UNIVERSIDAD DE GRANADA

ENERGY RECOVERY OF SCREENING WASTE FROM WASTEWATER TREATMENT PLANTS AS SOLID RECOVERED FUEL

VALORIZACIÓN ENERGÉTICA DEL RESIDUO DEL DESBASTE PROCEDENTE DE ESTACIONES DEPURADORAS DE AGUA RESIDUAL COMO COMBUSTIBLE SÓLIDO RECUPERADO

Juan Jesús de la Torre Bayo

Programa de Doctorado en Ingeniería Civil

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Juan Jesús de la Torre Bayo

Directores: Jaime Martín Pascual y Juan Carlos Torres Rojo

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Área de Tecnologías del Medio Ambiente (Departamento de Ingeniería Civil)

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D. Jaime Martín Pascual, Doctor Ingeniero de Caminos, Canales y Puertos y Profesor Titular de la Universidad del Departamento de Ingeniería Civil de la Escuela Técnica Superior de Ingenieros de Caminos, Canales y Puertos de la Universidad de Granada, y D. Juan Carlos Torres Rojo, Doctor en Ingeniería Ambiental por la Universidad de Granada y Gerente de la empresa EMASAGRA S.A.,

MANIFIESTAN:

Que la presente memoria titulada "VALORIZACIÓN ENERGÉTICA DEL RESIDUO DEL DESBASTE PROCEDENTE DE ESTACIONES DEPURADORAS DE AGUAS RESIDUALES COMO COMBUSTIBLE SÓLIDO RECUPERADO", presentada por D. Juan Jesús de la Torre Bayo para optar al Grado de Doctor con Mención Internacional por la Universidad de Granada, ha sido realizada bajo nuestra dirección en el Departamento de Ingeniería Civil de la Universidad de Granada y, por ello, autorizamos la presentación de la misma.

Granada, a 13 de julio de 2023

Fdo. Jaime Martín

Fdo. Juan Carlos Torres

Memoria presentada por D. Juan Jesús de la Torre Bayo para optar al Grado de Doctor con Mención Internacional por la Universidad de Granada.

Fdo. Juan Jesús de la Torre Bayo

TÍTULO DE DOCTOR CON MENCIÓN INTERNACIONAL

Con el fin de obtener del Título de Doctor por la Universidad de Granada con Mención Internacional, que el Real Decreto 99/2011 establece en su artículo 15, se han cumplido los siguientes requisitos:

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(ii) Parte de la tesis doctoral se ha redactado y presentado en una de las lenguas habituales para la comunicación científica en su campo de conocimiento, distinta a cualquiera de las lenguas oficiales en España.

(iii) La tesis ha sido informada por un mínimo de dos expertos doctores pertenecientes a alguna institución de educación superior o instituto de investigación no española.

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La presente tesis doctoral se presenta como reagrupamiento de los trabajos de investigación publicados por el doctorando en medios científicos relevantes en su ámbito de conocimiento. Para ello se han cumplido los siguientes requisitos:

(i) Se ha presentado un informe de los directores de la tesis respecto a la idoneidad de la presentación de la tesis bajo esta modalidad.

(ii) Los coautores de los trabajos han aceptado por escrito la presentación de los mismos como parte de la tesis doctoral. Al no haber coautores no doctores, no ha sido necesaria su renuncia a presentar dichos trabajos en otras tesis doctorales.

(iii) Los artículos que configuran la tesis doctoral están publicados o aceptados con fecha posterior a la obtención del título de grado y del máster universitario, no habiendo sido utilizados en ninguna tesis anterior y haciéndose mención a la Universidad de Granada a través de la afiliación del doctorando.

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La principal fuente de financiación de esta Tesis Doctoral ha sido la siguiente:

(i) Contrato con cargo a Grupos, Proyectos y Convenios. Número de Referencia de Contrato OTRI: 4325. Objeto: Valorización energética de residuos de desbaste como combustible sólido recuperado para lograr el residuo cero en EDAR. Entidades participantes: Universidad de Granada y EMASAGRA S.A. Responsable: Dª Montserrat Zamorano Toro. Duración: 15/11/2019 - 30/06/2023.

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La presente Tesis Doctoral forma parte de los estudios realizados bajo el soporte económico del proyecto con número de Referencia de Contrato OTRI: 4325, y que tiene como título "Valorización energética de residuos de desbaste como combustible sólido recuperado para lograr el residuo cero en EDAR".

Los trabajos descritos en la presente memoria se encuentran recogidos en las siguientes publicaciones y comunicaciones a congresos.

PUBLICACIONES INTERNACIONALES INCLUIDAS EN EL JCR

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De La Torre Bayo, J. J.; Martín Pascual, J.; Torres Rojo, J. C.; Zamorano Toro, M. Waste to Energy from Municipal Wastewater Treatment Plants: A Science Mapping. Sustainability 2022, 14 (24), 16871. https://doi.org/10.3390/su142416871. Impact Factor: 3.876 (Q2 Environmental sciences).

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Zamorano Toro, M., De la Torre Bayo, J.J., Martín Pascual, J; Torres Rojo, J.C.; Pennellini, S.; Bonoli, A. Production of SRF from Municipal Wastewater Plants Screening Waste: A New Approach Towards Zero Waste. 7th Edition of Global Energy Meet. Boston, MA. (United States of America). From 06 to 10 March 2023.

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CONCLUSIONS

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RESUMEN

En las últimas décadas los principios de la economía circular se han establecido como un campo de actuación inevitable en las estaciones depuradoras de aguas residuales (EDAR). El incremento de los costes energéticos junto con las obligaciones ambientalmente sostenibles en la gestión de los residuos está cambiando el enfoque dentro de la gestión del agua residual. El concepto de biofactoría redefine a las EDAR implementando procesos para la producción de energía y el avance hacia la consecución del residuo cero. Como parte de la gestión de las aguas residuales, además de la existencia de procesos de reciclaje de arenas y grasas, el desarrollo de tecnologías de conversión de residuos en energía, reconocidas por el concepto en inglés "waste to energy" (WtE), reside principalmente en el ámbito de los lodos. Sin embargo, el residuo de desbaste procedente del pretratamiento es el único que aún no dispone de proceso de valorización, siendo generalmente desechado mediante eliminación en vertedero.

Si bien existen estudios sobre las posibles alternativas a la eliminación en vertedero del residuo de desbaste, éstos están centrados en procesos de digestión anaeróbica y aún no se han estudiado otro tipo de vías de valorización de este residuo. En consecuencia, esta investigación ha tenido como objetivo principal el análisis en profundidad de la posibilidad de valorización energética del residuo de desbaste mediante su transformación en combustible sólido recuperado (CSR). A modo de establecer el estado del arte, se ha analizado la evolución científica de las tecnologías WtE en las EDAR municipales mediante una revisión bibliométrica. En relación con el trabajo de laboratorio, se han realizado los ensayos conducentes a la caracterización física y química del residuo del desbaste procedente de la Biofactoría Sur de Granada (España). A continuación, a nivel experimental y teniendo como principales etapas los procesos de secado, trituración y densificación, se produjo CSR sin densificar y densificado, en forma de pellets. La calidad del combustible generado se determinó mediante su caracterización, teniendo como base una nueva propuesta de clasificación del CSR que se desarrolló a partir de varias normativas existentes, tanto a nivel nacional como internacional. Finalmente se realizaron ensayos de laboratorio y balances de energía para los procesos de combustión, gasificación y pirólisis, dirigidos a evaluar el aprovechamiento del CSR producido mediante su valorización en procesos termoquímicos. Además, el proceso de producción de CSR, tanto densificado como sin densificar, se ha sometido a estudios de viabilidad ambiental y económica.

Los resultados del mapeo científico muestran un exponencial incremento de publicaciones WtE en EDAR, identificando al mismo tiempo la falta de estudio sobre los residuos del desbaste. El residuo analizado muestra una caracterización, mayormente compuesta de textiles sanitarios (52.1%) además de otras fracciones como papel, plásticos y vegetales. Esta composición se podría asemejar a aquella que compone la fracción rechazo procedente de residuos sólidos urbanos (RSU). Asimismo, su poder calorífico, y los contenidos en Cl y Hg, hacen viable su conversión en CSR según la norma ISO 21640:2021. Se comprobó la viabilidad técnica de la producción de CSR, tanto no densificado como densificado. Para la producción de pellets las variables de entrada fueron la humedad del residuo y el tamaño de prensa, concluyendo que las condiciones óptimas de peletización fueron para una humedad del 10% y con matrices con relaciones de compresión de 6/20, 6/24 y 8/32. Conjuntamente, la determinación de las características del CSR obtenido demostró que se cumplen los requerimientos de la clasificación propuesta, teniendo como destino más favorable de valorización las plantas de producción de energía a partir de residuos.

Para el estudio medioambiental y económico se establecieron cuatro escenarios de producción de CSR, diferenciando el tipo de secado y el producto final, como alternativa a la eliminación del residuo del desbaste en vertedero. El análisis de costes y beneficios, realizado mediante la obtención del Valor Neto Actual (VAN) empleando la simulación de Monte Carlo (MC), concluyó que la

eliminación en vertedero es el escenario más negativo en términos económicos. Además, al incluir a este análisis la monetización de las emisiones de CO_2 generadas se obtuvo que no es una solución viable. Esta conclusión fue también ratificada por el estudio de impacto ambiental, desarrollado mediante el Análisis de Ciclo de Vida (ACV) y para el cuál se utilizó el software SimaPro 9.2. Con esta herramienta se certificó que el vertedero supone el impacto más negativo en 6 de las 11 categorías ambientales analizadas según la metodología CML-IA baseline v3.08. Además, a partir de estos estudios, se obtuvo que el escenario más viable en términos económicos y ambientales sería la generación de CSR sin densificar y mediante la utilización del secado térmico en su proceso de producción.

Desde la perspectiva de la valorización energética se compararon los procesos termoquímicos de combustión y gasificación. El análisis teórico de la combustión se realizó mediante balance de energía, teniendo como corriente de entrada el CSR con diferentes niveles de humedad. Para la gasificación se realizaron ensayos a escala de laboratorio donde se determinaron los gases de salida producidos, a partir de los cuáles se llevó a cabo otro balance de energía, en este caso para la combustión de esos gases. La combustión del CSR en sólido fue el proceso más efectivo, con un beneficio energético máximo de 178.63 MJ por cada 100 kg de CSR en bruto (con humedad del 77.3%), mientras que la gasificación ofreció resultados máximos de 42.48 MJ para la misma cantidad de CSR. Desde un punto de vista no energético, la pirólisis, también analizada en base al diseño experimental a escala de laboratorio, reflejó la viabilidad de generar productos de valor añadido, como el char o el líquido de pirólisis.

ABSTRACT

In recent decades the principles of the circular economy have established themselves as an unavoidable field of action in wastewater treatment plants (WWTPs). Increasing energy costs and environmentally sustainable obligations in waste management are changing the focus on wastewater management. The biorefinery concept redefines WWTPs by implementing processes for energy production and moving towards zero waste. As part of wastewater management, in addition to sand and grease recycling processes, waste to energy (WtE) technologies development lies mainly in sludge. However, only the screening waste from pre-treatment still has no recovery process and is generally disposed of by landfill disposal.

Although there are studies on the possible alternatives to landfill disposal of the screening waste, these are focused on anaerobic digestion processes. This is the only type of recovery route for this waste that has yet to be studied. Consequently, the main objective of this research has been the in-depth analysis of the possibility of energy recovery of the screening waste through its transformation into solid recovered fuel (SRF). In order to establish the state of the art, the scientific evolution of WtE technologies in municipal WWTPs was analyzed using a bibliometric review. Tests leading to the physical and chemical characterization of the screening waste from the Biofactoría Sur in Granada (Spain) were carried out concerning the laboratory work. Then, at an experimental level and having as main stages the drying, crushing and densification phases, SRF was produced without densification and densified in the form of pellets. The fuel quality was determined by utilizing its characterization based on a new SRF classification proposal developed based on several existing national and international regulations. Finally, laboratory tests and energy balances were carried out for the combustion, gasification, and pyrolysis processes to evaluate the use of the SRF produced through its valorization in thermochemical processes. In addition, the SRF production process, both densified and non-densified, has been subjected to environmental and economic feasibility studies.

The results of the scientific mapping show an exponential increase of WtE publications in WWTPs, identifying at the same time the lack of study on the waste from desludging. The analyzed waste mainly comprises sanitary textiles (52.1%) and other fractions such as paper, plastics and vegetables. This composition could be similar to the rejection fraction from municipal solid waste (MSW). Furthermore, its calorific value and Cl and Hg contents make it viable for conversion into SRF according to ISO 21640:2021. The technical feasibility of SRF production, both non-densified and densified, was tested. For pellet production, the input variables were residue moisture and press size, obtaining that the optimum palletization conditions were for a moisture content of 10% and with matrices with compression ratios of 6/20, 6/24 and 8/32. Together, the determination of the characteristics of the SRF obtained showed that the requirements of the proposed classification were met, with the most favourable destination for recovery being waste-to-energy plants.

For the environmental and economic study, four SRF production scenarios were established, differentiating the drying type and final product as an alternative to landfill disposal of the screening waste. The cost-benefit analysis, carried out by obtaining the Net Present Value (NPV) using Monte Carlo (MC) simulation, concluded that landfill disposal is the most pessimistic scenario in economic terms. Furthermore, including the monetization of the CO2 emissions generated in this analysis concluded that it is not a viable solution. The environmental impact study also ratified this conclusion, developed using Life Cycle Analysis (LCA) and for which the SimaPro 9.2 software was used. This tool certified that the landfill has the most negative impact in 6 of the 11 environmental categories analyzed according to the CML-IA baseline v3.08 methodology. Furthermore, from these studies, it was obtained that the most viable scenario in economic and environmental terms would be the generation of SRF without densification and by using thermal drying in the production process.

From the perspective of energy recovery, the thermochemical processes of combustion and gasification were compared. The theoretical combustion analysis was carried out through energy balance, having the SRF with different moisture levels as the input stream. Laboratory-scale tests were carried out for gasification to determine the output gases produced. In this case, another energy balance was carried out for the combustion of these gases. Solid SRF combustion was the most effective process, with a maximum energy benefit of 178.63 MJ per 100 kg of raw SRF (at 77.3% moisture). In comparison, gasification gave maximum results of 42.48 MJ for the same amount of SRF. From a non-energy point of view, pyrolysis, also analyzed based on the laboratory-scale experimental design, reflected the feasibility of generating value-added products, such as char or pyrolysis liquid.

NOMENCLATURE

ODS	Objetivo de Desarrollo Sostenible
EC	Economía Circular
EDAR	Estación depuradora de aguas residuales
RSU	Residuo sólido urbano
TMB	Tratamiento mecánico biológico
LER	Lista Europea de Residuos
DQO	Demanda química de oxígeno
SV	Sólidos volátiles
CSR	Combustible sólido recuperado
PCI	Poder calorífico inferior

SDG	Sustainable Development Goal
CE	Circular Economy
WWTP	Wastewater treatment plant
MSW	Municipal solid waste
MBT	Mechanical biological treatment
EWC	European waste catalogue
COD	Chemical oxygen demand
VS	Volatile solids
TS	Total solids
WtE	Waste to Energy
SRF	Solid recovered fuel
RDF	Refuse derived fuel
WRRF	Wastewater resource recovery facility
HHV	Higher heating value
LHV	Lower heating value
Ar	As received
ND	Not determined
LCA	Life cycle assessment
NR	Not recommended

NOMENCLATURE

Dd	Diameter of pelletizing die
Lc	Compression length of pelletizing die
Dd/Lc	Compression ratio of pelletizing die
EfW	Energy from waste plants
Dp	Pellet diameter
Lp	Pellet length
DP	Pellet density
BD	Bulk density
DU	Durability
Mp	Pellet moisture
M	Non-densified SRF moisture
HD	Hardness
NR	Not recommended
NPV	Net Present Value
NPV_f	Financial Net Present Value
NPV_s	Social Net Present Value
MC	Monte Carlo
OMC	Operation and maintenance costs
RE	Annual revenues
BE	Industrial benefit
SP	Simulated price
FU	Functional unit
ADP	Abiotic depletion
ADP fossil	Abiotic depletion fossil
GWP100a	Global warming potential
ODP	Ozone layer depletion potential
HTP	Human toxicity potential
FAETP	Freshwater aquatic ecotoxicity potential
MAETP	Marine aquatic ecotoxicity potential
TETP	Terrestrial ecotoxicity potential

NOMENCLATURE

POI	Photochemical oxidation
AP	Acidification potential
EP	Eutrophication potential
ER	Equivalence ratio
S/C	Steam to carbon ratio
TGA	Thermogravimetric analysis
Cp	Specific heat capacity

INTRODUCCIÓN, MOTIVACIÓN Y OBJETIVOS

1 MARCO CONCEPTUAL Y NORMATIVO

A lo largo de los últimos años la preocupación sobre el medio ambiente y el agotamiento de los recursos naturales ha ido en continuo crecimiento (Nugraha et al., 2020). La Agenda 2030 de las Naciones Unidas, que incluye los 17 Objetivos de Desarrollo Sostenible (ODS), fue aprobada por los dirigentes mundiales en 2015 y supone un nuevo marco para el desarrollo sostenible a nivel mundial. En este sentido, la Unión Europea (UE) parte de una sólida posición en materia de desarrollo sostenible con un firme compromiso por ser pionera en la aplicación de la citada Agenda. Por este motivo sus políticas consideran prioritario el desarrollo sostenible, siendo la economía circular (EC) uno de sus pilares básicos (Fernández González et al., 2017).

