost associated for PS treatment cost showed values within the common

values found for conventional composting processes. Additionally, the expansion of the technology including also the SCENA technology leaded to positive economic performance in comparison to conventional WWTP.

8. In contrast, the advanced composting as the sludge valorisation solution in conventional EBPR systems estimated a more reasonable break-even point even when applying a sensitivity analysis dependent on the selling price of the BBF produced. Economically feasible scenarios would be possible from medium treatment capacity WWTPs between 25,880 and 51,578 PE for fair and lower than estimated fair selling price, demonstrating the competitivity of the solution proposed considering the current fertilising products market. Within this scenario and considering the current European WWTP infrastructure, 90% of the population served would be covered by this alternative. Moreover, this technological solution would be able to recover 155K tons of phosphorus, meeting 17.6% of the current phosphorus demand in Europe. Phosphorus recovery alternative would be cheaper (0.5-7.0 € kg^{-1}P) than other solutions based on crystallisation or extraction of phosphorus from ashes.
9. The solution proposed permits the alignment of the water sector with the European circular economy action plan, contributing to the end-of waste status of sludge, to develop a safe food chain, to balance the nutrient flows in European regions among others. The main threats found for the effective implementation of advanced composting technology would be associated mainly to the legal restrictions and limited access to the market of the BBF obtained. This kind of product can't compete with the prices of mineral fertilisers and the acceptance of consumers seems to be still limited. Overcoming legal barriers, further investment on the infrastructure of water sector and standardisation of the quality of recovered products would help to effectively achieve the European market.

FUTURE WORK:

- The control algorithm of biodrying is currently being upgraded towards a more automatised algorithm based on moisture evaporation models. The assessment of the overall and daily process efficiency of biodrying process with the new automatised control will be done by means of all the innovative indexes developed in the present thesis, demonstrating the accuracy and utility of the indexes developed.

- Biodrying performance indexes will be improved by incorporating environmental performance-related parameters, focusing mainly on CO2 eq. emissions.
- Upscaling of the biodrying technology for sludges and other organic wastes with high moisture content and low porosity is currently being assessed. The ultimate objective is the successful implementation of the biodrying technology in real scale.
- We are currently also working on the combustion process of the biomass fuel obtained through biodrying. Next steps will be focused on the combustion process of biodried product in conventional biomass boilers and the assessment of its efficiency and monitoring of combustion gases.
- Advanced composting will be improved by deepening in the solutions for the mitigation of ammonia emissions through the use of adsorbing media. The assessment of several adsorbing media and their dosing trials will be performed considering the technical, environmental (monitoring of polluting gaseous emissions) and economic dimensions of advanced composting leading to the identification of the best-performing solution.
- A deeper study of the agronomic performance of the BBF will be performed to assess mineralisation of nutrients, leaching risk of pollutants present in the BBF, etc., including also the BBFs obtained from the advanced composting trials amended with adsorbing media.
- A deep techno-economical comparison between biodrying and anaerobic digestion of sludges is foreseen to be made which is expected to give solid arguments towards the biodrying alternative in some contexts.
- Additionally, and although it was not specifically considered in this thesis, the centralised treatment of these feedstocks is foreseen to improve the economic indicators, since CAPEX would be shared among different WWTP facilities.
- The results obtained in this thesis are expected to be relevant in terms of overcoming policy constrains related to the resource recovery from sludges. We are already going beyond the results obtained here and contributing significantly in working groups dealing with sludge management strategies. In this context, are currently working together with the Catalan

Waste Agency and the Catalan Water Agency and being a key actor in the development of the sludge management plan of Catalonia that will cover a period from 2023 to 2033.

8. REFERENCES

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APPENDIX

LIST OF APPENDICES

APPENDIX I. Users guide od biodrying and advanced composting pilot plant in the framework of SMART Plant

BIODRYING & COMPOSTING PILOT PLANT

USER MANUAL

English document-v1

June 2018

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CONTENTS

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Tables

Figures

1. INTRODUCTION

The mobile pilot plant for biodrying and composting (named Downstream SMARTech B) has been designed to produce high-quality fertilisers by composting (according to Fertiliser Regulation 2003/2003 and Sewage Sludge uses in Agriculture (86/278/EEC)) and high-calorific potential resources from organic wastes by biodrying (accomplishing the Renewable Energy Directive (2001/77/EC). Both processes are stablished according to the waste management hierarchy defined in the Waste Framework Directive (2008/98/EC).

Figure 1 shows the schematic 3D view of the mobile pilot plant designed, constructed and commissioning in the BETA Tech. Center facilities. Both technologies (composting and biodrying) consist mainly of the following components: (i) reactor, (ii) scale, (iii) air compressor, (iv) monitoring devices, (v) control system, (vi) metallic structure and (vii) mixer. The monitoring and controlling equipment of each system is composed by an airflow meter, oxygen and carbon dioxide sensor, and temperature probe. Especially for biodrying, a scale is also placed under the reactor to know and monitoring the weight loss.

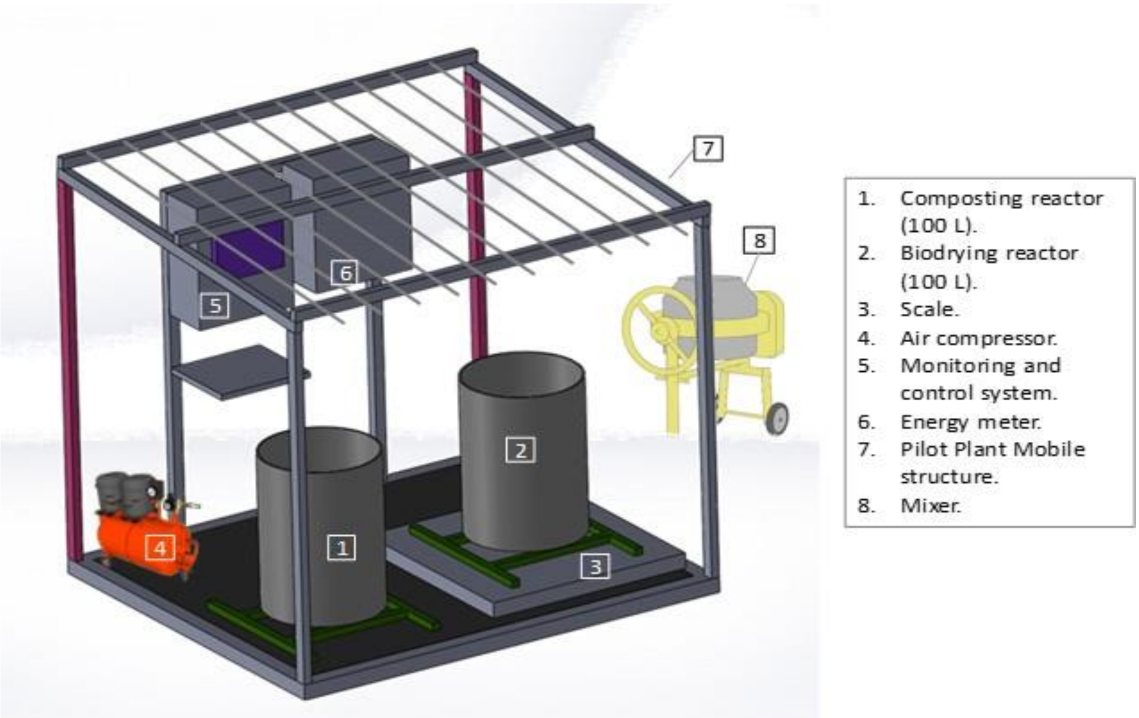


Figure 20 Schematic 3D view of the composting and biodrying pilot plant.

The operation of the pilot plant is fully automatic thanks to the development and implementation of an advanced system to data monitoring and control. This system has been implemented in combination with the Arduino hardware, as electrical signal input, and LabVIEW software (National Instruments). Moreover, the monitoring system can be operated remotely.

In the Table 1, a brief technical, functional and operational description of the main components of the pilot plant is detailed.

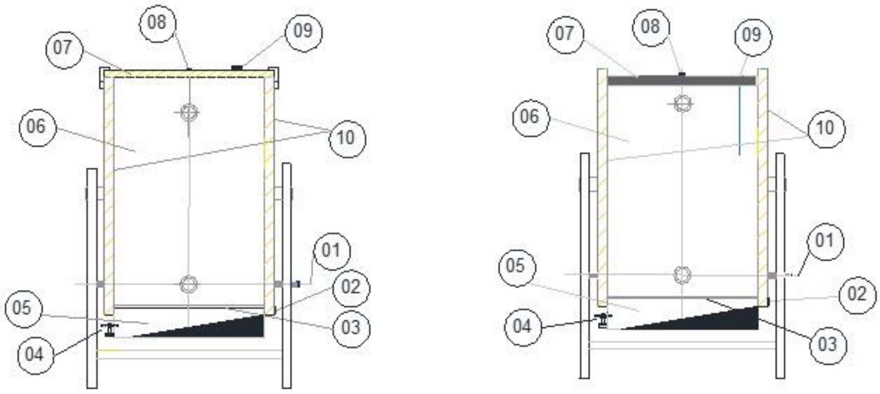
Table 14 Description of the main components of the Downstream SMARTech B.

Component	Technical Description
01-Composting reactor 02-Biodrying reactor	Both reactors have an operative volume of 100 L, each one. They have a cylindrical geometry and have been constructed with stainless steel (AISI 316L). The lateral walls are isolated with stone wool to avoid the loss of temperature and maintain adiabatic conditions.
03-Scale	The scale (max. 600 kg ± 0.1) sited under the biodrying reactor allows determining the mass reduction of the process. This mass reduction is an indirect measure of the drying rate.
04-Air compressor	The air compressor provides the air necessary to maintain the aerobic conditions in both processes (biodrying and composting). This air is supplied under the requirements to guarantee the optimum working conditions of the airflow meters (mainly 2.5 bar and 0% of moisture content).
05-Monitoring and control system	Both bioreactors are monitored and controlled simultaneously with the same hardware (Arduino ONE) and software (Labview 2017®, National Instruments). Each pilot plant is provided of: (i) an oxygen sensor, (ii) a carbon dioxide sensor, (iii) a temperature probe and (iv) an airflow meter. Specifically for the biodrying pilot reactor, a scale is also included. All data are monitored, registered and saved in a PC. Specific and advanced airflow regulation has been developed and implemented to optimise the processes keeping the minimum energy consumption. The pilot plant monitoring system can be visualized and controlled remotely by an on-line application.
06-Energy meter	All electric and electronic device are connected to the energy meter system to assess the specific and global energy consumption of the pilot plant.
07-Pilot plant structure	The mobile structure has been designed and builds specifically for this pilot plant. The structure has been made with stainless steel (AISI 316L) to support the external conditions (low and high temperatures depending on the season and a high presence of corrosive gasses like NH3 generated during the aerobic biodegradation of organic matter).

Component	Technical Description
08-Mixer system	An industrial mixer is included in the plant in order to facilitate the initial mixing of the sewage sludge with the complementary material (normally wood chips or pruning wastes). The sludge, with an initial moisture content between 70 and 80 % (expressed on wet basis), is mixed with this kind of material in order to (i) provide an enough material structure to guarantee the airflow circulation and (ii) reduce the high initial humidity content.

2. BIOREACTORS STRUCTURE

Two pilot bioreactors have been designed for biodrying and composting pilot plant. Both bioreactors have a similar structure and characteristics. Figure 2 shows a detail of their structure and characteristics. The bioreactors have a cylindrical geometry with an operative volume of 120L (height: 0.85m; diameter: 0.5 m). Both bioreactors have lateral walls thermally isolated with polyurethane foam (2 cm) and the internal and external surfaces are made with stainless steel. They operate in batch model regime. In order to facilitate the operations of charge and discharge, as well as the cleaning of the internal structure of the bioreactors, a horizontal rotatory axis (point 01) has been included. This axis is located 15 cm upper the bottom part of the bioreactors. At the bottom of the lateral wall, two orifices are situated. At right, a hole of 6 mm diameter (point 02) is located to supply the inlet airflow from airflow meter. At left side, a bigger hole (24 mm diameter) is placed for leachates removal (point 04). A perforated plate is fitted into the bottom (point 03). The function of the plate is to separate the bioreactor in two chambers: the bioreaction chamber (point 06) with a volume of 100 L and the air distribution and leachate accumulation chamber (point 05) with a volume of 20 L. The basis of the bottom chamber has a 45 degree slope to facilitate the leachates removal. On the top (point 07) of the bioreactors, an impermeable and adiabatic cover is placed for composting bioreactor and permeable cover is placed for biodrying reactor. The cover of composting is made with the same material of the reactor structure (stainless steel). Instead, the top cover of the biodrying reactor consists of a layer of 10 cm width made with straw. This cover allows maintaining adiabatic conditions inside the bioreaction chamber and guarantee the evaporation of the water. At the top cover, two more orifices are situated. One hole is placed at the centre to insert the temperature probe (point 08). The other orifice (point 09) is used to remove the exhaust gases in order to analyse the composition of the outgoing gases. Specifically for the biodrying reactor, the gasses generated during the process are suctioned through an external fan to generate a gas flow passing through the gasses meters, since it is not a totally closed bioreactor, as composting bioreactor.



COMPOSTING REACTOR		*BIODRYING REACTOR*	
01-Rotation axis	**06**-Bioreaction chamber	**01**-Rotation axis	**06**-Bioreaction chamber
02-Inlet airflow	**07**-Top cover	**02**-Inlet airflow	**07**- Permeable top cover
03-Perforated plate	**08**- Temperature hole	**03**-Perforated plate	**08**- Temperature hole
04-Leachate collector	**09**- Outgoing gases	**04**-Leachate collector	**09**- Outgoing gases
05-Air distribution chamber	**10**-Isolate sidewalls	**05**- Air distribution chamber	**10**-Isolate sidewalls

Figure 21 Detail and structure of the bioreactors.

Figure 3 shows a specific image of the each one of the above-described components.

01-Rotation axis		**06**-Bioreaction chamber	
02-Inlet airflow		**07**- Top cover	***Composting***  ***Biodrying*** 

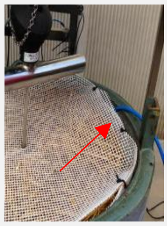
03-Perforated plate		**08**- Temperature probe hole	***Composting***  ***Biodrying*** 
04-Leachate collector		**09**-Outgoing gases	***Composting***  ***Biodrying*** 
05-Air distribution chamber		**10**-Isolate sidewalls	

Figure 22 Images of the specific components of the bioreactors.

3. LIST OF ELECTROMECHANICAL EQUIPMENT

In the Figure 4, the Piping and Instrumentation Diagram of the pilot plant is showed. In this scheme all manual and electromechanical compounds of the plant are identified.

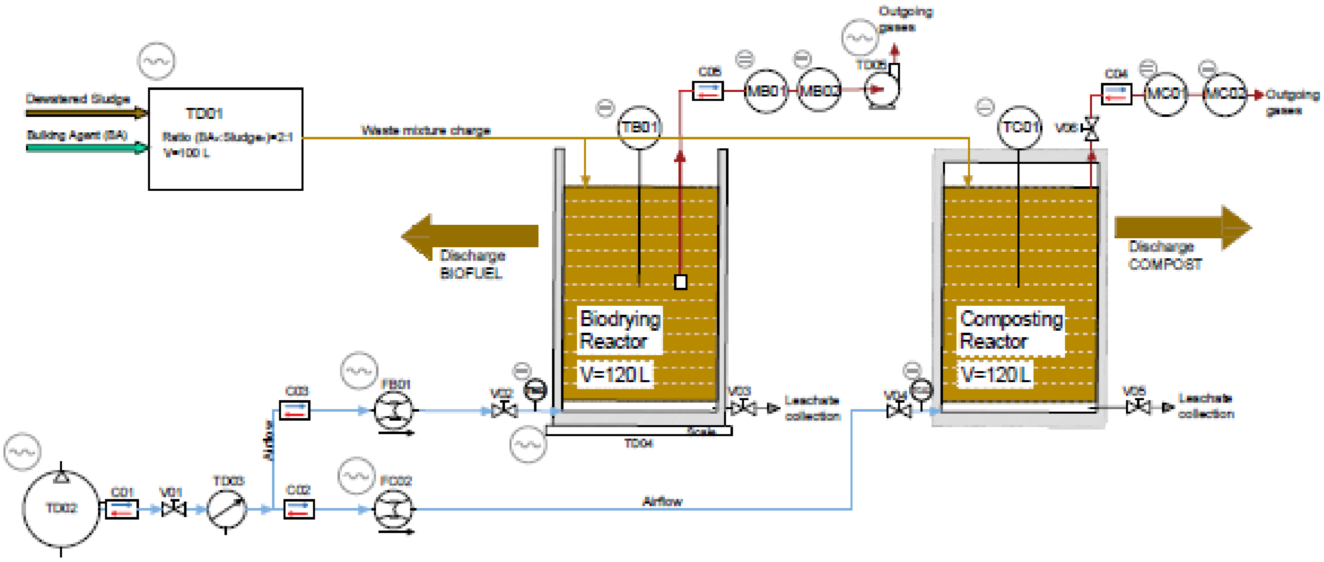


Figure 23 Piping and Instrumentation Diagram of Downstream Smartech B.

Table 2 presents the technical specifications of the electromechanical devices of the pilot plant, as weel as the manual accessories.

Table 15 List of electromechanical equipment and manual accessories of the Downstream Smartech B pilot plant.

MIXER UNIT					
CODE	**DESCRIPTION**	**MAKE/MODEL**	**UNIT**	**RANGE**	**INSTALLED POWER**
TD01	Rotation unit for sludge and bulking agent mixture (volume ratio 1:2)	PRO 90 PLUS	-	-	0.25 kW
COMPOSTING (C) & BIODRYING (B) REACTORS *List of electromechanical equipment included in both reactors*					
CODE	**DESCRIPTION**	**MAKE/MODEL**	**UNIT**	**RANGE**	**INSTALLED POWER**
TD02	Air compressor	DNX 2050	bar	0.1 / 5	1.5 kW
TC01 TB01	Pt100 for the measurement of the inlet airflow temperature	Isertech S.A/Pt100	ºC	-20 / 100	0.35 kW

TC02	Pt100 for the measurement of the material temperature (3 points at different heights).		°C	0 / 100	
TB02					
MC01	Oxygen content in the outgoing gases	Alphasense/O2 A2	%	0 / 21	
MB01					
MC02	Carbon Dioxide content in the outgoing gases	Alphasense/ IRC A1	%	0 / 20	
MB02					
FC01	Airflow meter	Bronkhorst/D-6311-DR	$L \cdot min^{-1}$	0.2 / 10	0.05 kW
FB01				0.4 / 20	0.05 kW
TD02	Suction fan	-	-	-	0.05 kW
TD03	Scale platform to measure the reduction of water during the biodrying process	Gram Precision / k3-k3i	kg	0.5 / 500	0.23 kW
Data acquisition, monitoring and control system					
TD05	Laptop	NUC/NUC6CAYH	-	-	0.36 kW
TD06	Data acquisition hardware	Arduino ONE	-	-	
TD07	Energy consumption meter	Wellness Smart	-	-	-
Manual Accessories					
V01	On/off valve: inlet airflow				
V02	On/off valve: leachate removal				
V03	On/off valve: inlet airflow				
V04	On/off valve: leachate removal				
V05	On/off valve: outgoing gases				
C01	Water condensation chamber of the air compressor.				
C02	Water condensation chamber before composting airflow meter.				
C03	Water condensation chamber before biodrying airflow meter.				
C04	Water condensation chamber before composting gasses meters.				
C05	Water condensation chamber before biodrying gasses meters.				

4. WORKING PROCEDURE

The pilot plant of composting and biodrying works in batch cycles. For composting, there is a residential time of 2 weeks inside the reactor (decomposition phase). And then the material goes to a maturation phase (outside) extended during 8 additional weeks in a dynamic windrow (manually operated). Instead, for biodrying the process time in the reactor depends on the initial moisture content of the material and, at least, is extended during 2 or 3 weeks.

The detail of the working protocol of the pilot plant is summarised as follows (more information can be found in the working manual of the plant). The steps to operate the pilot plant are to:

- **Switch on the mechanic and electronic devices**: switch on the air compressor (TD02), open the valve (V01) and regulate the relative pressure up to 2.5 bar (TD03). Switch on the laptop, the weight scale (TD03) and the electrical panel. The weight scale must be tare to know the mass of the material processed and monitoring the loss of the mass along the process.
- **Remove the water content in the water traps**. There are five water traps in the pilot plant. Three of them (C01, C02, C03) are placed to eliminate the water content of the air before the mass airflow meters. The other two are placed before oxygen and carbon dioxide sensors (C04, C05) in order to condensate the water content of the outgoing gases and avoid any damage to the sensors.
- **Open the monitoring and controlling program.** Composting Pilot Plant.ni and Biodrying Pilot Plant.ni; LabVIEW 2017). See section 5.
- **Check that all devices operate adequately**. Verify the oxygen and carbon dioxide sensors with the span gas. Test the communication between the monitoring and controlling software and the electronic devices (sensors, temperature probes and airflow meters and controllers).
- **Preparation of the solid organic waste to treat**: mix (TD1) the adequate ratio “sludge/bulking agent” to provide the enough Free Air Space (FAS) in the material and guarantee aerobic conditions. Normally, at this process scale, the most appropriate volume ratio is 1:2 (sludge/bulking agent).
- **Insert the mixture inside the specific bioreactor (C, B).** The mass of the waste inserted must be quantified (TD04).
- **Close the top cover of the reactor.** The cover is impermeable for composting and permeable for biodrying.
- **Insert the temperature probe** (TC01; TB01) **in the corresponding hole of the top cover** *(Figure 3; point 8)*.
- **Connect the outgoing gases hole** *(Figure 2; point 9)* **on the top cover to the sensors.** In both reactors, there is a condensation chamber (C04, C05) before the sensors. Specifically for biodrying, the top cover is permeable and therefore, the bioreactor is not totally closed.

Because of this, after the sensors a suction fan (TD05) is located to generate a gas flow passing through the sensors.

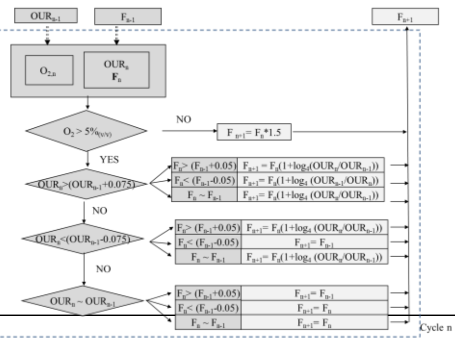
- **Close the leachate removal valve** (V03, V05).
- **Connect the output of the airflow meter to the inlet point placed at the bottom of the reactor** *(Figure 2; point 2)***.** Inlet air can be pre-heated if necessary. To do so, switch on the electrical resistances. The temperature probe (TC02, TB02) monitors the temperature of the inlet airflow.
- **Start the monitoring and controlling software after selecting the most appropriate airflow controller system.** Different controllers for airflow regulation have been implemented in the system: manual, cyclic controller, control depending on the oxygen content or temperature evolution, and advanced controllers based on the biological activity measurement evolution. The later have been specifically developed in this project.

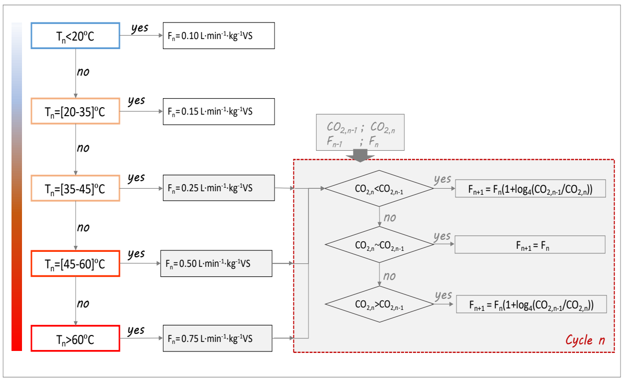
Although the pilot plant is automatized and can be remotely controlled and operated, it is recommended to do some in-situ periodic measurements and verifications to guarantee the normal performance of the entering plant.

5. MONITORING AND CONTROLLING SYSTEM

In the desktop of the PC thye biodrying and composting sofwares are located. There are two different softwares for biodrying and other two for composting. In the Table 3 the specifications of the softwres are detailled.

Table 16 Specifications of the biodrying and composting monitoring and controlling systems availables in the pilot plant.