La EC trata de mantener durante el mayor tiempo posible el valor de los materiales y recursos en el ciclo productivo (Unión Europea, 2016). Se trata de un modelo que busca valor en el mantenimiento y la reparación de productos, alargando su vida útil y evitando el despilfarro de recursos que supone la generación de residuos, facilitando la menor dependencia de recursos estratégicos, reforzando el tejido social y la economía local. Para ello, es necesario impulsar el reciclaje de los residuos, la recuperación y reutilización de nutrientes biológicos y técnicos, el uso de fuentes de energía renovables, entre otros, pero también la transformación de un modelo que cambia de consumidores a usuarios, en el que el fabricante optimiza la reutilización y remanufactura (Jain et al., 2022). Igualmente, para la eficacia del modelo es necesario promover la declaración y utilización de los denominados subproductos así como de la pérdida de condición de residuo. Este nuevo paradigma está adquiriendo una gran importancia en el diseño de sistemas productivos, administraciones, o ciudades (Kayal et al., 2018), con perspectivas de tener un impacto positivo en el crecimiento de la economía, la creación de empleo, la huella de carbono y el consumo de recursos naturales (Schröder et al., 2018).

El marco normativo que tiene como objetivo la evolución hacia una EC y una correcta gestión de los residuos es amplio tanto en su alcance como en su ámbito. En este trabajo se ha realizado una síntesis de las principales normativas

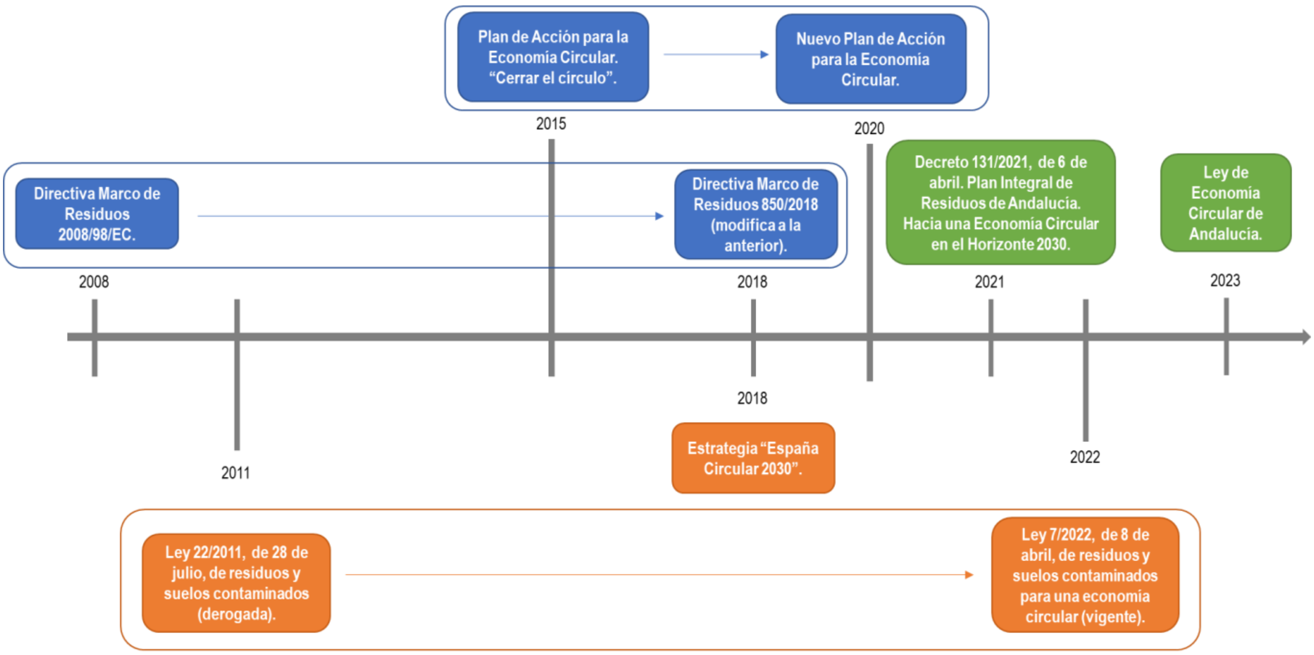


Figure 1. Resumen del Marco Normativo.

centrándose en las más recientes y las que actualmente están en vigor. La Figure 1 muestra la evolución temporal del marco legislativo de forma esquemática. Las normativas y los paquetes de medidas recopilados se diferencian a nivel europeo, estatal y autonómico.

Con el objetivo de facilitar y promover la transición hacia la EC, la Comisión Europea diseñó en 2015 su Plan de Acción para la Economía Circular que, bajo el lema "cerrar el círculo", incluyó medidas de desarrollo en ámbitos como la producción, el consumo, la gestión de residuos y el mercado secundario de materias primas (Europa, 2015). En el año 2019, la Comisión Europea publicó un informe sobre la aplicación del Plan de Acción para la Economía Circular, en el cual se recogían los resultados y retos pendientes para conseguir una EC, competitiva y neutra en emisiones de carbono. Dicho informe puso de manifiesto el logro de sus 54 acciones, las cuales habían sido ejecutadas o lo estaban siendo, además de mostrar que la EC se presenta como una oportunidad de negocio, desarrollo de nuevos modelos empresariales y de mercado. Entre los retos pendientes por alcanzar, el informe de la Comisión apuntaba, entre otros, a la necesidad de llevar a cabo mayores esfuerzos en la aplicación de la nueva legislación sobre residuos y en fomentar los mercados de materias primas secundarias (European Commission, 2019).

Continuando con los esfuerzos iniciados en 2015, y con la declaración de emergencia climática del Parlamento europeo, así como el impulso del Pacto Verde Europeo impulsado por parte de la Comisión (Europeo et al., 2019), en el año 2020 se aprobó un segundo Plan de Acción de Economía Circular (Comisión Europea, 2020). Esta nueva iniciativa nace apoyada en los resultados del plan anterior, así como en una serie de iniciativas que se interrelacionan para configurar una política marco de productos sostenibles fuerte y coherente con la jerarquía de residuos. El plan presenta un marco con un seguimiento riguroso del diseño de productos sostenibles, empoderamiento de los consumidores y compradores públicos, y circularidad en los procesos de producción. Además, implanta el estudio de las cadenas de valor en sectores clave de producción como: electrónica y TIC; baterías y vehículos; envases y embalajes; plásticos; productos textiles; construcción y edificios; y alimentos, agua y nutrientes. En este contexto, la Directiva 2008/98/EC (UE, 2008), marco de residuos, tenía como finalidad establecer una estrategia que permita alcanzar un conjunto de objetivos que buscan la reducción del consumo de recursos, así como la eliminación de residuos en vertedero, en el marco de una perspectiva sostenible, es decir que sea económica, social y ambientalmente posible. Tras la aprobación de los citados planes de acción, esta directiva ha sido modificada mediante la Directiva 850/2018 (Europeo et al., 2018) relativa al marco de residuos con la finalidad de incluir una serie de directrices relativas a la gestión de residuos y sus potenciales consecuencias relacionadas con el cambio climático, todas ellas en el marco de la EC. Entre los nuevos objetivos destacan la mayor restricción del depósito de residuos en vertederos, empezando por los flujos de residuos sujetos a recogida separada, como los de plásticos, metales, vidrio, papel y residuos orgánicos biodegradables, denominados biorresiduos, que aportaría evidentes beneficios medioambientales, económicos y sociales. Además, la viabilidad técnica, medioambiental o económica de procesos de reciclado u otro tipo de valorización de materiales o energética debe tenerse en cuenta al aplicar las restricciones en vertedero.

España, en su paso hacia el establecimiento de un modelo económico adecuado al contexto actual, ha desarrollado una serie de herramientas normativas y de planificación para afrontar el proceso de transformación necesario. En febrero de 2018 fue aprobada la "Estrategia Española de Economía Circular. España 2030", la cual representa el marco estratégico y de actuación imprescindible para facilitar y promover la transición hacia la EC a partir de la colaboración entre la Administración general del Estado, las comunidades autónomas, las entidades locales y los demás agentes implicados, en especial, personas productoras y consumidoras de bienes (España y Economía, 2018). La estrategia final titulada

"España Circular 2030" fue aprobada en junio de 2020 alineando sus objetivos con la Directiva 850/2018. El plan se marca una serie de objetivos cuantitativos a alcanzar para el año 2030 entre los que están: la reducción en un 30% del consumo nacional de materiales en relación con el PIB; disminución del 15% residuos generados; incremento de la reutilización y preparación para la reutilización hasta llegar al 10% de los residuos municipales generados; y mejora en un 10% la eficiencia en el uso del agua (Española, 2020).

En relación con la gestión de residuos, la Ley 7/2022, de 8 de abril, de residuos y suelos contaminados para una Economía Circular que deroga a la ley anterior (Ley 22/2011, de 28 de julio), tiene como objetivos principales la reducción en peso del 15% de los residuos generados en 2030 respecto al 2010, la limitación del uso de plásticos mediante la reducción del 70% de la comercialización de estos en 2030, restricciones en la eliminación de residuos en vertedero, y obligaciones para los gestores de residuos con objeto de fomentar la reutilización, el reciclaje y la valorización (Jefatura del Estado, 2022). Esta Ley sigue las directrices de la Directiva 850/2018 a fin de garantizar la correcta aplicación de la jerarquía de residuos, imponiendo para ello una serie de actuaciones que debe ser empleada como orden prioritario en la legislación y política de prevención y gestión de residuos, de acuerdo con el siguiente orden: prevención; preparación para la reutilización; reciclado; otro tipo de valorización, incluida la valorización energética; y eliminación.

A nivel autonómico, el Decreto 131/2021, de 6 de abril, aprueba el Plan Integral de Residuos de Andalucía. Hacia una Economía Circular en el Horizonte 2030 (PIRec 2030). Entre las principales novedades del plan se encuentra la recogida separada de biorresiduos, los centros de preparación para la reutilización o la atención a la creciente presencia de residuos plásticos en la costa. A partir de 8 objetivos generales se formula un plan integral de impulso a la EC, que desarrolla tres programas: el de prevención, el de gestión y el de concienciación, sensibilización y comunicación. Los residuos sólidos municipales concentran el mayor número de actuaciones pretendiendo para 2035 un 10% máximo de depósito en vertedero y un 65% mínimo de preparación para reutilización y reciclado (Junta de Andalucía, 2021).

Posteriormente, a partir del 1 de mayo de 2023, entró en vigor la Ley de Economía Circular de Andalucía, pretendiendo crear un marco normativo adecuado para el desarrollo de la EC en el ámbito competencial de la Comunidad Autónoma. Entre los aspectos más relevantes que supone su implantación se encuentra la puesta en marcha del Registro público andaluz de análisis de ciclo de vida, con aspectos relativos a este análisis en productos, obras y servicios. La norma prevé que los proyectos de actividades de valorización energética sean considerados como inversiones empresariales de interés estratégico, además especifica el procedimiento para que un residuo valorizado pueda ser considerado como subproducto para su uso en una actividad o proceso industrial. Dentro de la aplicación de la EC al ciclo integral del agua, se aborda específicamente la utilización de las aguas regeneradas, con especial atención a las masas de agua clasificadas en mal estado, así como al uso de lodos de depuradora (Andalucía, 2023).

2 GESTIÓN SOSTENIBLE DE AGUAS RESIDUALES

La gestión del agua tiene un papel relevante en la consecución de los ODS para la Agenda 2030 marcada por las Naciones Unidas, de hecho, repercute directamente en el ODS 6: Agua limpia y saneamiento. Además, las metas de este objetivo trascienden al resto de ODS, ya que ningún objetivo puede ejercerse con plenitud si no existe la garantía previa del derecho universal al agua (Obaideen et al., 2022). De hecho, (Schröder et al., 2018) en un estudio sobre las relaciones y sinergias más fuertes entre las directrices de EC y las metas de los ODS identificó el ODS 6 como uno de ellos. En este trabajo, (Schröder et al., 2018) también afirman que las prácticas de EC asociadas con los sistemas de circuito cerrado para el reciclaje y reutilización de aguas residuales (Nick Jeffries, 2017) y el reciclaje de lodos de depuradora (Angelakis and Snyder, 2015) serán indispensables para alcanzar las metas del citado ODS, además de algunas de las incluidas en la meta 14.1, relativa al ODS14 sobre ecosistemas marinos.

Las estaciones de depuración de aguas residuales (EDARs) tienen como objetivo disminuir la carga contaminante de las aguas después de su uso (bien sean urbanas, industriales o una mezcla de ambas) antes de devolverlas al medio natural, para lo cual se emplean diferentes tratamientos físicos, químicos y biológicos. En las últimas décadas, el número de EDARs ha aumentado en todo el mundo para cumplir con las normativas existentes y mitigar el deterioro del medio acuático (Di Fraia et al., 2018). Además, se prevé que este aumento de EDARs genere un aumento del consumo energético, que ya representa un porcentaje importante de los costes energéticos de un municipio, con cifras que oscilan entre el 24 (Wu et al., 2021) y el 44% (Cardoso et al., 2021). Esta situación puede suponer un aumento considerable de las emisiones de CO_2 y un incremento sustancial de los costes de operación (Cardoso et al., 2021). El consumo energético de las EDARs municipales puede variar significativamente en función de factores internos como la producción de energía in situ a partir de fuentes renovables (Lindtner et al., 2008), las tecnologías aplicadas (Mamais et al., 2014) y la intensidad de éstas; y externos como por ejemplo las tarifas energéticas (Lindtner et al., 2008). Es por ello que, como sistema productivo, resulta fundamental la optimización de los procesos en la EDARs (Newhart et al., 2019), los cuales tengan en consideración el nuevo paradigma de la EC que busca la redefinición de estas instalaciones en biofactorías. Esta meta pasa por implantar nuevos modelos de trabajo basados una serie de principios básicos que incluyen la reutilización de agua, la autosuficiencia energética, la reducción y la valorización de los residuos que se generan durante el proceso productivo. En este sentido, las infraestructuras existentes para la gestión de las aguas residuales, en los sistemas de los países industrializados, no siempre son adecuadas para soportar la EC, por lo que deberán optimizarse buscando la aplicación de los principios citados (Schröder et al., 2018).

Debido al alto contenido en materia orgánica presente en estos procesos, las EDAR también representan una rica fuente de energía química (Lindtner et al., 2008). Por tanto, la redefinición de las EDARs en biofactorías pasa, necesariamente, por las tecnologías de conversión de los residuos en energía, reconocidas por el concepto “waste to energy” (WtE) y que podrían tener su aplicación en los residuos que generados en las distintas fases de la depuración (residuo del desbaste, arenas, grasas y lodos). Estas tecnologías se han aplicado en las EDARs principalmente para la gestión de los lodos, lo que ha permitido la producción de biogás a partir de la digestión y codigestión de la materia orgánica (Gandiglio et al., 2017), que posteriormente se convierten en electricidad y calor; Además, como opciones sotenibles para el tratamiento de lodos también se han realizado procesos de producción de biodiesel (Liu et al., 2021), combustión (Liang et al., 2021) y cocombustión (Zhang et al., 2020), gasificación (Migliaccio et al., 2021a) y pirólisis (Agar et al., 2018). Además de los lodos también existen tecnologías para el reciclaje de las arenas y las grasas, generalmente enfocadas a la agricultura (Valdés López et al., 2021).

Sin embargo, aunque los lodos representen la mayor parte de los residuos generados en una EDAR y los estudios científicos se centren principalmente en su tratamiento, existen aparte otros residuos sólidos como el desbaste, que se genera en el pretratamiento. Este residuo actualmente no dispone de una gestión adecuada al marco normativo expuesto anteriormente, por lo tanto, existe una oportunidad de aprovechar estas fuentes de energía durante el proceso de tratamiento de las aguas residuales (Gandiglio et al., 2017).

3 RESIDUO DE DESBASTE

A la entrada de la EDAR, el agua residual contiene gran cantidad de material voluminoso que debe ser eliminado, con el fin de permitir las siguientes fases del tratamiento, lo que genera una producción de residuos, tanto en la obra de llegada como en la fase de pretratamiento. Esta fracción de residuos está formada, entre otros, por una mezcla heterogénea de papel, restos sanitarios y plásticos, y queda clasificada con el código Lista Europea de Residuos (LER) 19 08 01 (Consejo del Parlamento Europeo, 2014), y descrita como residuos de cribado, dentro del capítulo 19 Residuos de las instalaciones para el tratamiento de residuos, de las plantas externas de tratamiento de aguas residuales y de la preparación de agua para consumo humano y de agua para uso industrial, concretamente en el subcapítulo 19 08 Residuos de plantas de tratamiento de aguas residuales no especificados en otra categoría.

La cantidad de residuos del desbaste generada en una EDAR depende directamente del tipo de red que haya implantada en el municipio. Asimismo, la lluvia es un factor meteorológico que, en el caso de redes de saneamiento unitarias, produce un arrastre y limpieza de los residuos depositados en tiempo seco, en la vía pública y en la propia red de saneamiento; así, en días lluviosos la producción de desbaste puede llegar a ser un 50% mayor que en días secos (Canler and Perret, 2004). La cuantía de residuos del desbaste también depende del equipamiento, del pretratamiento y de la luz de paso del tamizado de la EDAR (Le Hyaric et al., 2009); así, según Le Hyaric (2009), hay gran diferencia entre la luz de tamiz de 6 mm que proporciona unas tasas de generación de 0,08 kg/año.heq, y 3 mm, que las incrementa hasta los 0.15 kg/año.heq, sendas en materia seca. Por último, añadir que el proceso de compactación frecuentemente utilizado, produce una disminución en el volumen del residuo generado (Clay et al., 1996).

El residuo del desbaste supone en torno al 2% del total de los residuos generados durante el proceso de depuración de aguas (Le Hyaric et al., 2010), con valores, en materia seca, que oscilan desde 0.08 kg/año.heq (Le Hyaric et al., 2010) hasta 1.1 kg/año.heq (Kaless et al., 2016). Esta variabilidad en los datos muestra la complejidad en el análisis y caracterización de este tipo de residuos, con resultados de producción muy heterogéneos en los diferentes estudios realizados. Además, se pone de manifiesto la baja producción de residuos de desbaste, si se les compara con la generación de otras fracciones producidas en el proceso de depuración, como los lodos, que en Andalucía se sitúa entre 19 y 31 kg/año.heq, en materia seca (Granados González, 2015).

En relación a las fracciones que lo componen en términos generales, el residuo del desbaste está formado principalmente por textiles sanitarios, papel y cartón, vegetales, plásticos y pequeños porcentajes de finos o materiales metálicos (Gregor et al., 2013; Le Hyaric et al., 2009; Naud et al., 2007). La predominancia de textiles sanitarios, cuya presencia ha ido creciendo progresivamente con el paso de los años con el cambio de los hábitos de la sociedad (Wid and Horan, 2016) y que en la actualidad supera un valor promedio del 50% es una de sus características. En 1996 el estudio llevado a cabo por Clay et al. reflejaba unos porcentajes que se situaban en torno a un 25% de textiles sanitarios, cifra que en la actualidad se puede llegar a triplicar (Wid and Horan, 2016) y alcanzar máximos de 87,1% (Le Hyaric et al., 2009). El papel y los vegetales, materia potencialmente biodegradable, tienen un volumen relevante dentro del residuo, aunque también se observan importantes diferencias entre los estudios consultados y que oscilan entre 1,3 y 13, 1% (Wid and Horan, 2016) . La presencia de finos, es decir partículas de menos de 20 mm de diámetro que son muy difícilmente separables, se encuentran entre el 7,6 y 15,2% (Le Hyaric et al., 2009). Finalmente, se puede observar también una pequeña parte formada por plásticos, metales y materiales no biodegradables, que no superan en su conjunto entre el 3,1 y 9,7% (Le Hyaric et al., 2009; Wid and Horan, 2016).

En cuanto a sus características físicas, el contenido en agua influye directamente, entre otros, en el poder calorífico de los residuos y su estudio es fundamental para su correcta gestión (Arrechea et al., 2009). El residuo del desbaste muestra un elevado contenido en agua, incluso una vez compactado, que oscila entre un 70 y un 85% (Cadavid-Rodriguez and Horan, 2012; Kaless et al., 2016; Wid and Horan, 2016). El valor de este parámetro depende de factores como el clima, el tipo de tamizado o la presencia o no de un proceso previo de compactación; en este último caso, el proceso de reducción de volumen aumenta la densidad del residuo a valores que suelen oscilar entre 0,6 y 1 kg/L (Naud et al., 2007; Le Hyaric et al., 2009; F. Valiron, 1994), además de contribuir a la disminución de parte del agua presente en los residuos (Le Hyaric et al., 2009). Si se comparan estos valores con los obtenidos en la gestión de residuos sólidos urbanos (RSU), el rechazo procedente del tratamiento mecánico biológico (TMB), el más utilizado para su procesado posterior en digestión anaerobia (Pires et al., 2011), muestra valores de contenido en agua sustancialmente inferiores con valores próximos al 30% (Edo-Alcón et al., 2016). Otros parámetros físicos que permiten evaluar la calidad del producto es el contenido en sólidos volátiles y en cenizas. En el caso de la materia volátil se muestran valores en torno al 90% respecto al total de sólidos (Le Hyaric et al., 2010; Cadavid-Rodriguez and Horan, 2012; Wid and Horan, 2018). En cuanto al contenido en cenizas, se han encontrado datos en el estudio sobre digestión anaeróbica para residuos del desbaste desarrollado por Cadavid – Rodríguez & Horan (2012), que registran unos valores medios de 2,1%.

En relación con la caracterización química, el análisis elemental para determinar el contenido en carbono, hidrógeno, nitrógeno y azufre es frecuente en la caracterización de RSU (Edo-Alcón et al., 2016; Nasrullah et al., 2015). En la literatura sobre el residuo del desbaste, este tipo de determinaciones, tan sólo se han encontrado en el estudio de Cadavid – Rodríguez & Horan (2012) que tan sólo mostró valores para el contenido en carbono y nitrógeno, con un 48,5% [46,4 – 50,2] y 2,5% [2,3 – 2,9] respectivamente. El contenido en materia orgánica es el parámetro que muestra, en los estudios consultados, una heterogeneidad más elevada, con valores que oscilan desde un 77 hasta un 88%, y un porcentaje de materia orgánica oxidable en torno al 45% [37 – 51] (Le Hyaric et al., 2009). Sidwick, (1991) explica esto por factores relativos a, fenómenos meteorológicos que pueden variar la presencia de vegetales y materiales minerales, el tipo y longitud de la red de saneamiento, que puede beneficiar la eliminación de materia orgánica antes de su llegada a la EDAR; o las características del origen de las aguas residuales (zona urbana, industrial, actividad turística…). Los valores recogidos en los estudios consultados muestran valores para el poder calorífico inferior (PCI), en base seca, del residuo del desbaste que varían entre 17,70 MJ/kg (Sidwick, 1984) y 20,90 MJ/kg (Canler and Perret, 2004), cifras similares a las de la fracción rechazo de RSU (Gallardo et al., 2014; Bessi et al., 2016).

Aunque un pequeño porcentaje de los residuos de desbaste es incinerado, en la actualidad, su destino más habitual es la eliminación en vertedero (Wid and Horan, 2018; Clay et al., 1996). Esta solución, además de generar problemas ambientales, requiere un importante coste de transporte dada su elevada humedad; a esto se le añaden los posibles problemas en cuanto a la admisión del residuo en vertedero, por su alto contenido en materia orgánica (Cadavid-Rodriguez and Horan, 2012). Por otro lado, la eliminación de residuos mediante depósito en vertedero está obligada a desaparecer ya que con las nuevas restricciones planteadas en la Directiva 850/2018 (Europeo et al., 2018) en 2035 la cantidad en peso de residuos municipales vertidos tendrá que reducirse a un máximo del 10% (MITECO - Ministerio para la Transición Ecológica y el Reto Demográfico, 2020), por lo que es necesario buscar alternativas a la actual eliminación en vertedero de los residuos de desbaste procedentes de la EDAR y así contribuir al logro de objetivos establecidos. Finalmente, en el caso de España, la aprobación del nuevo impuesto sobre el depósito de residuos en vertederos, la incineración y la coincineración de residuos implicará un sobre coste de 40 € por tonelada para la eliminación del residuo de desbaste en vertedero.

El bajo porcentaje de producción que representan los residuos del desbaste en una EDAR, así como su clasificación como residuo no peligroso de acuerdo a lo recogido en el LER, ha sido una de las causas principales por la que la investigación científica sobre ellos haya recibido poco interés hasta la fecha, tal y como se pone de manifiesto en el bajo número de documentos publicados en esta materia, los cuales fundamentalmente se centran en la aplicación de procesos de digestión anaeróbica que permitan la producción de biogás, tanto específicamente para estos residuos, como incorporándolos al sistema de codigestión anaeróbica de la EDAR. Los trabajos revisados han puesto de manifiesto que el alto contenido en materia orgánica de la fracción de desbaste sugiere que la digestión anaeróbica puede ser un método viable de reciclaje y recuperación de energía (Md Rahman et al., 2018). Con el fin de analizar este tipo de procesos para los residuos de desbaste, Kaless (2017) determinó la demanda química de oxígeno (DQO) del residuo, concluyendo que podría enviarse a un proceso de digestión anaeróbica o como fuente de carbono para la desnitrificación durante el proceso de tratamiento de aguas. Estas soluciones podrían aportar una significante reducción de la huella de carbono de este residuo, derivada de su eliminación en vertedero; sin embargo, los estudios consultados también han puesto de manifiesto que la presencia de plásticos y textiles dificulta la incorporación del residuo a los digestores (Cadavid-Rodriguez and Horan, 2012). En consecuencia, la implementación de un digestor dedicado para este residuo a nivel industrial, según Le Hyaric (2010), no se encontró económicamente viable. Por ello se han reportado resultados que muestran la opción de la codigestión como una alternativa potencial al tratamiento de los residuos de desbaste, con una producción de biogás estabilizada con diversidad de contenidos de metano que van desde los 0,33 m^3/kg de sólidos volátiles (SV) (Cadavid-Rodriguez and Horan, 2012) hasta los 0,51 y 0,62 m^3/kg (SV) para Le Hyaric (2010).

La bibliografía revisada sobre residuos del desbaste abre la posibilidad de la viabilidad de la producción de un combustible a partir de estos residuos. Debido a su composición y a su PCI, se podría asimilar al rechazo procedente de RSU, para estudiar su valorización energética mediante procesos termoquímicos (Dong et al., 2010), en el marco de las tecnologías WtE. Sin embargo, no se han identificado estudios científicos que sean concluyentes en este punto.

4 OBJETIVOS

Por todo lo expuesto, la búsqueda de alternativas a la actual eliminación del residuo de desbaste en vertedero es clave en el desarrollo sostenible en la gestión de aguas residuales. Si bien existen estudios sobre este residuo, éstos no se centran en su posible valorización como combustible sólido. En consecuencia, esta investigación ha tenido como principal objetivo estudiar la viabilidad técnica, económica y medioambiental de la valorización energética del residuo de desbaste procedente de EDAR mediante la producción de combustible sólido recuperado. Para alcanzarlo, se han definido los siguientes objetivos secundarios:

- Revisión bibliográfica sobre la aplicación de tecnologías WtE para los residuos procedentes de EDAR.
- Estudio de la producción y caracterización del residuo de desbaste.
- Producción y caracterización a escala de laboratorio de CSR a partir del residuo de desbaste.
- Estudio de viabilidad económica para la implantación del proceso de producción de CSR.
- Análisis del impacto ambiental derivado de la potencial producción de CSR.
- Aplicación de tecnologías termoquímicas para la valorización del CSR producido.

Para el desarrollo de este trabajo se ha utilizado el residuo de desbaste producido en la Biofactoría Sur de Granada, gestionada por la Empresa Municipal de Abastecimiento y Saneamiento de Granada, S.A. (EMASAGRA), la cual trata más de 18 millones de m^3 anuales de agua residual procedente de unos 425000 habitantes equivalentes. Está instalación, ya ha alcanzado los objetivos de autosuficiencia energética; de hecho, en el año 2017 logró un valor de autoconsumo energético de un 82,5%, en abril de 2018 se marcó un máximo puntual de más de un 100% de autosuficiencia y en el año 2019 ya se consiguieron datos medios anuales de autosuficiencia energética, lo que le ha llevado a convertirse en un referente en Europa en el sector de gestión de aguas. Además, la Biofactoría Sur tiene marcada una hoja de ruta que facilite el reciclaje y valorización de los residuos generados en el proceso, buscando "Zero energy, zero waste" en los próximos años. Para ello la totalidad de los lodos, las arenas y las grasas generadas en la misma son ya recicladas en aplicaciones agrícolas, quedando en la actualidad únicamente que buscar una forma de reciclaje y/o valorización de los residuos de desbaste, en busca del residuo cero.

INTRODUCTION, MOTIVATION AND OBJECTIVES

1 CONCEPTUAL AND REGULATORY FRAMEWORK

Concern about the environment and the depletion of natural resources has been growing steadily over the last years (Nugraha et al., 2020). The United Nations 2030 Agenda, which includes the 17 Sustainable Development Goals (SDGs), was adopted by world leaders in 2015 and represents a new framework for sustainable development globally. In this regard, the European Union (EU) starts from a strong position in terms of sustainable development with a firm commitment to be a pioneer in the implementation of the Agenda. For this reason, its policies consider sustainable development a priority, with the circular economy (CE) being one of its fundamental pillars (Fernández González et al., 2017).

The CE seeks to maintain the value of materials and resources in the production cycle for as long as possible (Unión Europea, 2016). It is a model that seeks value in the maintenance and repair of products, extending their useful life and avoiding the waste of resources that waste generation entails, facilitating less dependence on strategic resources, and strengthening the social fabric and the local economy. To this end, it is necessary to promote the recycling of waste, the recovery and reuse of biological and technical nutrients, and the use of renewable energy sources, among others, but also the transformation of a model that changes from consumers to users, in which the manufacturer optimizes reuse and remanufacturing (Jain et al., 2022). Equally, for the model's effectiveness, promoting the declaration and utilization of so-called by-products and the loss of waste status is necessary. This new paradigm is gaining importance in the design of production systems, administrations, or cities (Kayal et al., 2018), with prospects of having a positive impact on the growth of the economy, job creation, carbon footprint and consumption of natural resources (Schröder et al., 2018).

The regulatory framework aimed at the evolution towards a CE and proper waste management is broad in scope and scope. In this paper, the principal regulations have been synthesized, focusing on the most recent ones and those currently in force. Figure 2 shows the temporal evolution of the regulatory framework in a schematic way. The regulations and packages of measures compiled are differentiated at European, national, and regional levels.

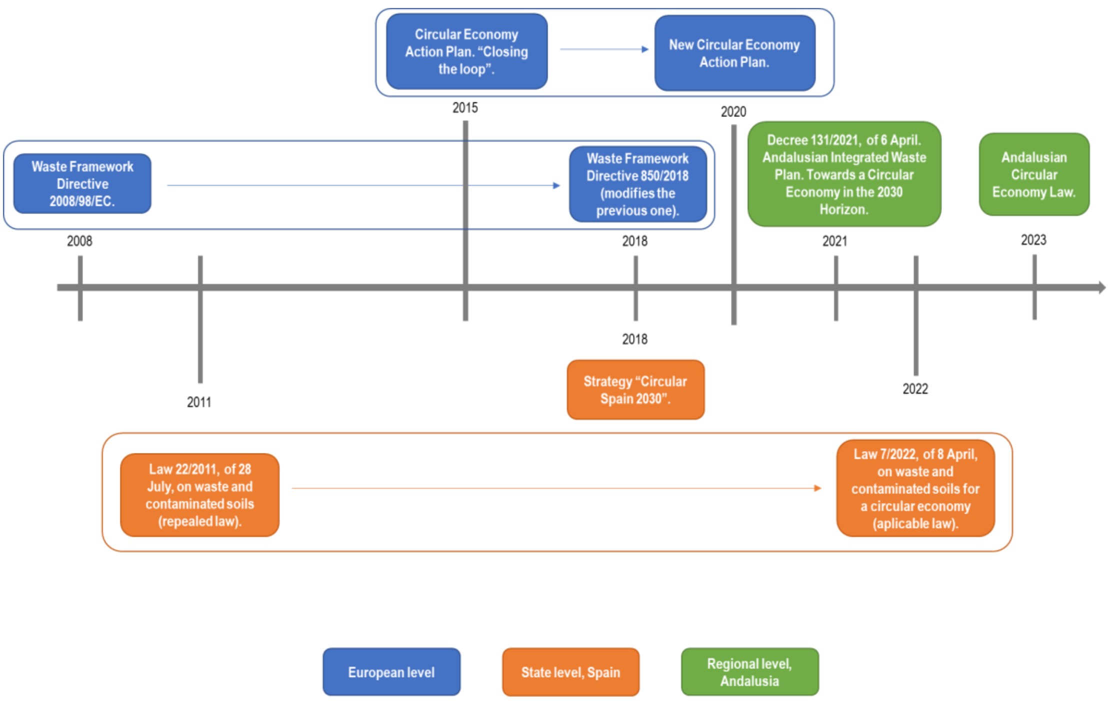


Figure 2. Summary of the Regulatory Framework

Intending to facilitate and promote the transition to the CE, in 2015, the European Commission designed its Circular Economy Action Plan, which, under the slogan "closing the loop", included development measures in areas such as production, consumption, waste management and the secondary raw materials market (Europa, 2015). In 2019, the European Commission published a report on implementing the Circular Economy Action Plan, in which the results and remaining challenges to achieving a circular, competitive, and carbon-neutral economy were set out. The report highlighted the achievement of its 54 actions, which had been or were being implemented. It showed that the CE is presented as a business opportunity, developing new business and market models. Among the remaining challenges to be met, the Commission's report pointed, among others, to the need for further efforts to implement the new waste legislation and promote markets for secondary raw materials (European Commission, 2019).