Name of the file	Specifications
- Composting Pilot Plant.vi - Biodrying_Pilot Plant.vi	They include **cyclic and manual aeration system**, and a constant monitoring of oxygen, carbon dioxide, temperature and airflow.
OUR_Composting.vi	It includes an especific and smart aeration control system depending on the biological activity rate of the process measured as Oxygen Uptake Rate (OUR). 

T_Biodrying.vi	It includes an especific and smart aeration control system depending on the temperature of the material and CO_2 emissions. 

5.1. BIODRYING AND COMPOSTING SOFTWARE INTERFACE

The interface of the different software implemented in the pilot plant are quite similar. Figure 5 shows the image of the general front panel where the main controllers are identified.

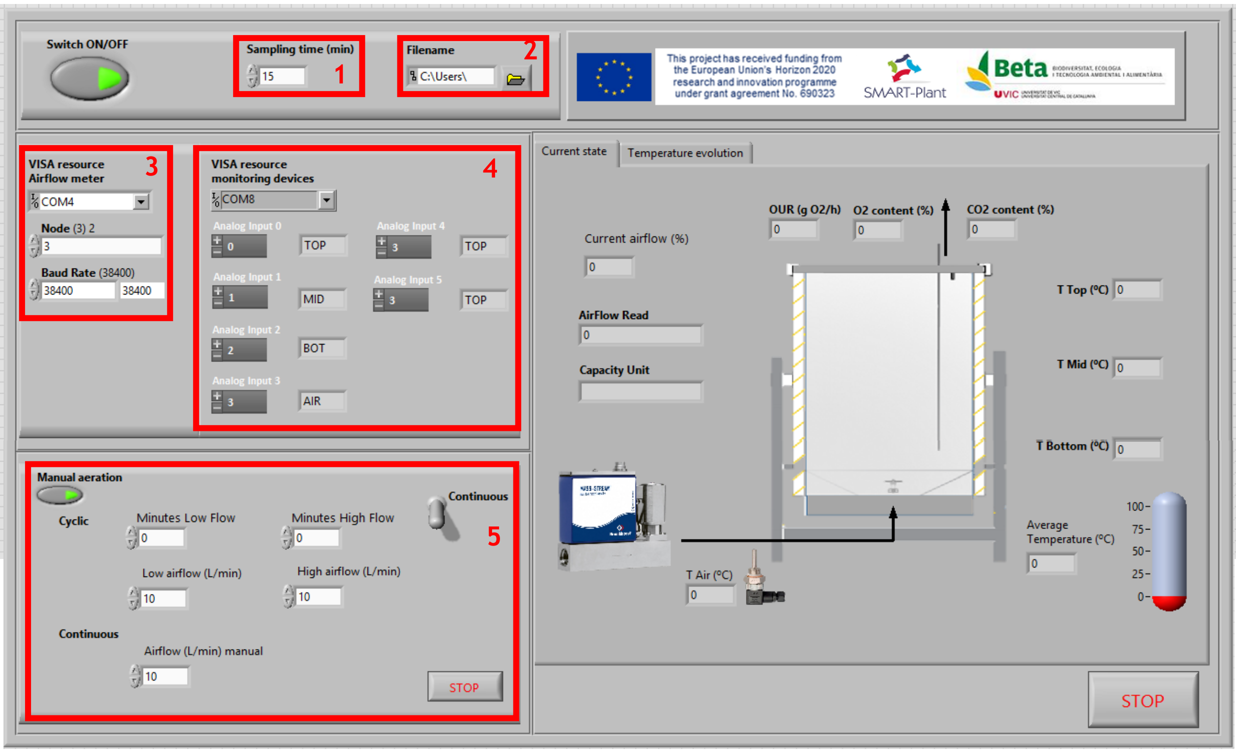


Figure 24 General front panel of the monitoring and controlling system for biodrying and composting pilot plant.

Next, the different items available in this front panel are described:

(1)	Sampling time definition (in minutes)
(2)	Excel file for data saving.
(3)	Airflow meter communication parameters (only change the COM port, if necessary)
(4)	Parameters for data acquisition communication (from Arduino UNO). Modification of COM port and analogic pin, if necessary.
(5)	Aeration strategies specifications.

Before starting general switch should be switched on (in green).

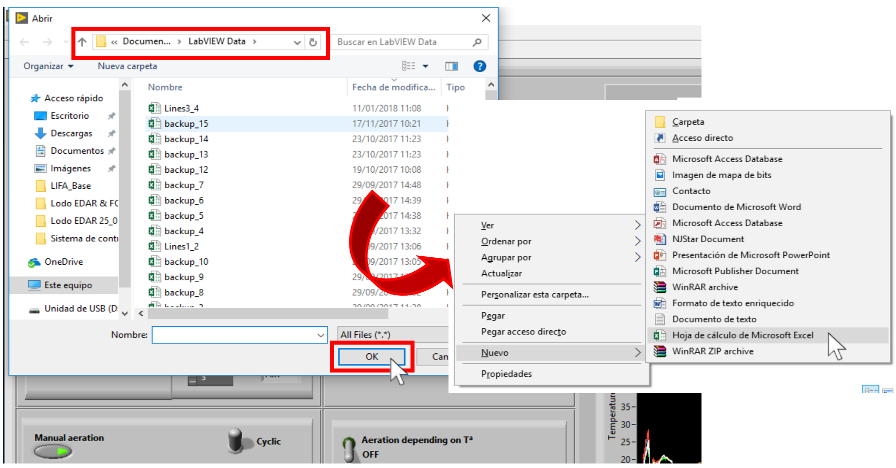
1. Sampling time selection.

The time period between data collection times can be modified using either up and down arrows disposed on software or by writing directly the time of interest (in minutes).

2. Exel filename and location selection.

Filename and its location in computer can be modified. To do so, user must click in folder image. Selection panel will be opened where user can select the location of data acquisition file, in .xlsx format. Software will not create the file and it must be created by clicking the right button of the mouse. Filename of the file created should be changed and click on OK button to accept changes.

Figure 25 Saving data indications.



3. <u>Airflow controller selection</u>.

COM port of airflow controller must be selected from a number of available ports. Neither node for data connection or baud rate should be changed.

- **COM 4** for 0,4- 20 L/min airflow controller
- **COM 9** for 2- 100 L/min airflow controller

4. <u>Acquisition data & analogic pin selection.</u>

COM location of acquisition system card (Arduino UNO) must be selected. Also, if you have done any electrical connection modification, it is necessary to verify the correct selection of each analogic.

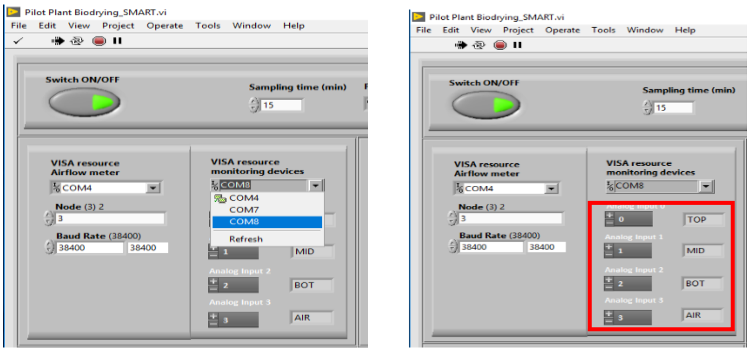
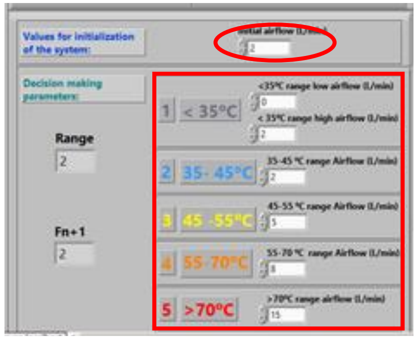
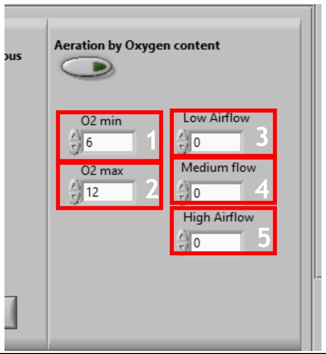
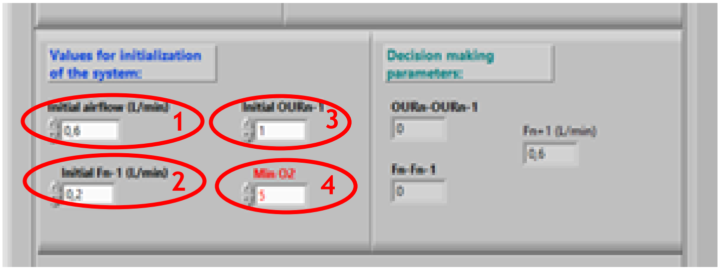


Figure 26 Arduino signals.

5. <u>Aeration strategies</u>

Strategy	Process	Specifications
Manual - Continuous - Cyclic	☒ Biodrying ☒ Composting	Manual aeration must be activated by clicking the button. Either continuous or cyclic aeration can be selected by moving the toggle switch up (cyclic) or down (continuous). **<u>Continuous</u>** aeration mode permits the selection of the aeration level of interest on L/min, either by clicking up and down arrow or by writing directly the interested value. For **<u>cyclic aeration mode</u>**, minutes for low aeration level and minutes for high aeration level should be selected by clicking on the arrows or writing the interested value directly on the space. 
Temperature aeration system	☒ Biodrying ☐ Composting	

Strategy	Process	Specifications
		Software will adapt aeration level depending on the range of temperature system (< 35ºC, 35-45ºC, 45-55ºC, 55-70ºC, > 70ºC). Software disposes of a number of algorithms for airflow level optimization (see the section T_Biodrying.vi on the table 3.). Based on the value of temperature in the middle of the reactor and CO_2 content in the exhaust gases, system will decide whether rise airflow rate or not.  For initialization, user must provide an airflow level for each range specified. For the first temperature range (< 35ºC), cyclic aeration was defined, where low aeration level will be set in the first 3 cycles, and high aeration level the next two cycles.
Oxygen aeration system	☐ Biodrying ☒ Composting	Low airflow (3) will be activated while oxygen content in exhaust gas is lower than the minimum specified on the O_2 min box (1). Similarly, high aeration (5) will be activated while O_2 content in exhaust air of the system is higher than the specified on the O_2 max box (2). While temperature of the system is between minimum and maximum temperature values specified, then medium aeration level (4) will be activated. 
OUR controller	☐ Biodrying ☒ Composting	Software is capable of calculating Oxygen Uptake Rate (OUR) of the system based on the oxygen content of the exhaust gas and the known inlet airflow level.

Strategy	Process	Specifications
		Software disposes of a number of algorithms for airflow level optimization (see the OUR_Composting.vi section on the table 3.). Based on the value of OUR, system will decide whether rise airflow rate or not. For the initialisation of system initial airflow (Fn; 1) and airflow from the previous cycle (Fn-1; 2) should be provided. Also, initial OUR from the previous cycle should be facilitated in order to avoid unable division by 0 (3). Finally, an alarm system has been included where, in case oxygen content is lower than a given minimum value (in %; 4), system will rise aeration level. It has to be taken into account that every time software is stopped to reinitialize it, initial values for previous cycles (OURn-1 and Fn-1) have to be adapted according to estimated expected values. If values for starting up are maintained excessive aeration will be set for next cycles. 

6. Current state screen.

This screen will show the update of last data taken: current airflow level, temperature, inlet air temperature and average temperature displayed in either numeric. The evolution of these parameters can also view on the graph (sheet evolution).