Continuing the efforts initiated in 2015, and with the declaration of climate emergency by the European Parliament, as well as the momentum of the European Green Pact driven by the Commission (Europeo et al., 2019), a second Circular Economy Action Plan was adopted in 2020 (Comisión Europea, 2020). This new initiative builds on the previous plan's results and several interlinked initiatives to shape a robust and sustainable product framework policy consistent with the waste hierarchy. The plan presents a framework with rigorous monitoring of sustainable product design, empowerment of consumers and public purchasers, and circularity in production processes. In addition, it implements the study of value chains in crucial production sectors such as electronics and ICT; batteries and vehicles; packaging; plastics; textiles; construction and buildings; and food, water, and nutrients. In this context, the Waste Framework Directive 2008/98/EC (UE, 2008), aimed at establishing a strategy to achieve a set of targets for the reduction of resource consumption as well as the elimination of waste to landfill in a sustainable perspective, i.e. one that is economically, socially and environmentally feasible. Following the adoption of the action mentioned above plans, this directive has been amended by Directive 850/2018 (Europeo et al., 2018) Waste Framework to include a series of guidelines regarding waste management and its potential consequences related to climate change, all within the framework of the CE. The new objectives include further restricting landfilling of waste, starting with waste streams subject to separate collection, such as plastics, metals, glass, paper, and biodegradable organic waste, known as bio-waste, which would bring clear environmental, economic and social benefits. In addition, the technical, environmental, or economic feasibility of recycling or other material or energy recovery processes should be considered when applying landfill restrictions.

In its move towards establishing an economic model appropriate to the current context, Spain has developed a series of regulatory and planning tools to address the necessary transformation process. In February 2018, the "Spanish Circular Economy Strategy. Spain 2030", which represents the essential strategic and action framework to facilitate and promote the transition towards the CE based on the collaboration between the General State Administration, the Autonomous Communities, local entities and the other agents involved, in particular, people producing and consuming goods (España and Economía, 2018). The final "Spain Circular 2030" strategy was approved in June 2020, aligning its objectives with Directive 850/2018. The plan sets a series of quantitative targets to be achieved by 2030, including a 30% reduction in the national consumption of materials about GDP; a 15% reduction in waste generated; an increase in reuse and preparation for reuse to 10% of municipal waste generated; and a 10% improvement in water efficiency (Española, 2020).

Concerning waste management, Law 7/2022, of 8 April, on waste and contaminated soils for a CE, which repeals the previous Law 22/2011, of 28 July, has as its primary objective the reduction in weight of 15% of waste generated in 2030 for 2010, limiting the use of plastics by reducing the marketing of plastics by 70% by 2030, restrictions on the disposal of waste in landfills, and obligations for waste managers to encourage reuse, recycling and recovery (Jefatura del Estado, 2022). This law follows the guidelines of Directive 850/2018 in order to ensure the correct application of the waste hierarchy, imposing a series of actions to be used as a priority order in waste prevention and management legislation and policy, according to the following order: prevention; preparation for reuse; recycling; another recovery, including energy recovery; and disposal.

At the regional level, Decree 131/2021, of 6 April, approves the Andalusian Integrated Waste Plan. Towards a Circular Economy in the 2030 Horizon (PIRec 2030). Among the main novelties of the plan are the separate collection of bio-waste, the preparation centers for reuse and the attention to the growing presence of plastic waste on the coast. Based on eight general objectives, a comprehensive plan to promote the CE is formulated, which develops three programs: prevention, management and awareness-raising, sensitization and communication. Municipal solid waste concentrates the most significant number of actions, aiming for a maximum of 10% landfill by 2035 and a minimum of 65% preparation for reuse and recycling (Junta de Andalucía, 2021).

Subsequently, as of 1 May 2023, the Andalusian Circular Economy Law came into force to create an appropriate regulatory framework for developing the CE in the Autonomous Community's sphere of competence. Among the most relevant aspects of its implementation is the implementation of the Andalusian Public Registry of life cycle analysis, with aspects related to this analysis in products, works and services. The regulation foresees that energy recovery activity projects will be considered business investments of strategic interest and also specifies the procedure for a recovered waste to be considered as a by-product for use in an industrial activity or process. Within the application of the CE to the integral water cycle, the use of reclaimed water is specifically addressed, with particular attention to bodies of water classified as being in poor condition and using sewage sludge (Andalucía, 2023).

2 SUSTAINABLE WASTEWATER MANAGEMENT

Water management has a relevant role in achieving the Sustainable Development Goals (SDGs) for the 2030 Agenda set by the United Nations; in fact, it directly impacts SDG 6: Clean Water and Sanitation. Moreover, the targets of this goal transcend the rest of the SDGs, as a goal can only be fully realized with the prior guarantee of the universal right to water. Schröder et al., (2018), in a study on the most significant relationships and synergies between CE guidelines and SDG targets, identified SDG 6 as one of them. In this work, Schröder et al., (2018) also argue that CE practices associated with closed-loop systems for wastewater recycling and reuse (Nick Jeffries, 2017) and sewage sludge recycling (Angelakis and Snyder, 2015) will be indispensable for achieving the SDG targets, in addition to some of those included in SDG14 target 14.1 on marine ecosystems.

Wastewater treatment plants (WWTPs) aim to reduce the pollutant load of water after use (whether urban, industrial or a mixture of both) before returning it to the natural environment, for which different physical, chemical, and biological treatments are used. In recent decades, the number of WWTPs has increased worldwide to comply with existing regulations and mitigate the aquatic environment's deterioration. Furthermore, this increase in WWTPs is expected to generate an increase in energy consumption, which already represents a significant percentage of a municipality's energy costs, ranging from 24 (Wu et al., 2021) to 44% (Cardoso et al., 2021). This situation can lead to a considerable increase in CO_2 emissions and a substantial increase in operating costs (Cardoso et al., 2021). The energy consumption of municipal WWTPs can vary significantly depending on internal factors such as on-site energy production from renewable sources (Lindtner et al., 2008), the technologies applied (Mamais et al., 2014) and their intensity; and external factors such as energy tariffs (Lindtner et al., 2008). This is why, as a productive system, it is essential to optimize the processes in WWTPs (Newhart et al., 2019), considering the new paradigm of the CE that seeks to redefine these facilities into biorefineries. This goal involves implementing new working models based on fundamental principles that include water reuse, energy self-sufficiency, and reducing and recovering waste generated during the production process. In this sense, existing infrastructures for wastewater management in industrialized countries' systems are not always adequate to support the CE, so they should be optimized by applying the principles mentioned above (Schröder et al., 2018).

Due to the high organic matter content in these processes, WWTPs also represent a rich source of chemical energy (Lindtner et al., 2008). Therefore, the redefinition of WWTPs into biorefineries necessarily involves waste-to-energy technologies, recognized by the "waste to energy" (WtE) concept, which could be applied to the waste generated in the different phases of wastewater treatment (screening waste, sand, fats and sludge). These technologies have been applied in WWTPs mainly for sludge management, which has allowed the production of biogas from the digestion and co-digestion of organic matter (Gandiglio et al., 2017), which are subsequently converted into electricity and heat; furthermore, biodiesel production (Liu et al., 2021), combustion (Liang et al., 2021) and co-combustion (Zhang et al., 2020), gasification (Migliaccio et al., 2021a) and pyrolysis (Agar et al., 2018) processes have also been realized as sustainable options for sludge treatment. In addition to sludge, there are also technologies for recycling sand and fats, generally focused on agriculture (Valdés López et al., 2021).

However, although sludge represents the majority of the waste generated in a WWTP and scientific studies are mainly focused on its treatment, other solid wastes, such as screening waste, are generated in the pre-treatment. This waste needs to be adequately managed within the regulatory framework outlined above; therefore, there is an opportunity to harness these energy sources during the wastewater treatment process (Gandiglio et al., 2017).

3 SCREENING WASTE

At the inlet of the WWTP, the wastewater contains a large amount of bulky material that must be removed to allow the following stages of treatment, which generates a waste production, both in the inlet works and in the pre-treatment phase. This waste fraction consists, among others, of a heterogeneous mixture of paper, sanitary waste and plastics and is classified under European Waste Catalogue (ECW) code 19 08 01 (Consejo del Parlamento Europeo, 2014) and described as screening waste under chapter 19 Wastes from waste treatment plants, off-site wastewater treatment plants and the preparation of water for human consumption and water for industrial use, specifically under subchapter 19 08 Wastes from wastewater treatment plants not otherwise specified.

The amount of waste generated in a WWTP depends directly on the type of network in place in the municipality. Rain is also a meteorological factor that, in the case of unitary sewage networks, produces a dragging and cleaning of the waste deposited in dry weather, on the public road and in the sewage network itself; thus, on rainy days, the production of screening waste can be up to 50% higher than on dry days (Canler and Perret, 2004). The amount of waste from desludging also depends on the equipment, the pretreatment and the screen aperture of the WWTP thus, according to Le Hyaric (Le Hyaric et al., 2009), there is a big difference between the 6 mm screen aperture, which provides a generation rate of 0.08 kg/year.heq, and 3 mm, which increases it to 0.15 kg/year.heq, both in dry matter. Finally, it should be added that the compaction process frequently used produces a decrease in the volume of waste generated (Clay et al., 1996).

Screening waste accounts for around 2% of the total waste generated during the water treatment (Le Hyaric et al., 2010), with values, in dry matter ranging from 0.08 kg/year.heq (Le Hyaric et al., 2010) to 1.1 kg/year.heq (Kaless et al., 2016). This variability in the data shows the complexity in the analysis and characterization of this type of waste, with very heterogeneous production results in the different studies carried out. In addition, the low production of waste from screening waste is evident when compared to the generation of other fractions produced in the treatment process, such as sludge, which in Andalusia is between 19 and 31 kg/year.heq, in dry matter (Granados González, 2015).

In general terms, the fractions that make up the waste are mainly made up of sanitary textiles, paper and cardboard, vegetables, plastics and small percentages of fines or metallic materials (Gregor et al., 2013; Le Hyaric et al., 2009; Naud et al., 2007). The predominance of sanitary textiles, whose presence has progressively increased over the years with changing societal habits (Wid and Horan, 2016) and now exceeds an average value of 50%, is one of its characteristics. In 1996, the study carried out by Clay et al. reflected percentages of around 25% of sanitary textiles, a figure that can now triple (Wid and Horan, 2016) and reach a maximum of 87.1% (Le Hyaric et al., 2009). Paper and vegetables, potentially biodegradable material, have a relevant volume within the waste, although there are also significant differences between the studies consulted, ranging from 1.3 to 13.1% (Wid and Horan, 2016). The presence of fines, i.e. particles less than 20 mm in diameter that are very difficult to separate, is between 7.6 and 15.2% (Le Hyaric et al., 2009). Finally, a small proportion of plastics, metals and non-biodegradable materials can also be observed, which do not exceed 3.1 to 9.7% (Le Hyaric et al., 2009; Wid and Horan, 2016).

Regarding physical characteristics, the water content directly influences, among others, the calorific value of the waste and its study is essential for its correct management (Arrechea et al., 2009). Screening waste shows a high water content, even after compaction, ranging from 70-85% (Cadavid-Rodriguez and Horan, 2012; Kaless et al., 2016; Wid and Horan, 2016). The value of this parameter depends on factors such as climate, type of screening or the presence or not of a previous compaction process; in the latter case, the volume reduction process increases the density of the residue to values usually ranging between 0.6 and 1 kg/L (Naud et al., 2007; Le Hyaric et al., 2009; F. Valiron, 1994), as well as contributing to the decrease of part of the water present in the residues (Le Hyaric et al., 2009). Suppose these values are compared with those obtained in municipal solid waste (MSW) management. In that case, the rejects from mechanical biological treatment (MBT), the most widely used for further processing in anaerobic digestion (Pires et al., 2011), show substantially lower water content values with values close to 30% (Edo-Alcón et al., 2016). Other physical parameters that allow the quality of the product to be assessed are the volatile solids and ash content. In the case of volatile matter, values of around 90% of total solids are shown (Le Hyaric et al., 2010; Cadavid-Rodriguez and Horan, 2012; Wid and Horan, 2018). As for the ash content, data were found in the study on anaerobic digestion for screening waste developed by Cadavid-Rodriguez & Horan (2012), which recorded average values of 2.1%.

About chemical characterization, elemental analysis to determine carbon, hydrogen, nitrogen and sulphur content is shared in MSW characterization (Edo-Alcón et al., 2016; Nasrullah et al., 2015). In the literature on MSW, such determinations have only been found in the study by Cadavid - Rodriguez & Horan (2012), which only showed values for carbon and nitrogen content, with 48.5% [46.4 - 50.2] and 2.5% [2.3 - 2.9] respectively. The organic matter content is the parameter that shows the highest heterogeneity in the studies consulted, with values ranging from 77 to 88% and a percentage of oxidizable organic matter around 45% [37 - 51] (Le Hyaric et al., 2009). Sidwick (1991) explains this by factors related to meteorological phenomena that can vary the presence of plants and mineral materials, the type and length of the sewage network, which can benefit the elimination of organic matter before it reaches the WWTP, or the characteristics of the origin of the wastewater (urban area, industrial, tourist activity...). The values collected in the studies consulted show values for the lower heating value (LHV), on a dry basis, of the residue from desludging that vary between 17.70 MJ/kg (Sidwick, 1984) and 20.90 MJ/kg (Canler and Perret, 2004), figures similar to those of the MSW reject fraction (Gallardo et al., 2014; Bessi et al., 2016).

Although a small percentage of waste from landfill is incinerated, it is currently most commonly disposed of in landfills (Wid and Horan, 2018; Clay et al., 1996). This solution, in addition to generating environmental problems, requires a significant transport cost given its high moisture content; to this must be added the potential problems regarding the admission of the waste to landfill due to its high organic matter content (Cadavid-Rodriguez and Horan, 2012). On the other hand, waste disposal by landfill is bound to disappear since, with the new restrictions set out in Directive 850/2018 (Europeo et al., 2018) in 2035, the amount by weight of municipal waste landfilled will have to be reduced to a maximum of 10% (MITECO - Ministerio para la Transición Ecológica y el Reto Demográfico, 2020), so it is necessary to seek alternatives to the current landfill disposal of waste from WWTP and thus contribute to the achievement of established objectives. Finally, in the case of Spain, the approval of the new tax on landfilling, incineration, and co-incineration of waste will imply an extra cost of €40 per ton for the disposal of landfill disposal of landfilled landfill waste.

The low percentage of production represented by the screening waste in a WWTP and their classification as non-hazardous waste according to the LER has been one of the main reasons scientific research on them has received little interest to date. This is evidenced by the few papers published on this subject, which mainly focus on the application of anaerobic digestion processes that allow the production of biogas, specifically for these wastes and by incorporating them into the anaerobic co-digestion system of the WWTP. The work reviewed has shown that the high organic matter content of the screening waste fraction suggests that anaerobic digestion can be a viable method of recycling and energy recovery (Md Rahman et al., 2018). In order to analyze this type of process for the screening waste, Kaless (2017) determined the chemical oxygen demand (COD) of the waste, concluding that it could be sent to an anaerobic digestion process or as a carbon source for denitrification during the water treatment process. These solutions could significantly reduce the carbon footprint of this waste from landfill disposal;

however, the studies consulted have also shown that the presence of plastics and textiles makes it challenging to incorporate the waste into digesters (Cadavid-Rodriguez and Horan, 2012). Consequently, the implementation of a dedicated digester for this waste at the industrial level, according to Le Hyaric (2010), was not found to be economically viable. Therefore, results have been reported showing the option of co-digestion as a potential alternative to the treatment of screening waste, with a stabilized biogas production with a diversity of methane contents ranging from 0.33 m^3/kg volatile solids (SV) (Cadavid-Rodriguez and Horan, 2012) to 0.51 and 0.62 m^3/kg (SV) for Le Hyaric (2010).

The literature reviewed on waste from landfill opens up the possibility of the feasibility of producing a fuel from these wastes. Due to its composition and LHV, it could be assimilated to MSW scrap to study its energy recovery through thermochemical processes (Dong et al., 2010) within the framework of WtE technologies. However, conclusive scientific studies have yet to be identified on this point.

4 OBJECTIVES

For all of the above reasons, searching for alternatives to the current disposal of landfill waste is critical to sustainable development in wastewater management. Although there are studies on this waste, they do not focus on its possible valorization as a solid fuel. Consequently, the main objective of this research was to study the technical, economic and environmental feasibility of the energetic valorization of waste from wastewater treatment plants using SRF production. To achieve this, the following secondary objectives have been defined:

- Literature review on the application of WtE technologies for waste from WWTP.
- Study of the production and characterization of the screening waste.

- Production and characterization at laboratory scale of SRF from screening waste.
- Economic feasibility study for the implementation of the SRF production process.
- Analysis of the environmental impact derived from the potential production of SRF.
- Application of thermochemical technologies for the recovery of the SRF produced.

For the development of this work, we have used the waste produced in the Biofactoría Sur of Granada, managed by the Municipal Water Supply and Sanitation Company of Granada, S.A. (EMASAGRA), which treats more than 18 million m3 per year of wastewater from about 425,000 equivalent inhabitants. This facility has already achieved its energy self-sufficiency objectives; in 2017, it achieved an energy self-consumption value of 82.5%; in April 2018, it reached a maximum of more than 100% self-sufficiency; and in 2019, it achieved average annual energy self-sufficiency data, which has led it to become a benchmark in Europe in the water management sector. In addition, Biofactoría Sur has a roadmap to facilitate the recycling and recovery of the waste generated in the process, aiming for "Zero energy, zero waste" in the coming years. To this end, all of the sludge, sand and grease generated in the plant are already recycled in agricultural applications, with the only thing left to do at present being to find a way to recycle and recover the screening waste in the search for zero waste.

CHAPTER 1

WASTE TO ENERGY FROM MUNICIPAL WASTEWATER TREATMENT PLANTS: A SCIENCE MAPPING[1]

1 INTRODUCTION

Recently, the number of WWTPs has increased around the world and additionally, more advanced, and energy-intensive treatment processes will be adopted in the following decades (Di Fraia et al., 2018). As a result, WWTPs are expected to increase energy consumption (Wu et al., 2021), varying significantly depending on several internal and external factors (Lindtner et al., 2008). As a consequence, reducing energy consumption and increasing the efficiency of energy production are both required thanks to process optimization (Gandiglio et al., 2017). Besides, because of the high content on organic matter, WWTPs are also a rich source of chemical energy that is about 9.3 times higher than the energy necessary to treat wastewater in these facilities (Frijns et al., 2013). It can be converted to

[1]The results shown in this Chapter were presented in: De La Torre Bayo, J. J.; Martín Pascual, J.; Torres Rojo, J. C.; Zamorano Toro, M. Waste to Energy from Municipal Wastewater Treatment Plants: A Science Mapping. Sustainability 2022, 14 (24), 16871. https://doi.org/10.3390/su142416871

energy for use during wastewater treatment (Newhart et al., 2019), especially from waste produced during the treatment process.

WtE technologies encompass a range of approaches aimed at treating waste and regenerating energy in the form of heat, electricity, or alternative biofuels, including biogas, syngas, bioethanol, biodiesel, and biohydrogen (Boloy et al., 2021). To enable the widespread adoption of these sustainable technologies, the conversion processes involved, such as pyrolysis, gasification, transesterification, composting, and anaerobic digestion, need to exhibit high efficiency and durability over extended periods, surpassing the limitations of traditional conversion methods like incineration (Kumar et al., 2023). Some of these technologies have been applied to produce energy waste produced from municipal WWTPs (Gandiglio et al., 2017), mainly for application in sludge treatment (Migliaccio et al., 2021b).

In this context, and since no similar reviews have been found, the objective of this Chapter is to analyze the scientific literature to identify the evolution of technologies applied to produce energy from municipal WWTPs. To do that, a science mapping approach has been applied because of its objective criteria for evaluating the work carried out by researchers (Noyons and Van Raan, 1998) and a macroscopic overview of large amounts of academic literature (Nunen et al., 2018). This study will contribute to the existing body of knowledge by highlighting the trends and patterns in the research field, establishing its research themes and mapping researcher networks. From the results of bibliometrics and in comparison, with the more exhaustive analysis of research, it is possible to know the current knowledge gaps from which future lines of research can be identified.

2 MATERIALS AND METHODOLOGY

This paper analyzes the available research on WtE from municipal WWTPs to result in the generation of mapping of the most relevant actors in the field in terms of sources, authors, countries, and papers, as well as its evolution, taking into account themes in terms of keywords. To do that, the methodology and tools applied were carried out in three main stages, as shown in Figure 3 and described below.

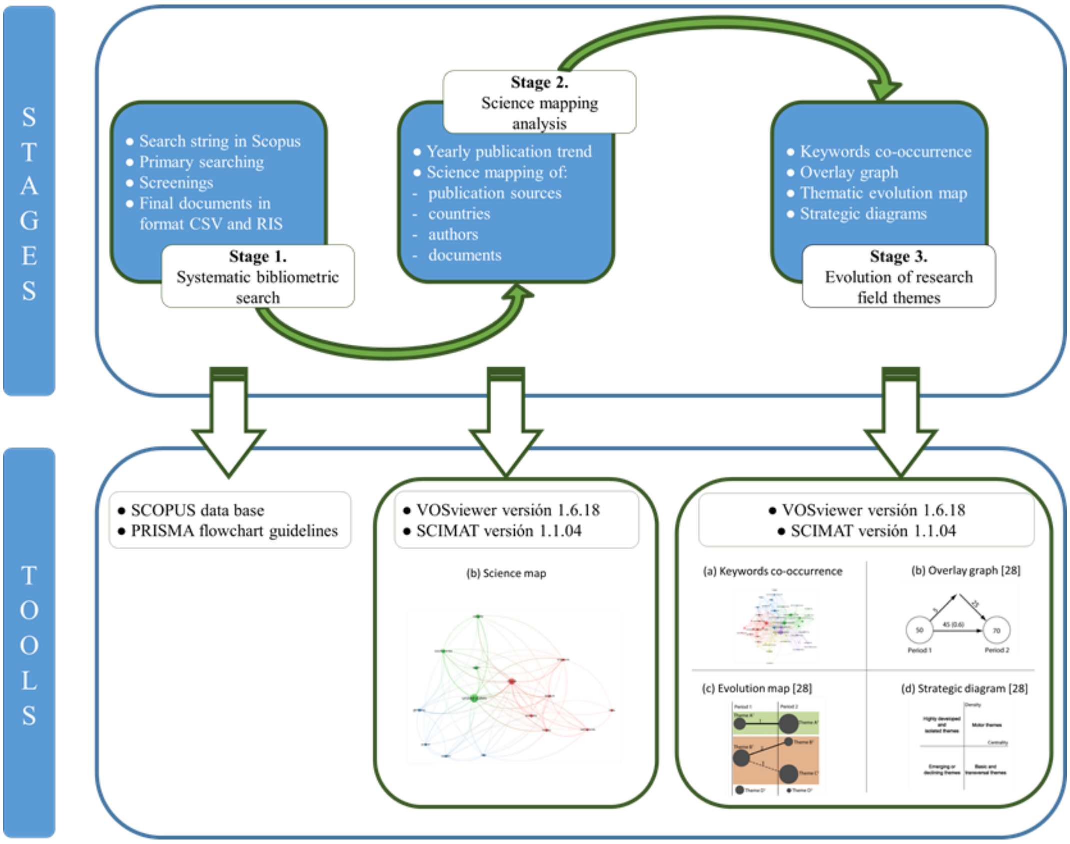


Figure 3. Summary of the methodology and tools applied (Cobo et al., 2011)

2.1 STAGE 1. SYSTEMATIC BIBLIOMETRIC SEARCH

Scientific databases are necessary to choose documents in the research field. Because the Scopus database has a broader bibliometric scope and the most current data compared to Web of Science, the documents for the present evaluation were retrieved from Scopus (Aghaei Chadegani et al., 2013; Meho, 2019). A data refinement of the total number of primary searches for documents was approached according to the PRISMA flowchart guidelines to remove documents published in 2022 and that did not fall within the scope of the review, as well as to limit the research to journals and conference documents in English. The resulting papers were stored in both formats developed by comma-separated values (CSV) files and

Research Information Systems (RIS) for further analysis of the retrieved data using VOSviewer V1.6.18 and Science Mapping Analysis Software Tool (SciMAT V1.1.04), respectively. VOSviewer is a freely available open-source software tool that is commonly employed in a range of sectors and is highly recommended for creating maps (Yang et al., 2022). SciMAT is an open-source science mapping software tool that is based on a longitudinal science mapping approach. It incorporates methods, algorithms, and measures for all the steps in the science mapping workflow, from preprocessing to the visualization of the results (Cobo et al., 2011).

2.2 Stage 2. Science mapping analysis

The science mapping analysis of documents included:

- The yearly publication trend, including the total documents per year and the accumulative papers during the time horizon, was analyzed. The categorization of the horizon time into different subperiods was also included for better knowledge of the publication trend.
- Science mapping of publication sources, countries, authors, and documents. VOSviewer was applied to produce science maps (Figure3a) and construct tables with statistical values. A comprehensive quantitative analysis according to the number of documents and citations for each item was applied but also in terms of the value of the normal citation, average publication year, average citation, and average normalized citations (Kawshalya et al., 2020a). The normal citation is defined as the citation of all the articles within the same journal, author, or country; the average publication year is the average publication year of the articles; the average citation is the total citations per article; finally, the average normal citation is defined as the total number of citations divided by the average number of citations published in the same year and it is used to correct the misinterpretation that the old articles have more time for more citations than

the new ones (Kawshalya et al., 2020b). In networks generated, the size of nodes is related to their repercussion in terms of the number of documents, citations, or average normal citation; besides, the thickness and the colors of the linking lines indicate the inter-relatedness among them (Yang et al., 2022).

2.3 Evolution of research field themes

Finally, the evolution of research field themes was analyzed in terms of mapping of the co-occurrence of keywords (Figure 3b) generated with VOSviewer and the overlay graph (Figure 3c), thematic evolution map (Figure 1d), and strategic diagrams (Figure 3e) generated with SCIMAT. The overlay graph represents the number of items shared by different time periods in the time horizon. The keywords co-occurrence diagrams were used to identify the relationship and the connectedness of the articles. The evolution map rep-resents the number of documents associated with each theme and the relationship between them. Finally, the strategic diagrams represent themes. In terms of the number of documents or h-index, showing motors, highly developed and isolated, emerging, or declining and basic research topics (Cobo et al., 2011).

3 Results

The most relevant results obtained in the three stages defined sequence in previous section are summarized, analyzed, and discussed below.

3.1 Stage 1. Systematic bibliometric search

The Scopus database was searched for bibliometric data in July 2022 using the following search string, according to the combination of keywords in the title, abstract, or keywords: (*municipal wastewater* OR *municipal waste water* OR *domestic wastewater* OR *domestic waste water*) AND (*waste to energy* OR *WtE* OR *waste-to-*

energy OR *energy valorisation* OR *energy valorization* OR *energy production* OR *energy recuperation* OR *energy recovery*).

The total number of primary searches for documents in Scopus was 468. The data refinement according to the PRISMA flowchart guidelines to exclude documents published in 2022, those that did not fall within the scope of the review, as well as to limit the research to journals and conference documents in English, resulted in a total of 337 documents that were finally selected for the analysis (Figure 4). They were stored in both CSV and RIS file formats for further analysis in the next section.

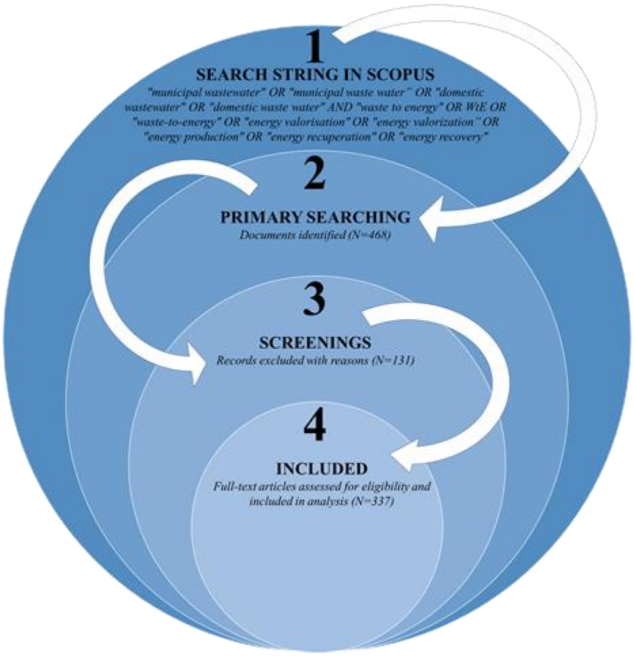


Figure 4. PRISMA flowchart.

3.2 Stage 2. Science mapping analysis

The yearly publication trend, science mapping, and evolution of research field themes were analyzed using SciMAT and VOSviewer. The most relevant results and their discussion are summarized below.

3.2.1 Yearly publication trend

The publications and citations of the documents included in the analysis of a considered field depict the developments and patterns in the research (Kawshalya et al., 2020b). As a consequence, the yearly publication trend, including the accumulative papers during the time horizon for the searching strings defined, is included in Figure 5. The first article was found in the year 1979, so the horizon time of this research was defined from this year to 2021. A preliminary analysis of Figure 5 shows a gradual increase in the number of publications, especially from 2008. According to the growth pattern, the horizon time was classified in the following three subperiods:

- Initial phase, or 1st subperiod. From 1979 to 2008, a total of 20 documents were published. This period is characterized by a very low number of documents per year; in fact, none or only 1 or 2 documents per year were published in most of the period, although in the last years, a slight increase to 5 papers was observed.
- Active phase with relative growth, or 2nd subperiod. From 2009 to 2015, a total of 100 documents were published in seven years. A significant increase in the number of documents was observed in comparison with the previous phase and coincided with the approval of the Directive 2008/98/EC on waste. It establishes a legal framework for treating waste in the EU designed to protect the environment and human health by emphasizing the importance of proper waste management, recovery, and recycling techniques to reduce pressure on resources and improve their use (Une-En, 2011); besides, it reinforces the waste hierarchy, which includes, in this order: prevention, preparing for reuse, recycling, other recovery (including energy recovery), and finally, disposal (Une-En, 2011). As a consequence, the development of technologies to produce energy from waste began to grow faster.

- Active phase with high growth or 3rd subperiod. From 2016 to 2021, a total of 217 documents were published in 6 years. On September 2015, the 2030 Agenda for Sustainable Development was adopted by the United Nations General Assembly with the aim of stimulating action in five critical areas: people, planet, prosperity, peace and partnership. This document contains 17 SDGs and 169 targets associated with achieving these goals by the year 2030, and it was driven the approval of policy frameworks around the world, for example the first EU action plan for the CE (From et al., 2015). In all these policy strategies different waste treatment operations classified as WtE processes are essential to fulfill the objectives included in them, so, as result, a very significant growth of this research field has been identified and it has been reflected in the increase in publications.