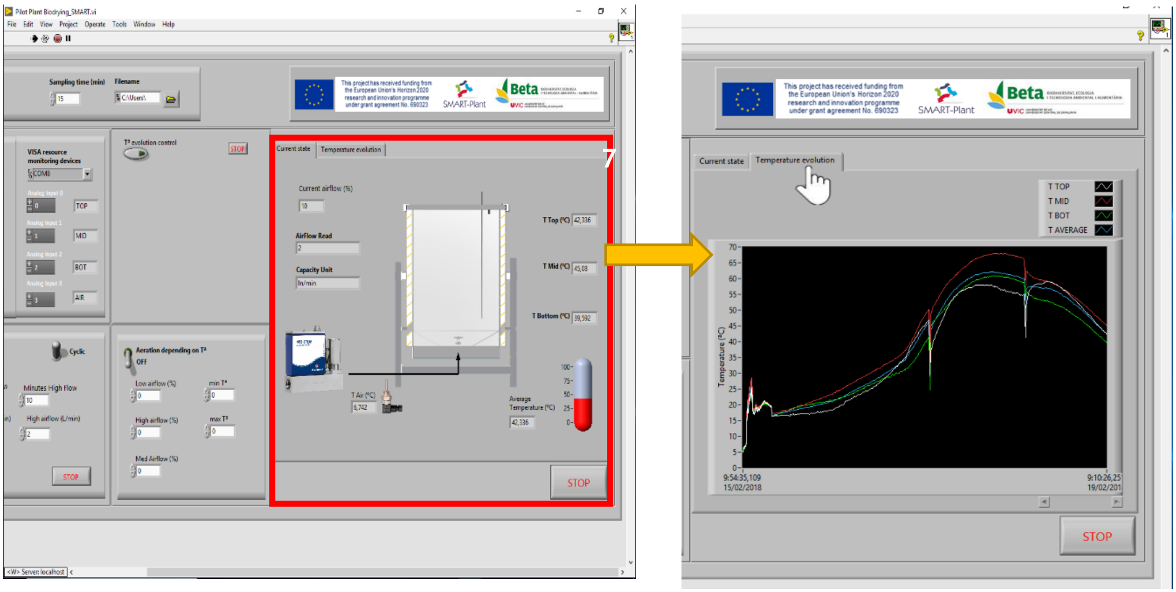
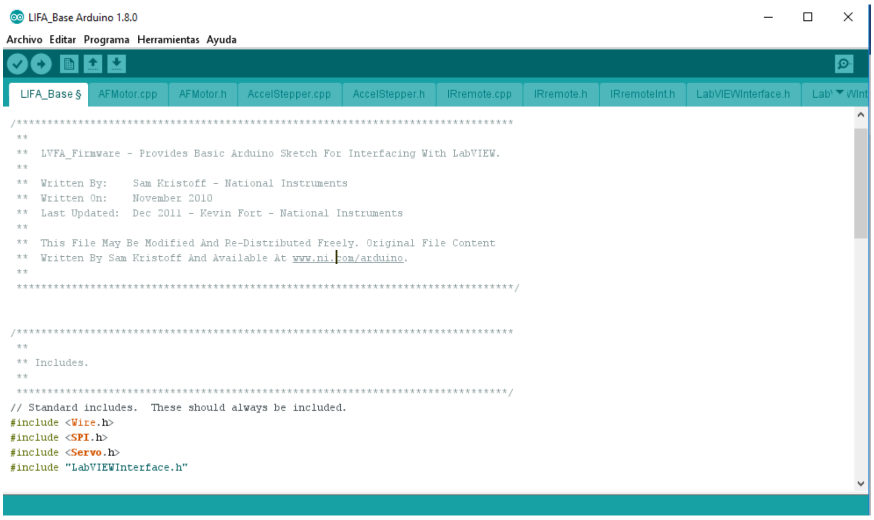


Figure 27 Monitoring screen.

5.2. LABVIEW – ARDUINO COMMUNICATION

In order to be able to make the communication between Arduino data acquisition system (VISA ports) and Labview software, Labview interface should be uploaded into Arduino board (only supported by 1.8.0). LIFA_base is the interface of Arduino, code can be consulted in the annex 1 of the present document. VI package Manager provides the communication between Arduino and Labview.



6. REMOTE MONITORING AND CONTROLLING

	UVIC PCs	Individual PCs / UVIC PCs (outside UVIC)
Software required	Professional LabView or LVRTE2017	
Internet	Local UVIC network: uvic.local	VPN (FortiClient-*username/password UVIC*)
Web browser	Internet Explorer	
Address	**Composting pilot plant** http://17-0025.uvic.local:8000/Pilot%20Plant%20Composting_SMART.html	**Composting pilot plant** http://10.3.4.204:8000/Pilot%20Plant%20Composting_SMART.html *Control OUR*: http://10.3.4.204:8000/Pilot%20Plant%20Composting_SMART_Control%20OUR.html
	Biodrying pilot plant http://17-0025.uvic.local:8000/Pilot%20Plant%20Biodrying_SMART.html	**Biodrying pilot plant** http://10.3.4.204:8000/Pilot%20Plant%20Biodrying_SMART.html

7. ENERGY MONITORING SYSTEM

Energy consumption of the plant is online monitored in real-time. Energy consuming devices are by-passed to analyse their energy consumption using a number of universal monitoring devices and collected data can be online consulted.

Figure 28 Image of the energy monitoring system.

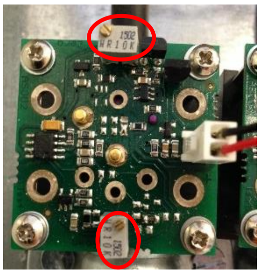
The different components of the energy monitoring system are identified below:

1 Power supply for energy monitoring
2 General power supply for Biodrying & Composting Plant
3 Biodrying monitoring devices
4 Biodrying air-flow controller
5 Composting monitoring devices
6 Composting air-flow controller
7 Online data remote connection

8. DEVICES SETTING UP AND MAINTENANCE

Before getting ready experimental mixtures, it is necessary to evaluate the proper running of all the devices of pilot plant (computer, temperature probes, gas sensors, airflow controllers, etc.).

Also, all water traps should be emptied and their progress during experimentation followed, and emptied again if necessary.



Gas sensors calibration must be verified before each experimentation using a span gas and re-calibrated if necessary. Oxygen sensors calibration is done through adjustment of potentiometer.

Calibration of CO_2 sensor should be done by manufacturer.

APPENDIX II. Calculation method for FI and SI

Table II-17. Scoring values (SV_i) and weighing factors (WF_i) used to determine fertility index (FI) and Safety Index (SI) of BF

	Score value (SV_i)						Weighing factor (WF_i)
Fertility parameters for FI	5	4	3	2	1	0	
Total Organic C (%, dm)	> 30	30- 25	25-20	20-15	15-10	< 10	5
Total N (%, dm)	> 5	5-3.5	3.5-2.5	2.5-1.5	1.5-0.5	< 0.5	3
Total P (%, dm)	>3	3-2.4	2.4-1.4	1.4-0.6	0.6-0.3	< 0.3	3
Total K (%, dm)	> 3	3- 2.4	2.4-1.5	1.4-0.7	0.6-0.4	< 0.4	2
C:N	< 4	4-7	7-10	10-14	14-20	> 20	4
Respiration activity (gO_2 $kgVS^{-1}h^{-1}$)	< 0.5	0.5-1	1-1.5	1.5-3	3-5	>5	4

	Score value (SV_i)						Weighing factor (WF_i)
Safety parameters for SI	5	4	3	2	1	0	
Cd (mg $kgTS^{-1}$)	< 0.3	0.3-0.7	0.7- 1.5	1.5-2	2-3	> 3	5
Cu (mg $kgTS^{-1}$)	< 35	35-70	70-150	150-300	300-400	>400	2
Ni (mg $kgTS^{-1}$)	<12	12-25	25-50	50-90	90-100	>100	1
Pb (mg $kgTS^{-1}$)	<22	22-45	45-100	100-150	150-200	>200	3
Zn (mg $kgTS^{-1}$)	<100	100-200	200-300	300-500	500-1000	>1000	1
Hg (mg $kgTS^{-1}$)	<0.2	0.2-0.4	0.4-1	1-1.5	1.5-2.5	>2.5	5
Cr (mg $kgTS^{-1}$)	<36	36-70	70-150	150-250	250-300	>300	3
Sanitisation degree	≥ 70ºC for 3d and either ≥65ºC for 5d or; ≥ 60ºC for 7d or; ≥ 55ºC for 14d					The rest not complying with sanitation requirement	5
PAH_{16}* (mg $kgTS^{-1}$)	<2	2-3	3-4	4-6	6-10	>10	3

Table II-18. Calculation of Fertility Index of the BFs obtained

Fertility parameters for FI	Si				Wi
Item	**Batch 3**	**Batch 4**	**Batch 5**	**Batch 6**	
Total Organic C (%, dm)	5	5	5	3	5
Total N (%, dm)	3	4	3	4	3
Total P (%, dm)	5	5	5	3	3
Total K (%, dm)	2	3	2	5	2
C:N	3	3	3	3	4
Respiration activity (gO_2 $kgVS^{-1}h^{-1}$)	4	4	5	4	4
FI value	**3,86**	**4,10**	**4,05**	**3,52**	

Table II-19 Calculation of Safety Index of the representative BF obtained

Safety parameters for SI		
Item	Si	Wi
Cd (mg/kgTS)	5	5
Cu(mg/kgTS)	4	2
Ni (mg/kgTS)	5	1
Pb (mg/kgTS)	5	3
Zn (mg(kgTS)	3	1
Hg (mg/kgTS)	3	5
Cr (mg/kgTS)	5	3
Sanitisation degree	0	5
PAH_{16}* (mg $kgTS^{-1}$)	5	1
SI value	**3,5**	

APPENDIX III. Detailed methodology for the calculation of economic parameters for LCC models

Table III-20. Sludge characteristics considered in the LCC models developed

Parameter	Unit	CS	PRS	PS (Sidetream SCENA)	PS (mainstream EBPR)
Moisture content	% (fb)	70%	81%	83%	80%
Volatile solids	% (db)	90%	79%	75%	75%
LHV of pelletised product	MJ kg^{-1}	13.58	15.57		
N content in sludge (TKN)	% (db)			7%	6%
P content in sludge (TP)	% (db)			4.2%	3.5%
K content in sludge (TK)	% (db)			0.6%	0.6%

Table III-21. Process performance and design parameters considered in the LCC models developed

Parameter	Unit	Biodrying	Advanced composting
Sludge to B.A mixing ratio	(v:v)	2 to 1	1 to 2
H2O removal	t H2O evap. t-1VSfeed	1.9 for CS biodrying and 2.7 for PRS biodrying (as moisture content varies, free water is assumed also to be higher in PRS)	4.88
VS consumption	t VS cons. t-1 VS feed	0.15	0.55
N volatilisation	t N lost t-1VSfeed		0.05
Maximum specific air suplly	m3 min-1t-1VS	3.21 for CS biodrying and 3.47 for PRS biodrying	3.21
Average specific air consumption	m3 h-1t-1VS	39.17 for Cs biodrying and 135.2 for PRS biodrying	38.75
TRH per windrow	days	12	20 in active degradation phase in windrows
B.A recovery	%	95% (+ sensitivity analysis between 80-95%)	95%
MC targeted for pelleting	%	12%	

Table III-22. Economic parameters assumed for the estimation of investment costs.

Item	Unit	Cost
Concrete	€/m3	120
Insulation material	€/m3	128.8
Cover material	€/m2	40.13
Stainless steel	€/kg	2.76
Stainless steel density	kg/m3	7700
Other auxiliary equipment: piping, valves, instrumentation, etc.	€	A cost 40% of windrows construction cost for small-capacity plants (construction cost below 10,000€), 30% of windrow construction cost for medium-capacity plants (between 10,000€ and 100,000€ of construction cost); 15% of windrow construction cost for big capacity plants (between 100,000 and 500,000€ of construction cost) and 5% of construction cost in very high-capacity plants (above 500,000€ of construction cost).
Blower unitary cost	€	Forced aeration assumed with a unique blower considered per each windrow. The working capacity of the blower estimated by the maximum specific airflow required for the system: 1600€ for maximum blower capacity required of 2000 m3/h, 3700€ for blower capacities between 2000 and 4000 m3/h and 4500€ for blower capacities above 4500m3/h. Additionally, replacement of all the blowers installed every 6 years was assumed.

Table III-3 cont.

Mixing truck	€	Acquisition of a mixing truck whenever the yearly mixture to treat exceeded 15,000t, an additional mixing truck every 15,000t. When sludge mass was lower, rental of a mixing truck was assumed. The cost assumed for a mixing truck was 50,000€ and replacement of trucks every 20 years was assumed.
Biomass boiler	€	Cost of biomass boiler was calculated with considering the linear regression parameters built according to the boiler prices found in the market upon the output sieved mass to be pelletised to dry the excess of moisture content. (biomass boiler price = 7258.3* Ln(output mass to be pelletised-pellets to be sold)+1726.8)
Civil works	€	Fixed 9,560€ and 15,000€ civil works cost for small and medium capacity plants (<100,000€) summed to an additional 50% or 20% of partial construction costs. This fixed cost was reduced to 10,000€ for big and very big treatment capacity plants summed to an additional 15% or 5% of partial construction cost.
Engineering services	€	Fixed 7,000€ and 10,000€ engineering services cost for small and medium capacity plants (<100,000€) summed to an additional 10% or 7.5% of partial construction costs. This fixed cost was reduced to 10,000€ for big and very big treatment capacity plants summed to an additional 15% or 5% of partial construction cost.