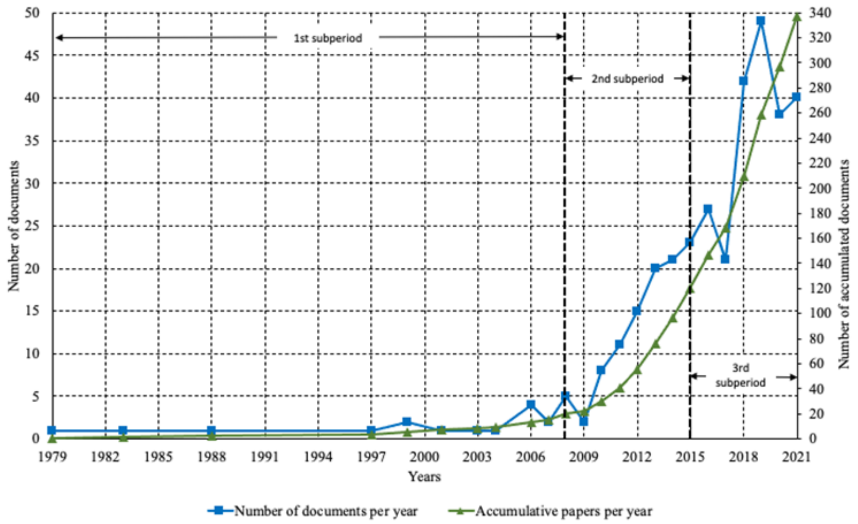
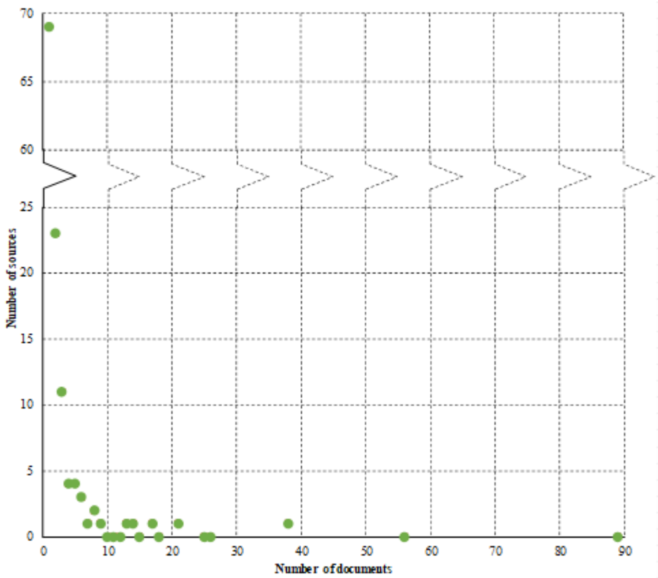


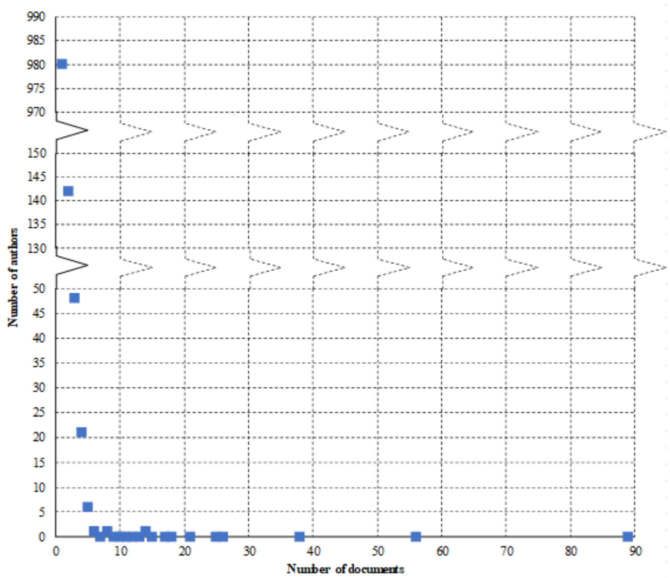
Figure 5. Number of documents in the horizon-time (1979–2021). Documents per year and accumulated documents

3.2.2 Science mapping

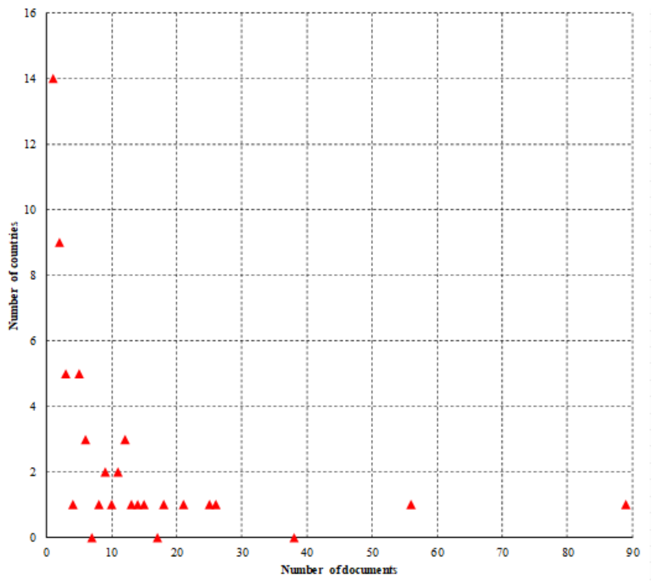
In a preliminary analysis, a total of 123, 1200, and 55 sources, authors, and countries, respectively, were included in Figure 6, which shows their relationship with the number of documents published. It shows a high concentration of the number of documents in the lower values of number of sources, authors and countries; specifically, 69 sources that represent 59% of them, 81.7% of the authors (980 authors), and 25.5% of the countries (14 countries) have published only one document in the field. These results highlight the diversity and breadth of the research field. As a result, in the case of the authors, Lotka's Law, which establishes that the number of those who have published a number of works is a fixed relationship with a constant number of authors who have published one or very few articles, is not fulfilled (Pao, 1985). To identify sources, authors, countries, and documents with a higher relevance in the field, VOSviewer science maps were used and the results are summarized and discussed below.



a)



b)



c)

Figure 6. Relationship between the number of documents and (a) the number of sources, (b) authors, and (c) countries.

- Publication sources mapping

In the case of sources mapping, journals with a minimum number of 8 documents of a source were identified, resulting in the 8 sources summarized in Table 1 in terms of publication count, total citations, and average normalized citations. The journal Bioresource Technology, which promotes research to support bioresource development, processing, and utilization in a sustainable manner, clearly leads the ranking with 38 documents, 3227 total citations, and an average normalized citations index of 1.5648; on the other hand, Water Research, which publishes studies on all aspects of the science and technology of the anthropogenic water cycle, water quality, and its management worldwide, leads the ranking in terms of average normalized citations (1.9456), with only 17 documents published (3^{rd} position in this ranking) but 1153 citations (2nd position in this ranking). Both journals are included in the top 3 ranking in terms of the number of documents, total citations, and average normal citations. Water Science and Technology, focused on all fields relevant to water research, is included in both top 3 rankings in terms of average normal citations ranking; however, it is only in the 8th position in the case of the number of documents and total citations. Finally, Journal of Cleaner

Production, a transdisciplinary journal with 9 documents published and 374 citations that is focused on cleaner production, environmental, and sustainability research, is in the 3rd position in average normal citations ranking, although it is in the 6th position in the other rankings.

Table 1. List of leading sources of publications in terms of publication count, total citations, and average normalized citations.

Position	Source	Number of documents	Total citations	Normal citations	Average citations	Average normalized citations
			Publication count			
1	Bioresource Technology	38	3227	59.4616	84.9211	1.5648
2	Water Science and Technology	21	498	6.9348	23.7143	0.3302
3	Water Research	17	1153	33.0759	67.8235	1.9456
4	Science of the Total Environment	14	344	13.9512	24.5714	0.9965
5	Journal of Environmental Management	13	393	17.8791	30.2308	1.3753
6	Journal of Cleaner Production	9	374	12.6744	41.5556	1.4083
7	Chemosphere	8	394	9.6039	49.25	1.2005
8	Chemical Engineering Journal	8	317	9.5925	39.625	1.1991
			Total citations			
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Table 2 summarizes the categories of the sources leading the ranking. A total of 9 categories were identified, showing the transdisciplinary of the field, which has been analyzed from different points of view. Nevertheless, ENVIRONMENTAL, SCIENCES (F) and ENGINEERING, ENVIRONMENTAL (E) are the most representative with 6 and 4 journals included in them, respectively, which emphasizes the environmental dimension of the field, as well as the use of the engineering to develop solutions to recover the energy from waste from municipal wastewater treatment plants. On the other hand, it is important to highlight that although the category ENERGY & FUEL (C), which is directly related to the production of energy from the waste of municipal wastewater treatment plants, is only identified in one of the journals, this one is the journal Bioresource Technology, with the larger number of documents published and total citations.

Table 2. Categories of the sources leading the ranking.

	A	B	C	D	E	F	G	H	I
Bioresource Technology	✓	✓	✓						
Chemical Engineering Journal				✓	✓				
Chemosphere						✓			
Journal of Cleaner Production					✓	✓		✓	
Journal of Environmental Management						✓	✓		
Science of the Total Environment						✓			
Water Research					✓	✓			✓
Water Science and Technology					✓	✓			✓
TOTAL	1	1	1	1	4	6	1	1	2

A. AGRICULTURAL ENGINEERING
B. BIOTECHNOLOGY & APPLIED MICROBIOLOGY
C. ENERGY & FUELS
D. ENGINEERING, CHEMICAL
E. ENGINEERING, ENVIRONMENTAL
F. ENVIRONMENTAL, SCIENCES
G. ENVIRONMENTAL STUDIES
H. GREEN & SUSTAINABLE SCIENCE & TECHNOLOGY
I. WATER RESOURCES

Finally, Figure 7 depicts a network visualization of the 8 sources included in the ranking in terms of citations. All of them are connected and the dimensions of the cycle imply the source's contribution; a bigger dimension signifies more influence in terms of the number of citations. For example, Bioresources Technologies has the larger circle sizes than the other journals, which means that this journal has the higher impact on the research field in terms of total citations. Additionally, circles with the same color indicate groups of associated sources discovered using VOSviewer analysis. Three groups or clusters were detected and denoted by the distinct colors red, green, and blue, with 4, 2, and 2 journals in each one, respectively. Clusters are constructed using the scope of research outlets or the number of times they are co-cited, and the connections between sources that are close together are stronger than those amongst frames that are farther apart (Yang et al., 2022). Thus, Bioresources Technologies (Cluster 2 green) shows the strongest connection with Water Research (Cluster 1 red).

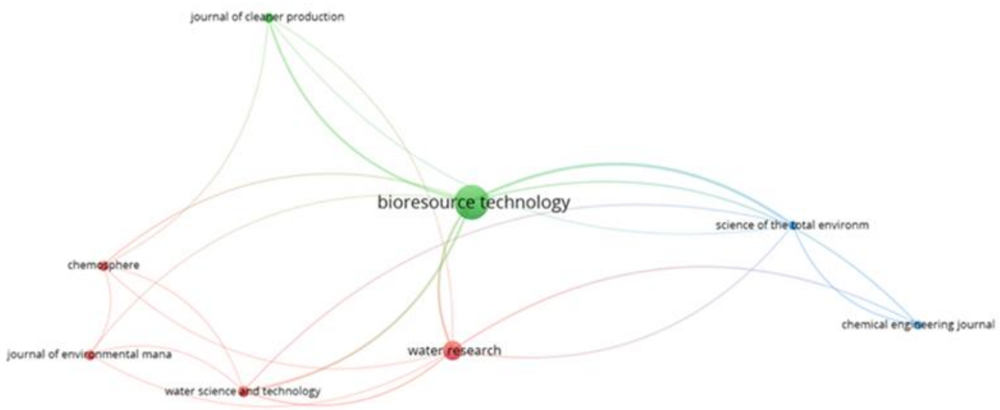


Figure 7. Network visualization of leading sources connected in terms of the number of documents.

- Countries mapping

Countries with a minimum number of 10 documents were identified, resulting in a total of 15 countries represented in the network visualization of Figure 8 in terms of total documents. Three groups or clusters were detected, denoted by the distinct colors red (Cluster 1), green (Cluster 2), and blue (Cluster 3) with 7, 4, and 4 countries in each one, respectively. An overlay visualization of the previous network shows that Belgium and Sweden, in purple color, are the countries that have been working on the field for the longest time, but countries in yellow color (i.e., Singapore, India, or China) have most recently joined. Based on the graphical description of the occupied countries, the larger size of the circles of the United States and China makes clear the leadership of these countries in terms of total documents, as mentioned above.

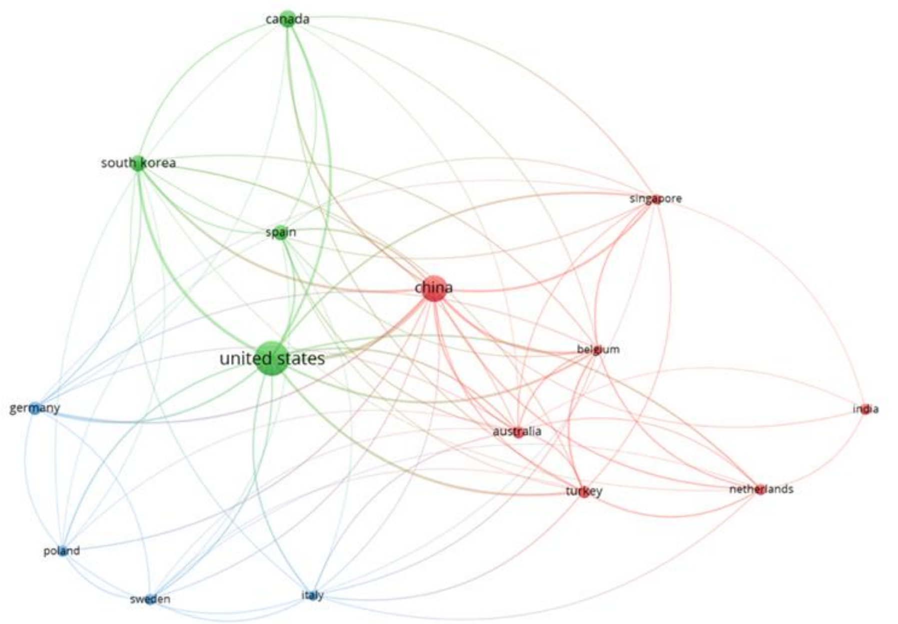


Figure 8. Network visualization of the countries included in the top 10 ranking lists in terms of total documents.

Table 3 shows the top 5 rankings of these countries in terms of the number of documents, total citations, and average normalized citations. A total of 9 countries are included in the three top lists. As shown in Figure 9, United States and China lead the ranking in terms of the number of documents and citations; in the case of the average normalized citations ranking, Australia and Netherlands are the leaders. Besides, China is the only country that is included in the three rankings, so it is considered the country with the stronger influence in the research on the field, followed by United States, South Korea, Australia, and Netherlands, which are included in two of the rankings; finally, the other 3 countries are included only in one of the lists (Spain, Singapore, and Belgium).

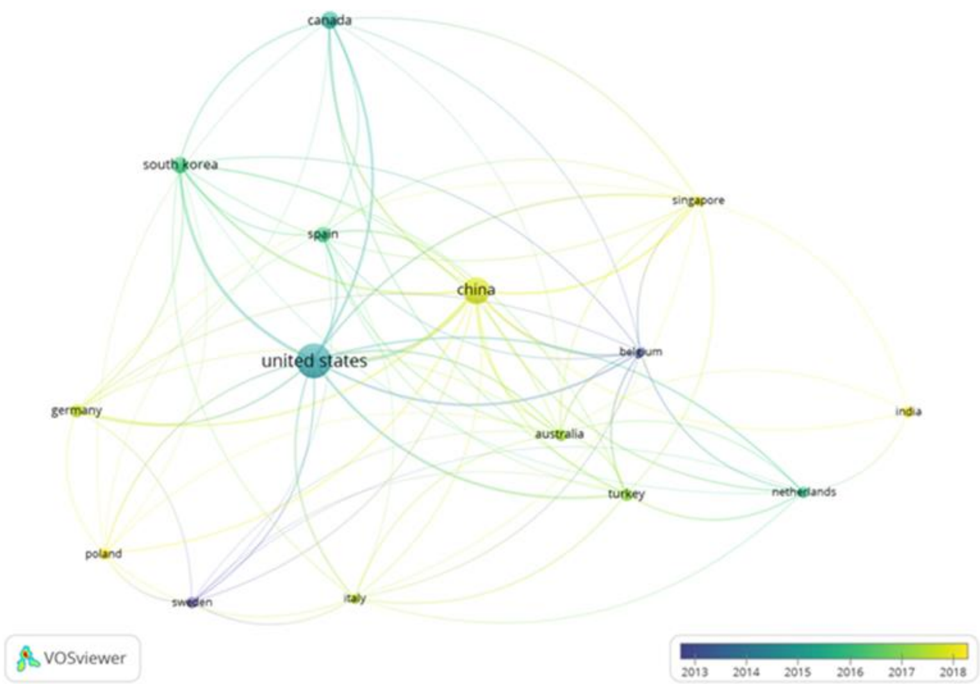


Figure 9. Overlay visualization of the countries included in the top 10 ranking lists.

Table 3. List of the leading five countries in terms of the publication count, total citations, and average normalized citations.

Position	Country	Number of documents	Total citations	Normal citations	Average citations	Average normailzed citations
			Publication count			
1	United States	89	5790	108.2587	65.0562	1.2164
2	China	56	2234	73.9738	39.8929	1.321
3	Canada	26	767	21.4767	29.5	0.826
4	South Korea	25	996	22.433	39.84	0.8973
5	Spain	21	578	17.6773	27.5238	0.8418
			Total citations			
1	United States	89	5790	108.2587	65.0562	1.2164
2	China	56	2234	73.9738	39.8929	1.321
3	Australia	15	1092	25.9684	72.8	1.7312
4	Netherlands	12	1073	18.2619	89.4167	1.5218
5	South Korea	25	996	22.433	39.84	0.8973
			Average normalized citations			
1	Australia	15	1092	25.9684	2017.1333	72.8
2	Netherlands	12	1073	18.2619	2015.8333	89.4167
3	Singapore	11	429	14.7389	2017.7273	39
4	China	56	2234	73.9738	2017.6429	39.8929
5	Belgium	10	785	13.1196	2013.4	78.5

- Authors mapping

Authors with a minimum number of 5 documents were identified, resulting in a total of 9 authors, 8 of them connected and represented in the network visualization in Figure 10 in terms of total citations. Two clusters can be identified, denoted by the color red (Cluster 1) and green (Cluster 2) with 5 and 3 in each one, respectively. Based on the graphical description of the active authors, the larger size of the circles of Liu Y. denoted the one with a stronger influence in terms of the number of documents. Table 4 summarizes the top 5 authors in terms of the publication count, total citations, and average normalized citations, and it includes 6 authors in total. Ersahin M.E. is included only in the ranking of total documents; Logan B.E. is included both in the total citations and average normalized citations; finally, Liu Y., Gu J., Zhang X., and De Clippeleir H. are in the three rankings, so they are considered the most influenced authors. The themes analyzed by these authors are centered on opportunities of technologies for achieving energy-efficient sewage treatment by minimizing energy consumption, for example, using nitrogen removal by anaerobic treatment or an integrated anaerobic moving bed biofilm reactor (AMBBR) and integrated fixed-biofilm and activated sludge sequencing batch reactor (IFAS-SBR) process but also the recovery of energy from sludge applying anaerobic digestion or nutrient removal using microbial fuel cell techniques.

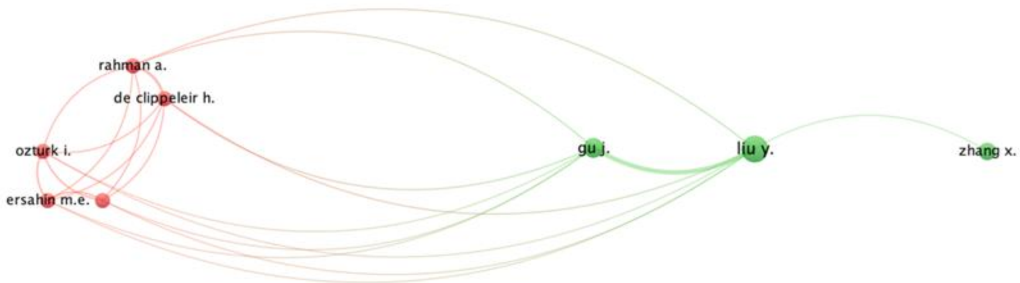


Figure 10. Network visualization of authors with a minimum number of five documents and connections between them denoting total citations.

Table 4. List of ranking countries in terms of publication count, total citations, and average normalized citations.

Position	Author	Number of documents	Total citations	Normal citations	Average citations	Average normalized citations
			Publication count			
1	Liu Y.	14	667	22.1932	47.6429	1.5852
2	Gu J.	8	412	14.0843	51.5	1.7605
3	Zhang X.	6	366	10.4878	61	1.748
4	De Clippeleir H.	5	239	5.2834	47.8	1.0567
5	Ersahin M.E.	5	127	3.9343	25.4	0.7869
			Total citations			
1	Logan B.E.	5	758	7.3728	151.6	1.4746
2	Liu Y.	14	667	22.1932	47.6429	1.5852
3	Gu J.	8	412	14.0843	51.5	1.7605
4	Zhang X.	6	366	10.4878	61	1.748
5	De Clippeleir H.	5	239	5.2834	47.8	1.0567
			Average normalized citations			
1	Gu J.	8	412	14.0843	51.5	1.7605
2	Zhang X.	6	366	10.4878	61	1.748
3	Liu Y.	14	667	22.1932	47.6429	1.5852
4	Logan B.E.	5	758	7.3728	151.6	1.4746
5	De Clippeleir H.	5	239	5.2834	47.8	1.0567

To analyze the collaboration between authors, the mapping of co-authorship by an author with a minimum of 3 documents and 10 citations per author has been done. Figure 11 shows the large number of small collaboration groups in which the 78 authors identified are working; it is possible to highlight the greater collaboration network that includes clusters in red, green, purple, and yellow color and 32 authors. With the objective of identifying this collaboration in terms of countries, the mapping of co-authorship by countries with a minimum of 10 documents (Figure 12) was developed, resulting in 15 countries that work together in the research field. Important collaborations between countries, which include 4 scientific communities, were identified, highlighting the one in red color with 6 countries, including Poland, Germany, Spain, Belgium, Netherland, and Sweden.

- Articles mapping

Table 5 and Table 6 summarize the list of top 10 documents in terms of total citations and normal citations, respectively, including a total of 16 documents, reflecting the document with the higher impact in the research field. Four of them are included in both rankings, so they could be considered the papers with the higher contribution to the research field. These documents have been published in 11 journals, highlighting *Bioresource Technology and Water Research*, with 4 and 3 papers published, respectively, and both of them identified as the most relevant in sources mapping. On the other hand, 8 of the 16 papers included in both top 10 documents rankings belong to the second subperiod or active phase with relative growth, 6 of them in the third one, and only 2 of them to the initial phase. Finally, the titles of the papers show different themes related to the research field and including WtE technologies as anaerobic treatments, biofuel production, microbial fuel cells, or biomass from algae mass cultivation integrated in wastewater treatment.

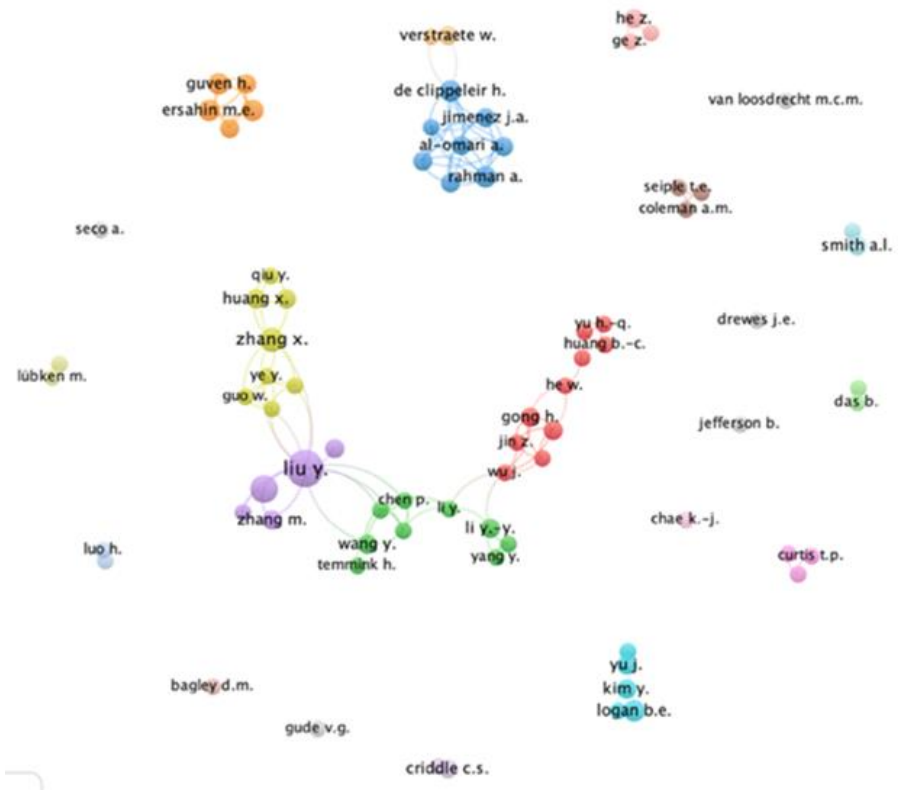


Figure 11. Network visualization co-authorship by authors with a minimum of 3 documents and 10 citations.

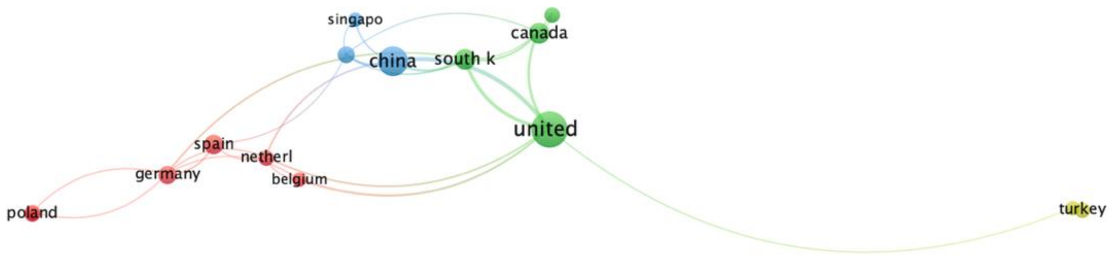


Figure 12. Network visualization co-authorship by countries with a minimum of 10 documents.

Table 5. List of top 10 documents in terms of total citations.

Position	Documents	Authors	Total citations	Normal citations	Publication year	Journal
1	Characterization of a microalga Chlorella sp. well adapted to highly concentrated municipal wastewater for nutrient removal and biodiesel production	Yecong Li, Yi-Feng Chen, Paul Chen, Min Min, Wenguang Zhou, Blanca Martinez, Jun Zhub, Roger Ruan	540	6.0923	2011	Bioresource Technology 102(8), 5138-5144
2	Sewage sludge as a biomass resource for the production of energy: Overview and assessment of the various options	Wim Rulkes	400	2.9718	2008	Energy Fuels, 22(1), 9–15
3	Perspectives on anaerobic membrane bioreactor treatment of domestic wastewater: A critical review	Adam L. Smith, Lauren B. Stadler, Nancy G. Love, Steven J. Skerlos, Lutgarde Raskin	316	3.4548	2012	Bioresource Technology 122, 149-159
4	Maximum use of resources present in domestic "used water"	Willy Verstraete, Pieter Van de Caveye, Vasileios Diamantis	309	1.7557	2009	Bioresource Technology 100(23), 5537-5545
5	Effectiveness of domestic wastewater treatment using microbial fuel cells at ambient and mesophilic temperatures	Youngho Ahn, Bruce E. Logan	303	2.3858	2010	Bioresource Technology, 101(2), 469-475
6	Long-term performance of liter-scale microbial fuel cells treating primary effluent installed in a municipal wastewater treatment facility	Fei Zhang, Zheng Ge, Julien Grimaud, Jim Hurst, Zhen He	240	4.3478	2013	Environmental Science Technology, 47(9), 4941–4948
7	Platforms for energy and nutrient recovery from domestic wastewater: A review	D.J. Batstone, T. Hülsen, C.M. Mehta, J. Keller	237	4.933	2015	Chemosphere, 140, 2-11
8	Experimental determination of energy content of unknown organics in municipal wastewater streams	Ioannis Shizas, David M. Bagley	224	1.0000	2004	Journal of Energy Engineering, 130(2)

Position	Documents	Authors	Total citations	Normal citations	Publication year	Journal
9	Energy capture from thermolytic solutions in microbial reverse-electrodialysis cells	Roland D. Cusick, Younggy Kim, Bruce E. Logan	212	2.3178	2012	Science, 335 (6075), 1474-1477
10	Autotrophic nitrogen removal from low strength waste water at low temperature	Tim L.G. Hendrick, Yang Wang, Christel Kampman, Grietje Zeeman, Hardy Temmink, Cees J.N. Buisman	189	2.0663	2012	Water Research, 46(7), 2187-2193

Table 6. List of top 10 documents in terms of normal citations.

Position	Documents	Authors	Total citations	Normal citations	Publication year	Journal
1	Characterization of a microalga Chlorella sp. well adapted to highly concentrated municipal wastewater for nutrient removal and biodiesel production	Yecong Li, Yi-Feng Chen, Paul Chen, Min Min, Wenguang Zhou, Blanca Martinez, Jun Zhub, Roger Ruan	540	6.0923	2011	Bioresource Technology 102(8), 5138-5144
2	Recent progress on biodiesel production from municipal sewage sludge	Xiaoyan Liu, Fenfen Zhu, Rongyan Zhang, Luyao Zhao, Juanjuan Qi	26	5.6522	2021	Renewable and Sustainable Energy Reviews, 135, 110260
3	One-year operation of 1000-L modularized microbial fuel cell for municipal wastewater treatment	Peng Liang, Rui Duan, Yong Jiang, Xiao Yuan, Zhang Yong Qiu, Xia Huang	160	5.4194	2018	Water Research, 141, 1-8
4	Platforms for energy and nutrient recovery from domestic wastewater: A review	D.J. Batstone, T. Hülsen, C.M. Mehta, J. Keller	237	4.933	2015	Chemosphere, 140, 2-11
5	Hydrochar derived from municipal sludge through hydrothermal processing: A critical review on its formation, characterization, and valorization	Huan Liu, Ibrahim Alper Basar, Ange Nzihou, Cigdem Eskicioglu	21	4.5652	2021	Water Research, 199, 117186
6	Long-term performance of liter-scale microbial fuel cells treating primary effluent installed in a municipal wastewater treatment facility	Fei Zhang, Zheng Ge, Julien Grimaud, Jim Hurst, Zhen He	240	4.3478	2013	Environmental Science Technology, 47(9), 4941–4948
7	Revealing the role of adsorption in ciprofloxacin and sulfadiazine elimination routes in microalgae	Peng Xie, Chuan Chen, Chaofan Zhang, Guanyong Su, Nanqi Rena, Shih-Hsin Ho	69	4.2427	2020	Water Research, 172, 115475

Position	Documents	Authors	Total citations	Normal citations	Publication year	Journal
8	COD capture: a feasible option towards energy self-sufficient domestic wastewater treatment	Junfeng Wan, Jun Gu, Qian Zhao, Yu Liu	154	3.8933	2016	Scientific Reports, 6, 25054
9	Towards a sustainable paradigm of waste-to-energy process: Enhanced anaerobic digestion of sludge with woody biochar	Yanwen Shen, Jessica L. Linville, Patricia Anne A., Ignacio-de Leon, Robin P. Schoene, Meltem Urgun-Demirtas	130	3.2865	2016	Journal of Cleaner Production, 135, 1054-1064
10	Municipal wastewater sludge as a sustainable bioresource in the United States	Timothy E. Seiple, André M. Coleman, Richard L. Skaggs	110	3.2673	2017	Journal of Environmental Management Volume, 197, 673-680

3.3 Stage 3. Evolution of the research field

Keywords of a document represent the core content of the considered article within the relevant domain of knowledge (Su and Lee, 2010), so their analysis is a way to recognize and indicate the essential area of the research field (Yang et al., 2022). In this case, the keywords analysis was developed with VOSviewer in terms of keywords co-occurrence and SCIMAT to analyze the evolution of themes using overlay graphs and strategic and thematic diagrams. The results are analyzed and discussed below.

3.3.1 Keywords co-occurrence

The co-occurrence of the authors' keywords, which take into account synonyms, various spellings, and plurals, full counting method, that means that each co-occurrence link has the same weight. Of the 798 keywords that resulted after merging different variants of a keyword, for instance, 52 met the threshold. For each of the 52 keywords, the total strength of the co-occurrence links with other keywords was calculated and a minimum of 5 occurrences were used for the keyword analysis, resulting in the 5 clusters in Figure 13 in the colors red (Cluster 1), green (Cluster 2), blue (Cluster 3), yellow (Cluster 4), and purple (Cluster 5). The most relevant characteristics in terms of influential keywords, theme assigned, and average normal citation of each cluster are summarized in Table 7. To do that, firstly, the terms with the highest average normal citation in each cluster were selected and the clusters were separated into the principal theme according to their keywords; then, an average number was obtained for the average publication year as well as for the average normalized citation of each cluster.

The analysis of the cluster characteristics shows that the most relevant are clusters 4 (yellow) and 2 (green), with 1.3160 and 1.0792 average normal citations values, 10 and 11 keywords assigned, average publication year of 2016, and wastewater plants as urban biorefinery and bioelectricity from wastewater treatment plants themes assigned, respectively. Both biorefinery and bioelectricity are emerging concepts that have been under research and development over the last decades, coinciding with increased restrictions in environmental regulations regarding the elimination of micropollutants, gas emissions and sludge management (Kamali et al., 2019). These terms continue to evolve to be applied for wastewater treatment plants as a solution to generate the highest benefit from these facilities and a step forward to pave the way for a bio-based CE and obtain as bio-based chemicals, biofuels, bioenergy, and food (Moreno-García et al., 2021). Thus, moving towards a CE, the WWTPs are now conceived as biorefineries because both wastewater and residues from them can be valorized for recovering nutrients, producing value added products, energy vectors, and biofuels (Boni et al., 2021). Although the size of the circles of keywords in cluster 4 is similar, it is slightly larger in the case of *microalga*, in reference to a bio-treatment that is particularly attractive because of their photosynthetic capabilities, converting solar energy into useful biomasses and incorporating nutrients such as nitrogen and phosphorus causing eutrophication (De la Noue and de Pauw, 1988). This technology, in the context of biorefinery, presents several challenges, as the growth of microalgae requires large amounts of carbon, nitrogen and phosphorus (Nitsos et al., 2020) having high harvesting, pre-treatment, and microalgal purification costs (Ghaffar et al., 2023).