Table III-23. Economic parameters assumed for the estimation of operation and maintenance costs

OPEX parameter	Unit	Value
Bulking Agent cost	€/t	60[1]
Energy buying price	€/kwh	0.17
Diesel price	€/L	1.25
Bulking agent required	t/y	Volumetric mixing ratio of 1 to 1.5 was assumed (to achieve an initial MC of 54%). At the end of the process, 90% of B.A recovery was assumed.
Mixing truck rental	€/y	Dedication of a mixing truck was estimated following the method suggested by Ruggieri et al., 2009. Therefore, according to the mixture volume to treat, mixture mixing days were estimated and then, truck rental cost calculated.
Average diesel consumption	L/t mixture	0.6[2] for biodrying and 9[3] for composting (only during curing stage)
Pelleting cost	€/t product	Amount of final sieved material estimated by experimental mass balance used to calculated pelleting cost[4]. Data given in used as a reference for the calculation of the cost.
Personnel cost	€/y	Operator requirement was estimated assuming one whole working to treat 6000t sludge per year. Using the operator requirement, the model calculates an entire or half of the working time of an operator for plants with a minimum treating capacity of 1800t sludge per year. For smaller plant capacity, the model assumes the working hours of an already hired operator for the management of biodrying or composting plant. Salary assumed for operator was of 46000€/y.
Maintenance	€/y	2.5% or 2.2% of the total CAPEX added to a base cost of 865€ for plants with overall CAPEX costs < 100,000€ and between 100,000€ and 1,000,000€, respectively. 2.0% of the total CAPEX for plants with overall CAPEX > 1,000,000€.
Insurance	€/y	0.75% or 0.55% of the total CAPEX added to a base cost of 865€ for plants with overall CAPEX costs < 100,000€ and between 100,000€ and 1,000,000€, respectively. 0.45% of the total CAPEX for plants with overall CAPEX > 1,000,000€.

[1]Value provided by the local composting plant (Aïgues de Manresa);[2] Psaltis & Komilis, 2019; [3]Colón et al., 2012; [4]Vavrová et al., 2018

Table III.24. Nutrient substitution prices for fertilizer market price estimation

Nutrient	Substitution prices
Nitrogen	0.86$/kg N substituted
Phosphorus	1.34$/kg P substituted
Potassium	0.24 $/kg K substituted

APPENDIX IV. List of emerging pollutants measured in BBF

Table IV-25. Analysed polycyclic aromatic hydrocarbons (PAH 16) through QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) method

Acenaphthene	Benzo(a)pyrene	Chrysene	Indene(1,2,3-c,d)pyrene
Acenaphthylene	Benzo(b)fluoranthene	Dibenzo(a,h)anthracene	Naphthalene
Anthracene	Benzo(g,h,i)perylene	Fluoranthene	Phenanthrene
Benzo(a)anthracene	Benzo(k)fluoranthene	Fluorene	Pyrene

Table IV-26. Analysed pesticides extracted through QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) method plus purification step and analysis in GC.

Aclonifen	Dichlobenil	Mepanipyrim
Acrinathrin	Dichlofluanid	Metalaxyl
Alachlor	Dichlorvos	Methidathion
Aldrin	Dicofol	Methoxychlor
Ametryn	Dieldrin	Metolachlor
Atrazine	Difenoconazol	Metribuzin
Azinphos-methyl	Dimethoate	Mirex
Benalaxyl	Disulfoton	Nonachlor
Bifenox	Endosulfan I	Oxadixyl
Bifenthrin	Endosulfan II	Oxyfluorfen
Bromopropylate	Endosulfan Sulphate	Paclobutrazol
Captafol	Endrin	Parathion-ethyl
Captan	Epoxiconazole	Parathion-methyl
Carbaryl	Ethion	Penconazol
Chlordane-cis	Fenamiphos	Pendimethalin
Chlordane-trans	Fenarimol	Permethrin
Chlordecone	Fenhexamide	Phorate
Chlorothalonil	Fenitrothion	Piperonylbutoxide
Chlorphenvinfos	Fenvalerate + Esfenvalerate	Procymidone
Chlorpyrifos	Fipronil	Prometryn
Chlorpyrifos-methyl	Flusilazole	Propargite
Chlozolinate	Folpet	Propazine
Cyanazine	HCH-a	Propiconazole
Cypermethrin	HCH-b	Pyridaben
Cyphenothrin	HCH-d	Pyrimethanil
Cyproconazole	HCH-gamma (Lindane)	Pyrimiphos-methyl
Cyprodinil	Heptachlor	Quinalphos
DDD-2,4'	Heptachlor epoxide trans	Quinoxyfen
DDD-4,4'	Hexachlorobenzene	Sebutylazine
DDE-2,4'	Hexaconazole	Simazine
DDE-4,4'	Iprodione	Tebufenpyrad
DDT-2,4'	Irgarol1051 (Cibutryn)	Terbuthylazine
DDT-4,4'	Isodrin	Terbutryn
Deltamethrin	Kresoxim- methyl	Tetraconazole
Desethylatrazine	Lambda-Cyhalothrin	Trifluralin
Diazinon	Malathion	Vinclozolin

Table IV-27. List of contaminants of emerging concern analysed in BF after QuEChERS (Quick, Easy, Cheap, Effective, Rugged and Safe) extraction method with the subsequent analysis through high performance liquid chromatography coupled to tandem mass spectrometry (HPLC-MS/MS). The list of compounds was selected according to the EU watch list (EC; 2018). The table shows the compound, their category (antibiotics, pesticides, insecticides or hormones) and their limit of detection (LOD) and limit of quantification limits (LOQ).

Compound	Class	LOD	LOQ
		µg kg $^{-1}$	µg kg $^{-1}$
Erythromycin	Macrolides (Antibiotics)		
Clarithromycin	Macrolides (Antibiotics)	0.3	1.7
Azithromycin	Macrolides (Antibiotics)	1.4	4.6
Ciprofloxacin	Fluoroquinolones (Antibiotics)	7.9	26.5
Thiacloprid	Neonicotinoids (insecticides)	0.5	1.8
Imidacloprid	Neonicotinoids (insecticides)	1.9	6.5
Thiametoxam	Neonicotinoids (insecticides)	1.6	5.5
Metaflumizone	Semicarbazone (insecticides)		
Clothianidin	Neonicotinoids (insecticides)	2.4	7.9
Acetamiprid	Neonicotinoids (insecticides)	0.8	2.6
Methiocarb	Carbamates (pesticides)	0.6	2.1
Estrone	Hormones	1.3	4.2
17-β estradiol	Hormones	11.4	37.9
17-α ethinyl estradiol	Hormones	21.4	71.4

APPENDIX V. Detailed gaseous emissions monitoring results from biodrying of CS and advanced composting of PS

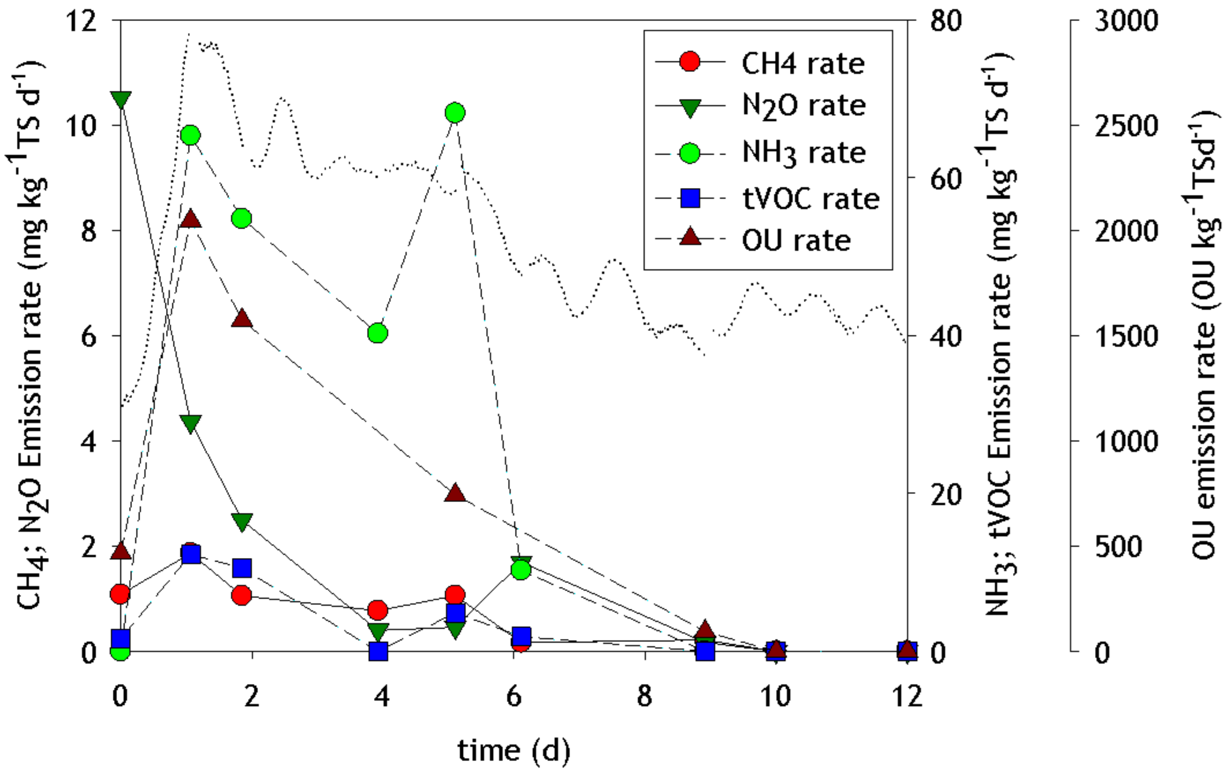


Figure V-29 Relevant gaseous emission profile in the worst case-scenario of biodrying of CS.

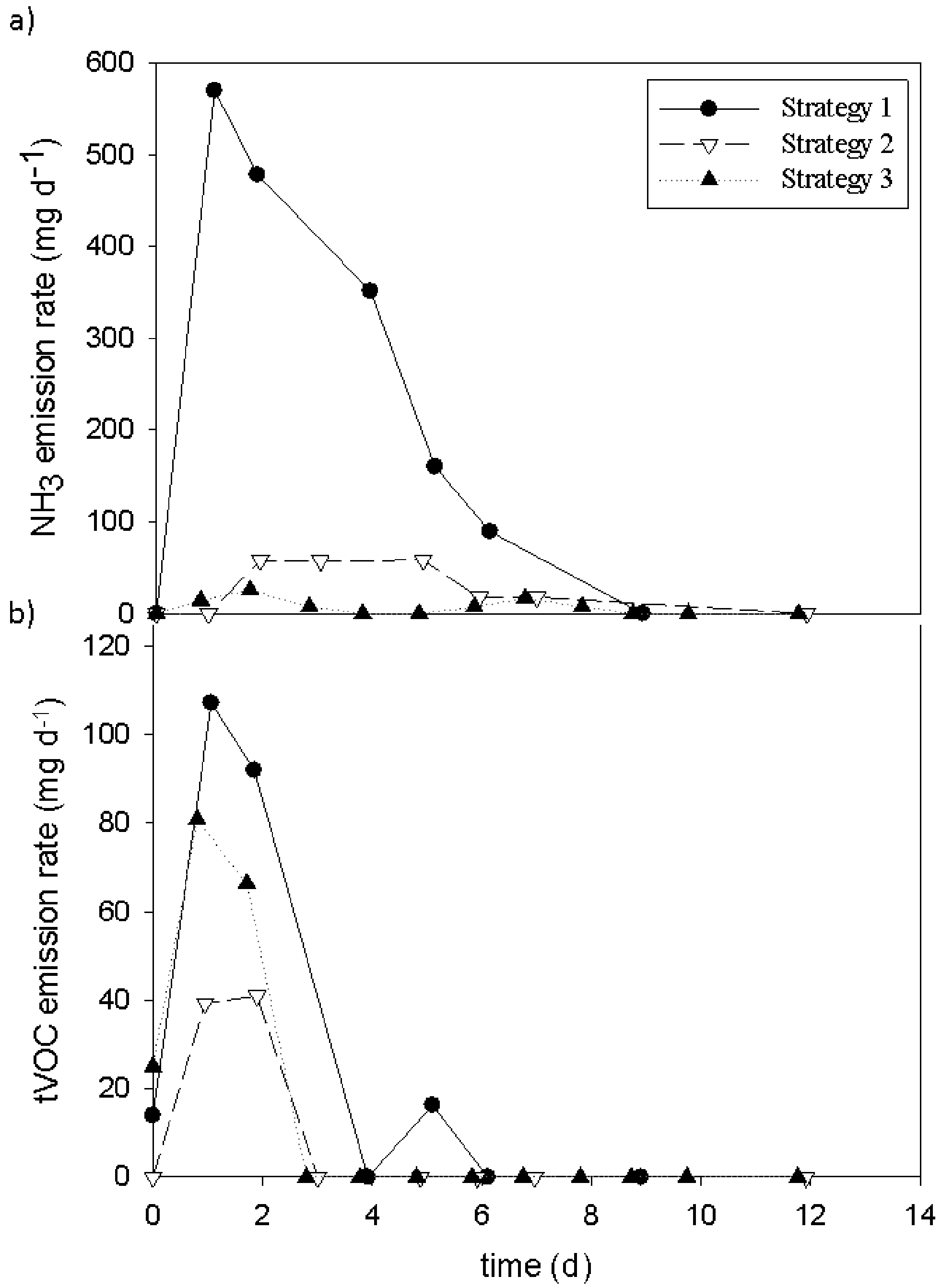


Figure V- 30 Comparison of NH_3 and VOCs emission profiles within biodrying processes of CS when applying different aeration strategies.

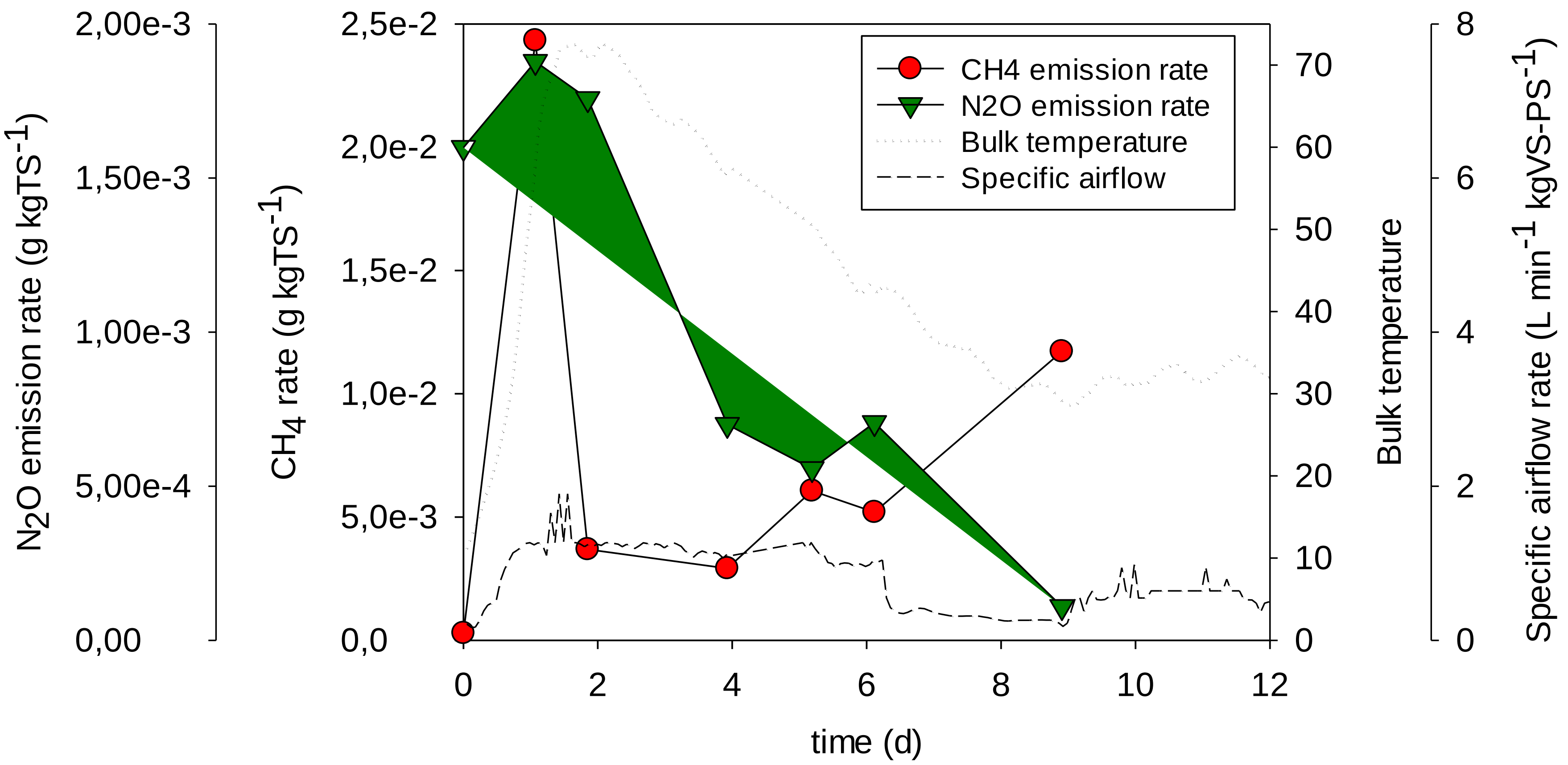


Figure V-31 GHG emission profile in a representative advanced composting trial of PS.

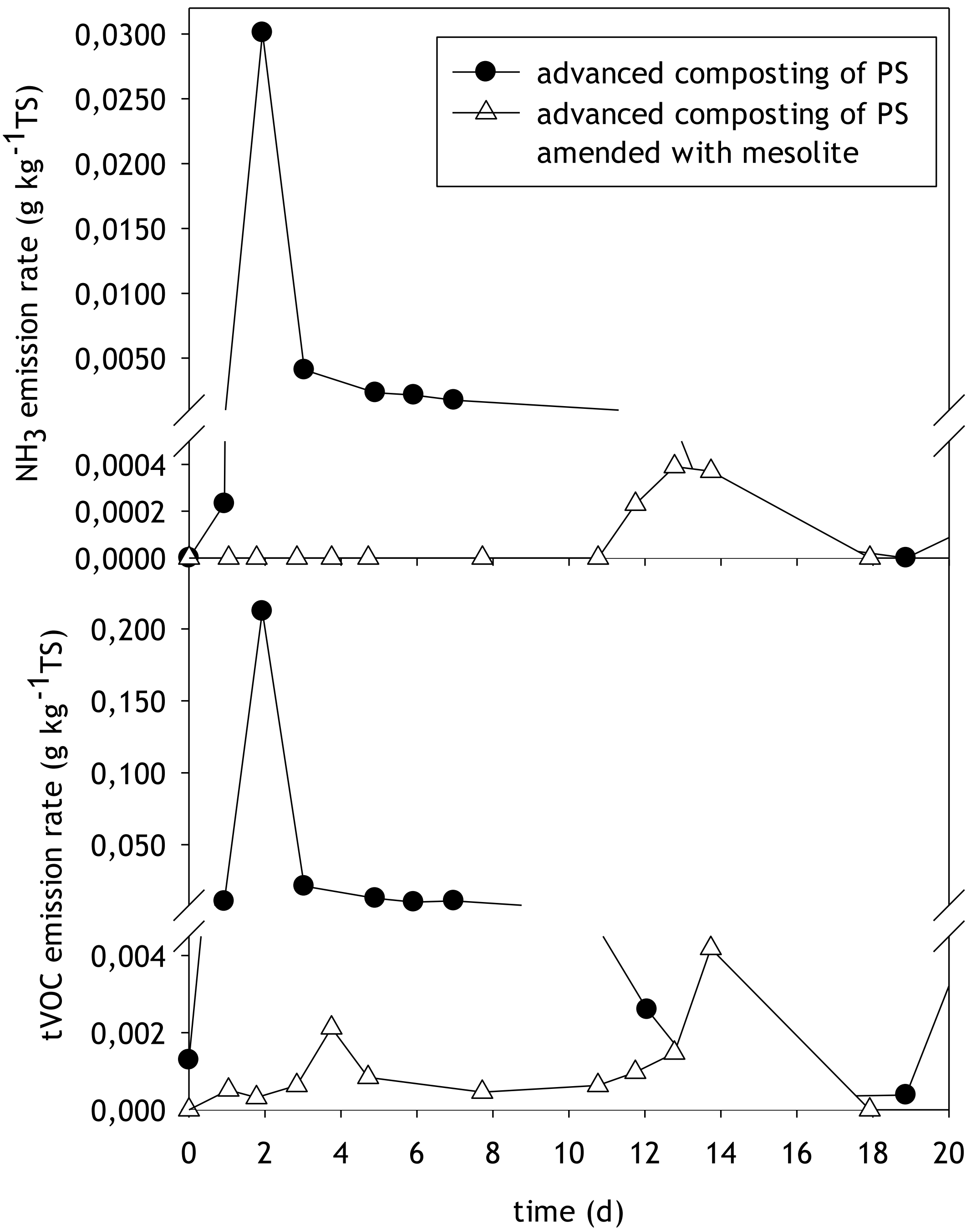


Figure V- 32 Comparison of NH_3 and VOCs emission profiles within advanced composting processes of PS without and with mesolite amendment.

APPENDIX VI. Detailed results of design and economic parameters from LCC models developed

Table VI-28. Design parameters of biodrying plant of CS.

Item	WWTP of Geestmerambacht	Plant size (P.E) 20000	40000	60000	80000	100000	150000	200000	250000	500000
Number of windrows required	1,0	1,0	1,0	1,0	1,0	1,0	1,0	1,0	1,0	1,0
Volume of windrows (m3)	64,0	4,8	9,7	14,5	19,4	24,2	36,3	48,5	60,6	121,2
Concrete needed (m3)	5,3	1,3	1,6	2,0	2,3	2,6	3,4	4,2	5,0	9,1
Insulation material needed (m3)	0,5	0,1	0,2	0,2	0,2	0,3	0,3	0,4	0,5	0,9
Cover material needed (m2)	32,0	2,4	4,8	7,3	9,7	12,1	18,2	24,2	30,3	60,6
Number of columns required	4,0	4,0	4,0	4,0	4,0	5,0	6,0	7,0	8,0	13,0
Stainless steel needed (kg)	249,6	249,6	249,6	249,6	249,6	312,0	374,4	436,8	499,2	811,2
Blower unitary capacity (m3/h)	1497,7	114,3	228,6	342,9	457,1	571,4	857,1	1142,8	1428,5	2857,0
Electricity consumption (kwh/y)	9451,6	8337,9	8424,4	8512,0	8424,4	8468,1	8578,2	8689,8	9392,4	9392,4
Diesel consumption (L/y)	579,5	44,2	88,4	132,7	176,9	221,1	331,7	442,2	552,8	1105,5
Pellets required to dry excess moisture content (t/y)	20,4	1,6	3,1	4,7	6,2	7,8	11,6	15,5	19,4	38,8

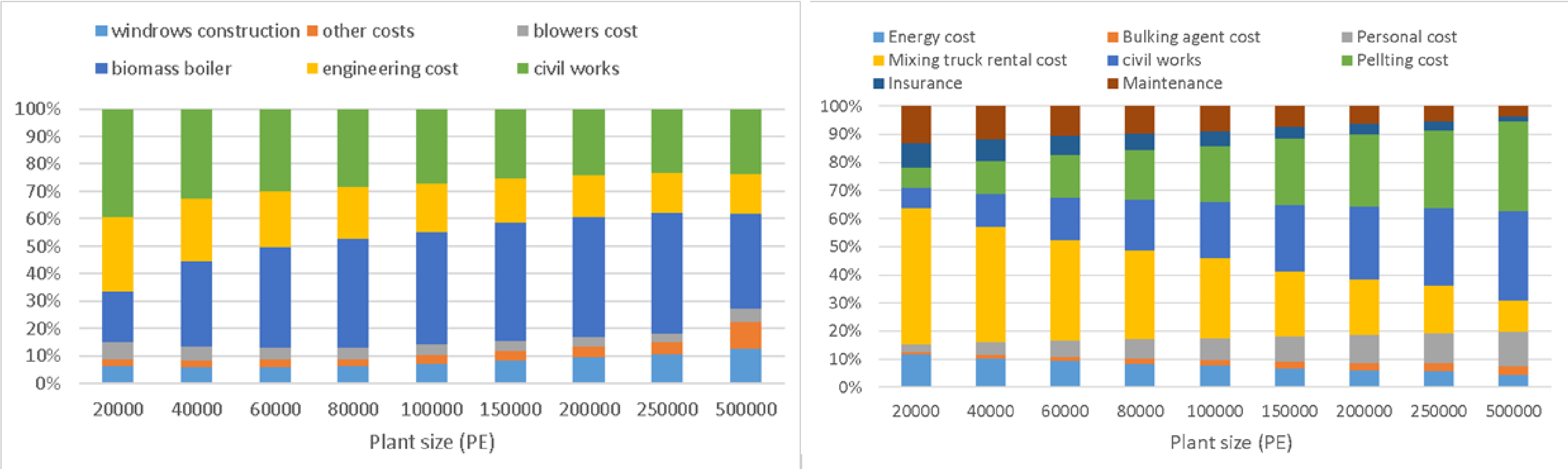


Figure VI-86. CAPEX and OPEX distribution in the categories considered in plant size dependent biodrying plant of CS.

Table VI-29 Economic parameters of biodrying plant of CS.

Parameter	WWTP of Geestmerambacht	Plant size (P.E) 20000	40000	60000	80000	100000	150000	200000	250000	500000
Overal CS production (t/y)	584 €	44,56	89,13	133,69	178,25	222,82	334,22	445,63	557,04	1114,08
CAPEX (€)	45,506 €	25,005 €	30,317 €	33,540 €	35,909 €	38,153 €	42,141 €	45,275 €	47,940 €	60,297 €
OPEX (€/y)	28,941 €	11,594 €	13,217 €	14,773 €	16,270 €	17,784 €	21,519 €	25,227 €	29,020 €	47,355 €
OPEX endlife (€)	723,513 €	289,860 €	330,434 €	369,316 €	406,760 €	444,598 €	537,979 €	630,671 €	725,494 €	1,183,883 €
Product selling (€/y)	16,157 €	1,226 €	2,451 €	3,677 €	4,902 €	6,128 €	9,192 €	12,255 €	15,319 €	30,639 €
Avoided costs (€/y)	11,681 €	891 €	1,783 €	2,674 €	3,565 €	4,456 €	6,684 €	8,913 €	11,141 €	22,282 €
Incomes (€/y)	27,838 €	2,117 €	4,234 €	6,351 €	8,467 €	10,584 €	15,876 €	21,168 €	26,460 €	52,920 €
Incomes endlife (€)	695,947 €	52,921 €	105,842 €	158,764 €	211,685 €	264,598 €	396,901 €	529,204 €	661,507 €	1,323,006 €
Benefits (€/y)	-1,103 €	-9,478 €	-8,984 €	-8,422 €	-7,803 €	-7,200 €	-5,643 €	-4,059 €	-2,559 €	5,565 €
Benefits endlife (€)	-27,566 €	-236,938 €	-224,592 €	-210,553 €	-195,075 €	-180,000 €	-141,078 €	-101,467 €	-63,987 €	139,123 €
NPV	-71,268 €	-246,443 €	-240,216 €	-230,318 €	-218,222 €	-206,377 €	-173,990 €	-140,104 €	-107,741 €	69,724 €
IRR 5 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A
IRR 10 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A
IRR 15 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	-17%
Payback period (y)	INF	INF	INF	INF	INF	INF	INF	INF	INF	INF

N,A refers to not available

Table VI-30, Design parameters of biodrying plant of PRS,

Item	WWTP of Almendralejo	Plant size (P,E) 20000	40000	60000	80000	100000	150000	200000	250000	500000
Number of windrows required	2	1	1	2	2	3	4	5	6	11
Volume of windrows (m3)	262	125	251	300	300	300	300	300	300	300
Concrete needed (m3)	80.2	39.1	17.7	91.6	91.6	142.6	193.6	244.6	295.6	550.6
Insulation material needed (m3)	8.0	3.9	1.8	9.2	9.2	14.3	19.4	24.5	29.6	55.1
Cover material needed (m2)	262	125	125.5	300	300	450	600	750	900	1650
Number of columns required	48	13	23	54	54	81	108	135	162	297
Stainless steel needed (kg)	2.995	1622.4	1435.2	3369.6	3369.6	5054.4	6739.2	8424	1.01E+04	1.85E+04
Blower unitary capacity (m3/h)	2014	962	3846	2885	3846	3205	3606	3846	4006	4371
Electricity consumption (kwh/y)	48585	23839	42379	63136	84359	104103	156520	210296	264810	541720
Diesel consumption (L/y)	2.395	1144	2287	3431	4574	5718	8577	11436	14295	28591
Pellets required to dry excess moisture content (t/y)	199	95	190	285	380	475	713	951	1188	2376

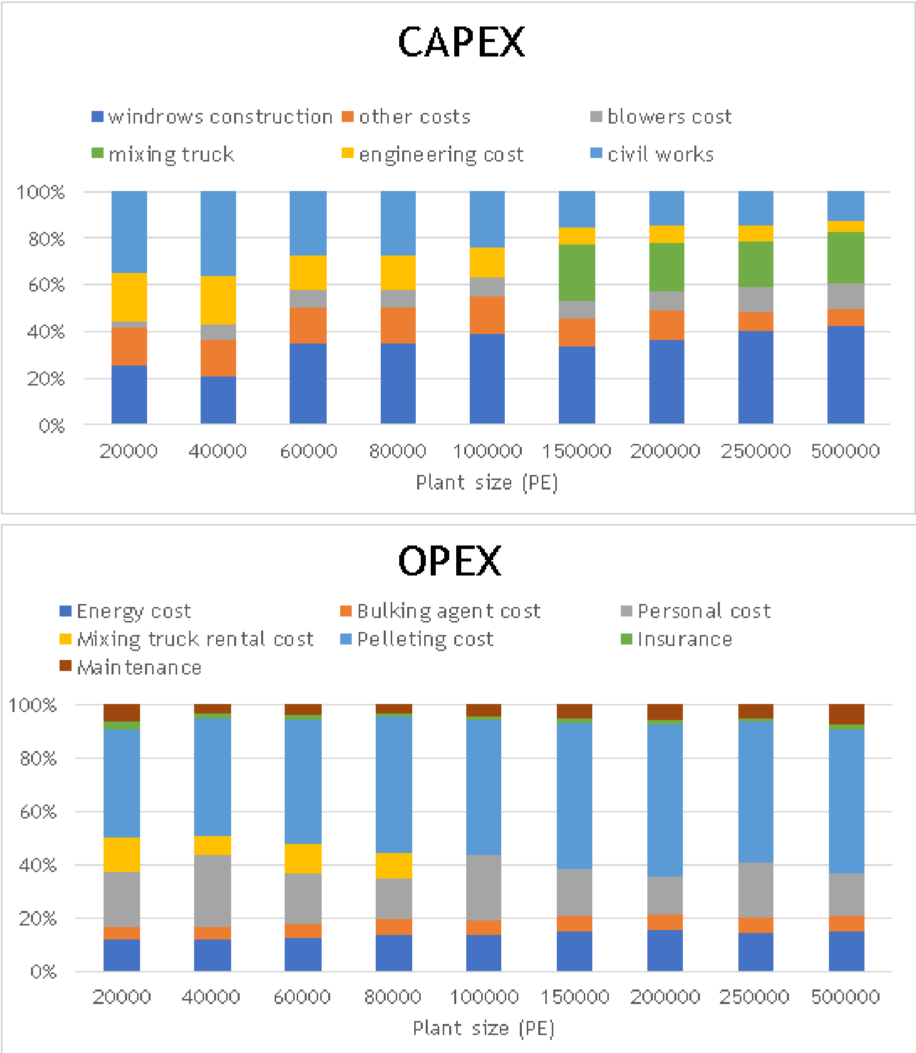


Figure VI-34, CAPEX and OPEX distribution in the categories considered in plant size dependent biodrying plant of PRS,

Table VI-31, Economic parameters of biodrying plant of PRS,

		Plant size (P,E)								
Parameter	**WWTP of Almendralejo**	**20000**	**40000**	**60000**	**80000**	**100000**	**150000**	**200000**	**250000**	**500000**
Overal PRS production (t/y)	2,613 €	1,248 €	2,496 €	3,744 €	4,991 €	6,239 €	9,359 €	12,478 €	15,598 €	31,196 €
CAPEX (€)	129,753 €	92,548 €	94,727 €	139,113 €	141,201 €	176,455 €	254,879 €	288,623 €	309,066 €	511,477 €
OPEX (€/y)	89,751 €	46,142 €	85,004 €	120,734 €	146,602 €	185,225 €	256,839 €	328,905 €	443,595 €	864,895 €
OPEX endlife (€)	2,243,779 €	1,153,546 €	2,125,105 €	3,018,346 €	3,665,045 €	4,630,628 €	6,420,982 €	8,222,633 €	11,089,871 €	21,622,378 €
Product selling (€/y)	39,455 €	18,838 €	37,676 €	56,514 €	75,353 €	94,191 €	141,286 €	188,381 €	235,477 €	470,953 €
Avoided costs (€/y)	52,269 €	24,957 €	49,913 €	74,870 €	99,827 €	124,784 €	187,175 €	249,567 €	311,959 €	623,918 €
Incomes (€/y)	91,724 €	43,795 €	87,590 €	131,385 €	175,179 €	218,974 €	328,461 €	437,949 €	547,436 €	1,094,871 €
Incomes endlife (€)	2,293,099 €	1,094,871 €	2,189,743 €	3,284,614 €	4,379,485 €	5,474,357 €	8,211,535 €	10,948,714 €	13,685,892 €	27,371,784 €
Benefits (€/y)	1,973 €	-2,347 €	2,586 €	10,651 €	28,578 €	33,749 €	71,622 €	109,043 €	103,841 €	229,976 €
Benefits endlife (€)	49,320 €	-58,674 €	64,638 €	266,268 €	714,440 €	843,728 €	1,790,553 €	2,726,081 €	2,596,021 €	5,749,406 €
NPV	-83,660 €	-147,383 €	-34,318 €	109,736 €	526,500 €	612,076 €	1,418,535 €	2,259,116 €	2,117,122 €	4,861,799 €
IRR 5 years	N,A	N,A	23	7	4	3	4	3	2	3
IRR 10 years	N,A	N,A	N.A	N.A	N.A	N.A	N.A	N.A	-66%	-28%
IRR 15 years	N,A	N,A	N.A	N.A	N.A	-15%	N.A	-37%	22%	53%
Payback period (y)	INF	INF	INF	-18%	14%	35%	14%	24%	58%	81%

N,A refers to not available

Table VI-32, Design parameters of advanced composting plants of PS,

Item	WWTP of Carbonera	25000	50000	100000	150000	200000	500000	750000	1000000	1500000
					Plant size (P,E)					
Number of windrows required	1	1	1	1	1	1	1	1	1	1
Volume of windrows (m3)	5	3	6	12	18	24	60	91	121	181
Concrete needed (m3)	1.3	1.2	1.4	1.8	2.2	2.6	5.0	7.1	9.1	13.1
Insulation material needed (m3)	0.1	0.1	0.1	0.2	0.2	0.3	0.5	0.7	0.9	1.3
Cover material needed (m2)	2.5	1.5	3.0	6.0	9.0	12.0	30.0	45.5	60.5	90.5
Number of columns required	4	4	4	4	4	4	5	6	7	10
Stainless steel needed (kg)	249.6	249.6	249.6	249.6	249.6	249.6	312.0	374.4	436.8	624.0
Blower unitary capacity (m3/h)	59.4	37.0	74.2	148.4	222.6	296.8	742.1	1113.0	1484.0	2226.1
Electricity consumption (kwh/y)	8296.1	8279.6	8307.1	8362.4	8418.1	8474.2	8818.9	9117.2	9426.2	10077.6
Diesel consumption (L/y)	213.8	133.3	267.3	534.5	801.8	1069.1	2672.7	4008.3	5344.7	8017.4

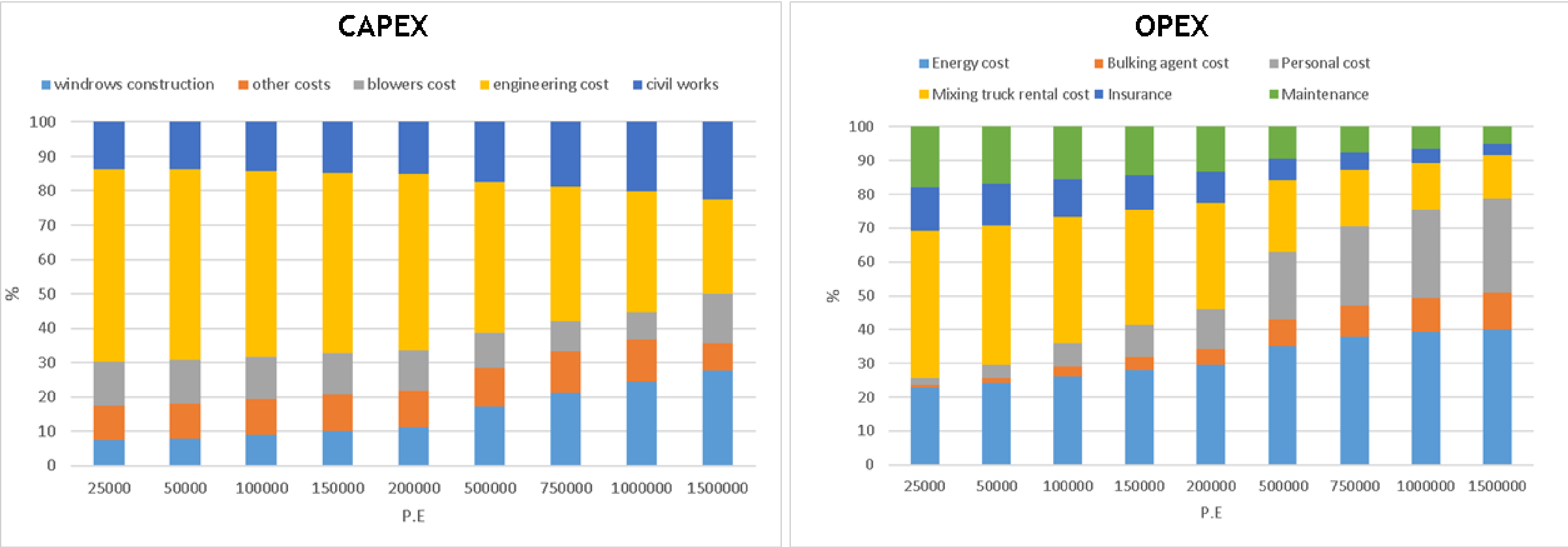


Figure VI-35, CAPEX and OPEX distribution in the categories considered in plant size dependent advanced composting plant of PS,

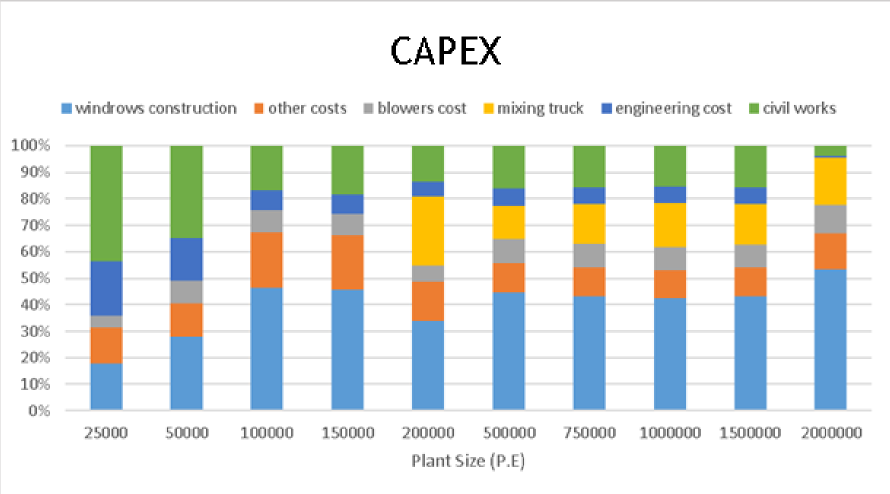
Table VI-33, Economic parameters of advanced composting plant of PS,

		Plant size (P,E)								
Parameter	**WWTP of Carbonera**	**25000**	**50000**	**100000**	**150000**	**200000**	**500000**	**750000**	**1000000**	**1500000**
Overal PS production (m3/y)	36,76	22,98	45,96	91,91	137,87	183,82	459,56	689,34	919,12	1378,68
Overal PS production (t/y)	29,41	18,38	36,76	73,53	110,29	147,06	367,65	551,47	735,29	1102,94
CAPEX (€)	12,592 €	12,483 €	12,646 €	12,973 €	13,300 €	13,628 €	15,915 €	17,930 €	19,890 €	25,638 €
OPEX (€/y)	6,837 €	6,916 €	7,296 €	8,054 €	8,812 €	9,571 €	14,131 €	17,935 €	21,740 €	30,321 €
OPEX endlife (€)	170,915 €	172,897 €	182,399 €	201,354 €	220,310 €	239,269 €	353,281 €	448,366 €	543,507 €	758,022 €
Product selling (€/y)	471 €	293 €	588 €	1,177 €	1,765 €	2,354 €	5,884 €	8,825 €	11,767 €	17,652 €
Avoided costs (€/y)	588 €	367 €	735 €	1,471 €	2,206 €	2,941 €	7,354 €	11,028 €	14,705 €	22,059 €
Incomes (€/y)	1,059 €	660 €	1,324 €	2,648 €	3,971 €	5,295 €	13,238 €	19,854 €	26,473 €	39,711 €
Incomes endlife (€)	26,470 €	16,504 €	33,095 €	66,190 €	99,285 €	132,381 €	330,951 €	496,341 €	661,816 €	992,768 €
Benefits (€/y)	-5,778 €	-6,256 €	-5,972 €	-5,407 €	-4,841 €	-4,276 €	-893 €	1,919 €	4,732 €	9,390 €
Benefits endlife (€)	-144,445 €	-156,393 €	-149,304 €	-135,163 €	-121,025 €	-106,888 €	-22,330 €	47,974 €	118,309 €	234,746 €
NPV	-147,587 €	-159,714 €	-153,897 €	-142,296 €	-130,696 €	-119,098 €	-50,080 €	7,178 €	64,516 €	154,654 €
IRR 5 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A
IRR 10 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	-22%
IRR 15 years	N,A	N,A	N,A	N,A	N,A	N,A	N,A	N,A	-9%	5%
Payback period (y)	INF	INF	INF	INF	INF	INF	INF	20	10	7

N,A refers to not available

Table VI-34, Design parameters of advanced composting plants of EBPR sludge,

		Plant size (P,E)								
Item	**WWTP of el Prat**	**25000**	**50000**	**100000**	**150000**	**200000**	**500000**	**750000**	**1000000**	**1500000**
Number of windrows required	34	1	1	2	3	3	8	13	17	25
Volume of windrows (m3)	300	126	253	300	300	300	300	300	300	300
Concrete needed (m3)	1723.6	18.8	35.7	183.2	285.2	285.2	795.2	1305.2	1713.2	2529.2
Insulation material needed (m3)	172.36	2.4	4.5	22.9	35.7	35.7	99.4	163.2	214.2	316.2
Cover material needed (m2)	5100	63.0	126.5	300.0	450.0	450.0	1200.0	1950.0	2550.0	3750.0
Number of columns required	476	6	12	28	42	42	112	182	238	350
Stainless steel needed (kg)	29702	374	749	1747	2621	2621	6989	11357	14851	21840
Blower unitary capacity (m3/h)	4295	1825	3651	3651	3651	4868	4564	4212	4295	4381
Electricity consumption (kwh/y)	399986	10566	13575	27150	40726	40726	112725	183223	241451	352352
Diesel consumption (L/y)	379425	4743	9486	18971	28457	37942	94856	142284	189712	284568



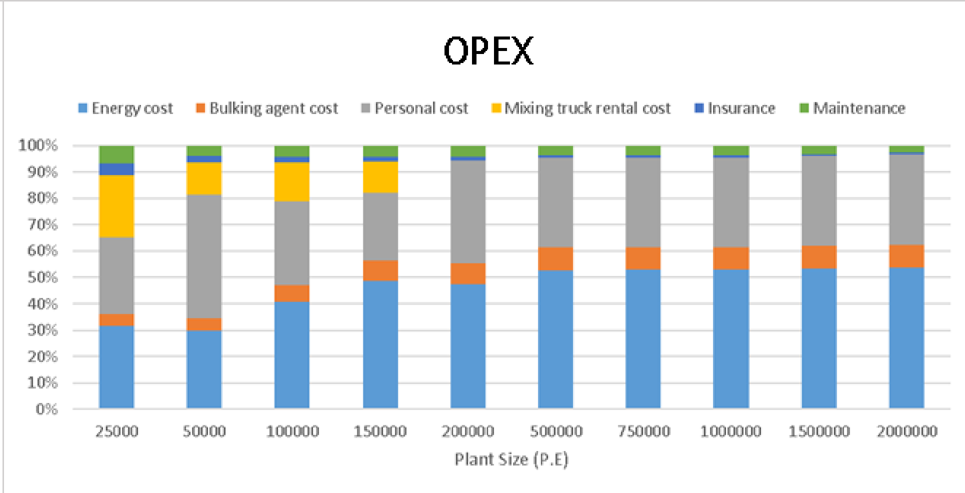


Figure VI-36, CAPEX and OPEX distribution in the categories considered in plant size dependent advanced composting plant of EBPR sludge,

Table VI-35, Economic parameters of advanced composting plant of EBPR sludge,

		Plant size (P,E)								
Parameter	**WWTP of el Prat**	**25000**	**50000**	**100000**	**150000**	**200000**	**500000**	**750000**	**1000000**	**1500000**
Overal PS production (t/y)	61,500	768,75	1537,5	3075	4612,5	6150	15375	23062,5	30750	46125
Overal PS production (m3/y)	246000	960,94	1921,88	3843,75	5765,63	7687,50	19218,75	28828,13	38437,50	57656,25
CAPEX (€)	6,753,307 €	34,392 €	43,109 €	90,002 €	140,475 €	190,475 €	395,044 €	663,683 €	888,594 €	1,288,415 €
OPEX (€/y)	969,872 €	25,330 €	49,076 €	72,108 €	89,992 €	118,442 €	270,132 €	408,692 €	543,964 €	806,592 €
OPEX endlife (€)	24,246,804 €	633,248 €	1,226,910 €	1,802,706 €	2,249,789 €	2,961,039 €	6,753,307 €	10,217,308 €	13,599,096 €	20,164,804 €
Product selling (€/y)	914,931 €	11,437 €	22,873 €	45,747 €	68,620 €	91,493 €	228,733 €	343,099 €	457,465 €	686,198 €
Avoided costs (€/y)	1,230,000 €	19,219 €	38,438 €	76,875 €	115,313 €	153,750 €	384,375 €	576,563 €	768,750 €	1,153,125 €
Incomes (€/y)	2,144,931 €	30,655 €	61,311 €	122,622 €	183,932 €	245,243 €	613,108 €	919,662 €	1,226,215 €	1,839,323 €
Incomes endlife (€)	51,478,338 €	766,387 €	1,532,773 €	3,065,538 €	4,598,312 €	6,131,077 €	15,327,692 €	22,991,542 €	30,655,385 €	45,983,077 €
Benefits (€/y)	1,175,059 €	5,326 €	12,235 €	50,513 €	93,941 €	126,801 €	342,975 €	510,969 €	682,252 €	1,032,731 €
Benefits endlife (€)	29,376,465 €	133,138 €	305,863 €	1,262,833 €	2,348,522 €	3,170,037 €	8,574,385 €	12,774,235 €	17,056,289 €	25,818,273 €
NPV	25,916,619 €	90,385 €	243,544 €	1,093,520 €	2,060,551 €	2,687,014 €	7,547,379 €	11,121,366 €	14,816,122 €	22,534,542 €
IRR 5 years	243%	#ZK!	-31%	100%	179%	176%	615%	308%	305%	375%
IRR 10 years	250%	-16%	27%	119%	189%	186%	616%	313%	309%	377%
IRR 15 years	250%	7%	35%	120%	189%	186%	616%	313%	309%	377%
Payback period	7	4	2	2	2	2	2	2	2	2

N,A refers to not available