However, the use of microalgae has showed significant advantages as it reduces emissions and leads to energy savings compared to conventional processes (Apandi et al., 2019). On the other hand, bioelectricity is a renewable and sustainable electricity produced from biomasses (Souza et al., 2022). The most recent promising technology for the recovery of energy from wastewater included *microbial fuel cells* (MFCs), a bioelectrochemical system (BES) that shows simultaneous power production while treating wastewater, and microbial electrolysis cells (MECs) that recover energy as biogas by treating wastewater (Subha et al., 2020); in fact, microbial fuel cell is one of the keywords with the higher co-occurrence value, according with its circle size in Figure 13. Although this technology is still in its infancy and must be tested on a industrial pilot scale and in natural wastewater (Shanthi Sravan et al., 2021), it promises in the future to cut the costs of wastewater management through the production of hydrogen and other fuels (Kadier et al., 2020). Finally, Cluster 5 (purple), identified with CE in wastewater treatment plants, shows the more recent average publication year and includes *energy recovery* as the keyword with a higher co-occurrence value, revealing that the municipal wastewater treatment plants play an important role due to the integration of energy production (De la Noue and de Pauw, 1988; Mo and Zhang, 2013).

Finally, the only waste from wastewater treatment which treatment has been boarded in depth has been sludge, which shows the largest volume amongst all the components removed during the process (Hanum et al., 2019a). Technologies based on sludge anaerobic digestion are considered to be economical and environmentally-friendly for treating municipal wastewater sludge (Muhammad Nasir et al., 2012), and they have been identified in different clusters with keywords as methane, gasification, and methane production, revealing the larger number of researchers related to this theme, as well as the consolidation of the technology, with more than 1,000 anaerobic digestion systems based on sewage sludge in operation or under construction throughout the world (Hanum et al., 2019b). Nevertheless, some studies have reported the poor efficiency and long processing times of anaerobic digestion (Harris et al., 2017; Raheem et al., 2018; Zhao et al., 2017). As alternatives, the application of thermochemical techniques to WWTP sludge has been studied in recent years (Samolada and Zabaniotou, 2014). Processes such as incineration, gasification or pyrolysis can improve the efficiency and profitability of sludge through the production of syngas, bio-oil or char and their various applications.

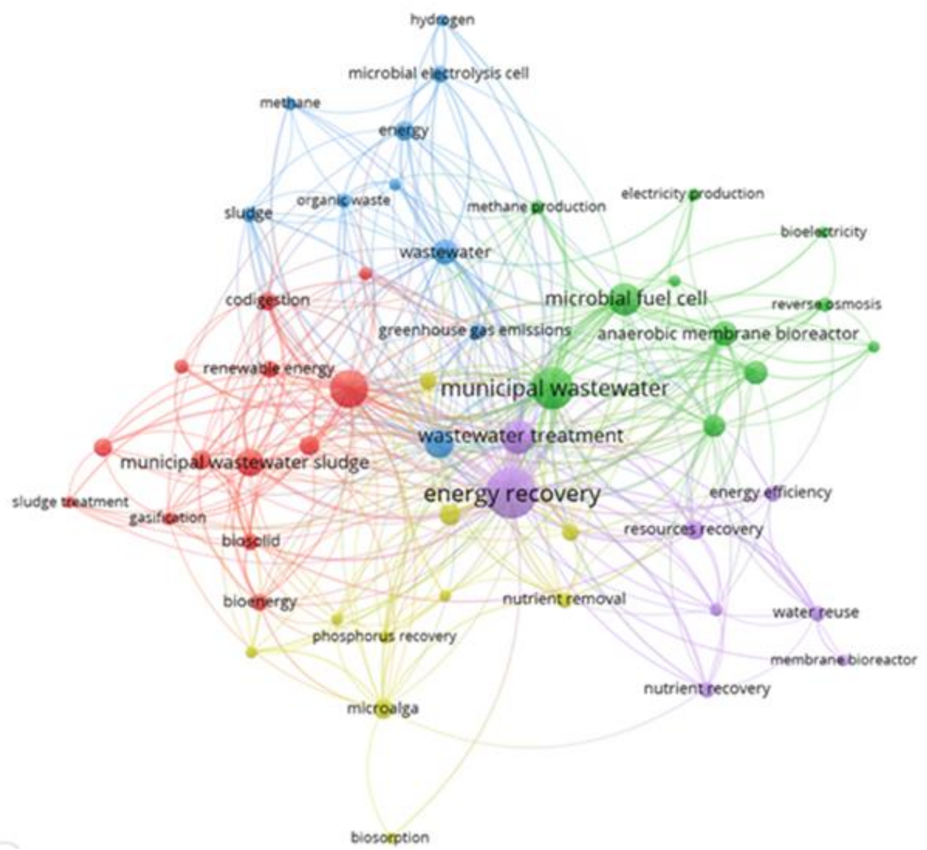


Figure 13. Network visualization of the authors' index keywords with a minimum of five occurrences in terms of average citations.

The absence of keywords that address the treatment of screening waste, from the point of view of obtaining energy, as a viable alternative to its current landfill disposal (L S Cadavid-Rodríguez and Horan, 2014; Clay et al., 1996; Wid and Horan, 2018) shows the low interest in this type of waste because they have been an unrepresented part of the waste generated compared to sludge Tsiakiri et al., 2021. In fact, only some works have dealt with the possible application of anaerobic digestion (Boni et al., 2021; L S Cadavid-Rodríguez and Horan, 2014; Mizanur Rahman et al., 2018;; Wid and Horan, 2018), although they have reported difficulties in the process because of the presence of plastics and textiles (Moreno-García et al., 2021) and a low methane production yield because it would require higher biomass concentrations (Boni et al., 2021). However, the higher fraction of sanitary textiles and other fractions, such as paper and cardboard, vegetables, and plastics, translate into a high lower calorific value and low values on the chlorine and mercury contents, so it could be considered suitable to produce solid fuel recovered for energy recovery in thermochemical processes (De la Torre-Bayo et al., 2022). At this point, as for sludge, the application of processes such as gasification or pyrolysis is considered a necessary field of study for screening waste.

Table 7. Description of keyword clusters in terms of number and influential keyword, theme assigned, and average normal citation of the cluster.

			Influential keywords				
Cluster number	Color	Number of keywords	Keyword	Average normal citations	Theme assigned	Average publication year	Average normal citations
1	Red	13	Bioenergy	1.3821	Application of anaerobic process	2016.4141	0.9355
2	Green	11	Electricity production	1.4127	Bioelectricity from wastewater treatment plants	2016.6438	1.0792
3	Blue	10	Energy	1.5025	Energy from wastewater treatment plants	2014.5849	0.9567
4	Yellow	10	Biorefinery	1.8614	Wastewater plants as urban biorefinery	2016.3955	1.3160
5	Purple	8	Nutrient recovery	1.2602	Circular economy in wastewater treatment plants	2017.0926	0.8317

3.3.2 Evolution of research themes field

The evolution of topics over time in the research field has been studied using overlay graphs, thematic evolution maps, and strategic diagrams. All of them were generated with SciMAT.

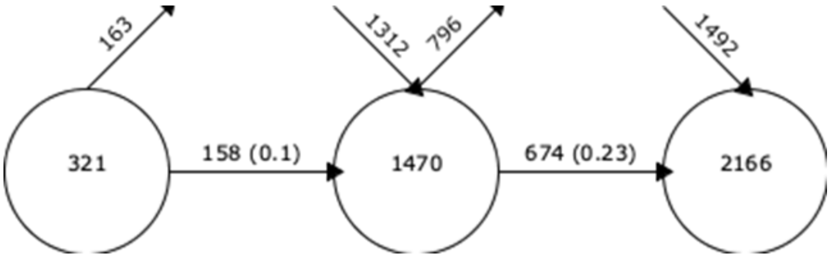


Figure 14. Overlay graph of the research field.

The overlay graph was used for a preliminary approximation to know the dynamic of changes of the themes across the horizon time (Cobo et al., 2011). As seen in Figure 12, the number of keywords of different periods, which is included in circles, increases from one period to the next one; in fact, the greater increase is observed from the initial to the active phase with relative growth (1st to 2nd subperiod) with an increase of 358% in comparison with the increment this and the active phase with high growth (2nd and 3rd subperiod) of 47.34%. In relation to the stability across the different periods, the analysis of the horizontal arrows, which represent the number of keywords shared by both subperiods, and the upper-incoming and upper-outcoming arrows of each one, which represent the number of new keywords in each period, shows the important renewal of terms related to the novelty of the field, much larger in the second period with 89.25% of new keywords; in fact, the Stability Index is lower (0.1) than in the case of the 3rd subperiod, which shows 68.8% of new keywords. Accordingly, with the yearly publication trend, and although the 3rd subperiod showed a very significant growth of the research field because of the possibility of applying WtE technologies according to policies to develop CE practices, in the 2nd subperiod, the development of technologies to produce energy from waste began to grow faster so the foundations for the technological development of solutions to obtain energy from waste were laid along it, coinciding with the approval of the Directive 2008/98/EC, which reinforced the waste hierarchy, boosting WtE technologies development.

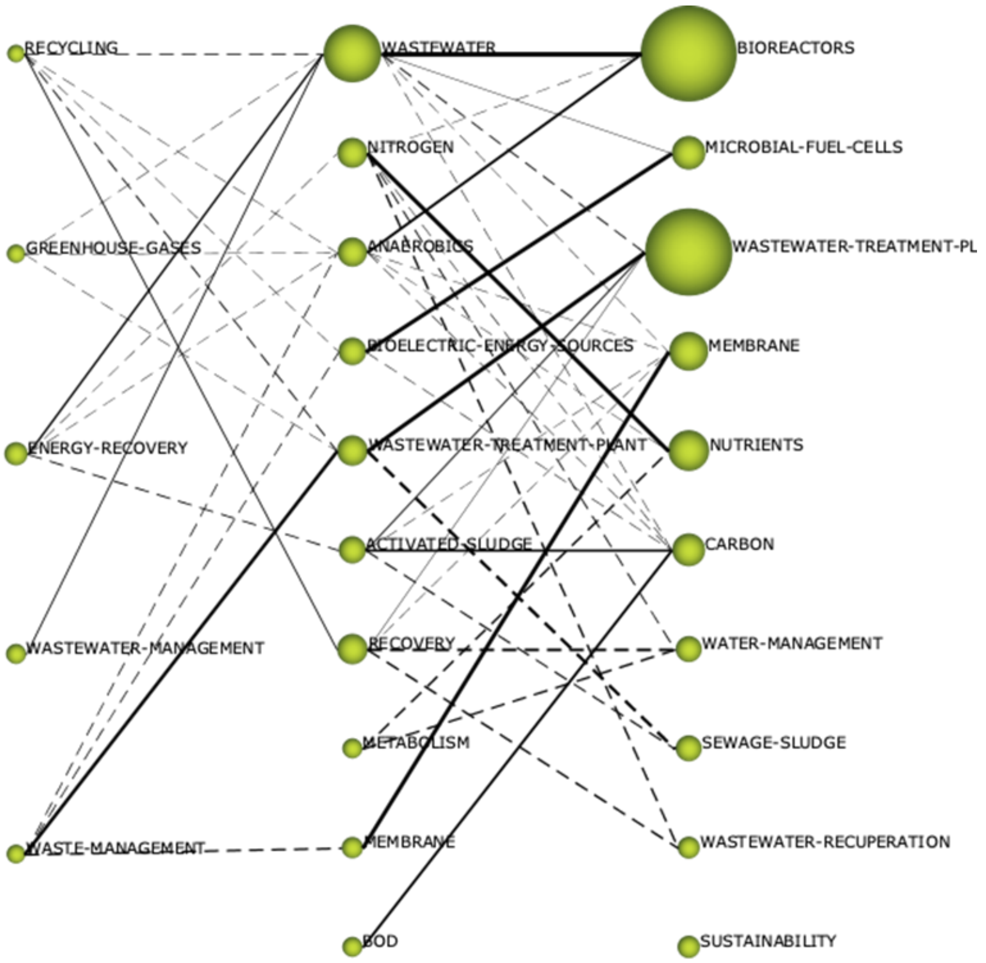


Figure 15. Thematic evolution map of the research field in terms of number of documents.

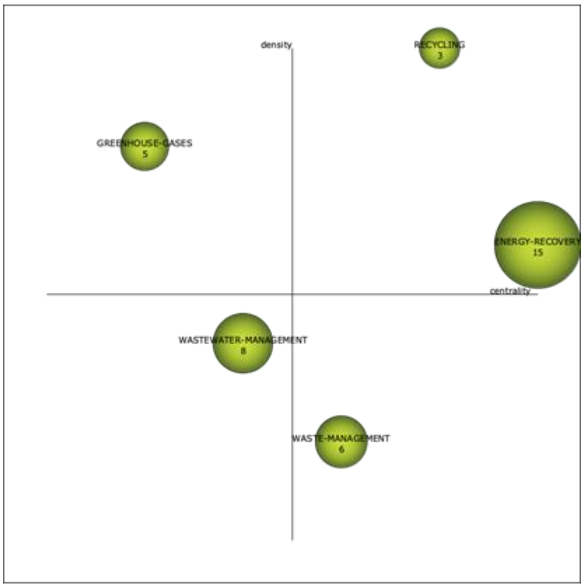
Figure 15 shows the thematic evolution maps of the research field in terms of the number of documents. Their analysis leads to the following:

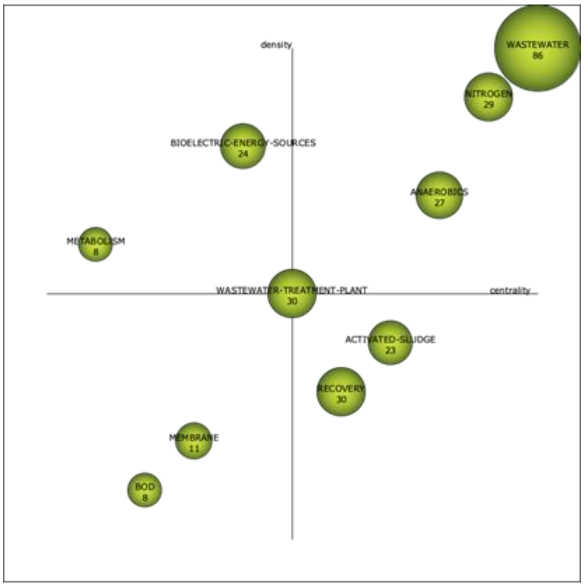
- None of the themes remained unchanged during all subperiods and only two have done during two of them; it is the case for WASTEWASTER TREATMENT PLANTS and MEMBRANE. Both of them were included in the 2nd and 3rd subperiods. These results are according with the great renovation in the themes of interest discussed in the overlay graph. In fact, it is clear that the evolution of WtE technologies in wastewater treatment plants, beginning with the anaerobic process (ANAEROBIOSIS), extensively applied throughout the world (Hanum et al., 2019b), and in the last years, the application of technologies to produce bioenergy, for example MICROBIAL FUEL CELLS.

- Some areas of the research field present great cohesion given that some of the identified themes are connected, with high thickness of the edges in many cases. For example, RECYCLING, in the 1st period, shows a large number of connections with themes in the 2nd periods, including WASTEWATER, ANAEROBICS, BIOELECTRIC ENERGY SOURCE, WASTEWATER TREATMENT PLANT, and RECOVERY. WASTEWATER, in the 2nd period, shows connections with the following themes in the 3rd period, BIOREACTOR, MICROBIAL FUEL CELLS, WASTEWATER TREATMENT PLANTS, MEMBRANE, and CARBON. These results reveal the potential of waste (De la Noue and de Pauw, 1988; Mo and Zhang, 2013).
- However, other areas of the research field present a lower cohesion without connections with other ones, for example, SUSTAINABILITY, or with a low number of them; this is the case for BOD (biological oxygen demand) and WASTEWASTER MANAGEMENT, which are only connected with CARBON and WASTEWATER, respectively. These results mean that they could be considered as the beginning of a new thematic area (Cobo et al., 2011), for example, as with SUSTAINABILITY, which appears in the last subperiod. The theme is now well described by keywords and it is not possible to detect its connections with others, for example, as with the theme BOD or CARBON, or the theme is connected with many thematic areas and it is difficult to categorize it (Cobo et al., 2011), for example, as with the themes WASTEWATER MANAGEMENT and WASTEWATER.
- The solid line reveals that the connected themes are labelled with the same keywords or the label of one them is part of the other one (Cobo et al., 2011),

i.e., WASTE MANAGEMENT and WASTEWATER TREATMENT PLANTS or RECYCLING and RECOVERY. All of them are related with the waste management concept and principles. A larger number of dotted lines is observed, meaning that the themes connected with them shared elements that are not the name of the themes, i.e., RECYCLING and ANAEROBICS, GREEN HOUSE GASES, and WASTEWATER TREATMENT PLANTS, or ENERGY RECOVERY and ANAEROBICS, sharing different themes related not only with the treatment of waste produced in wastewater treatment facilities but also with energy production or the positive effects in terms of greenhouse gas reduction.

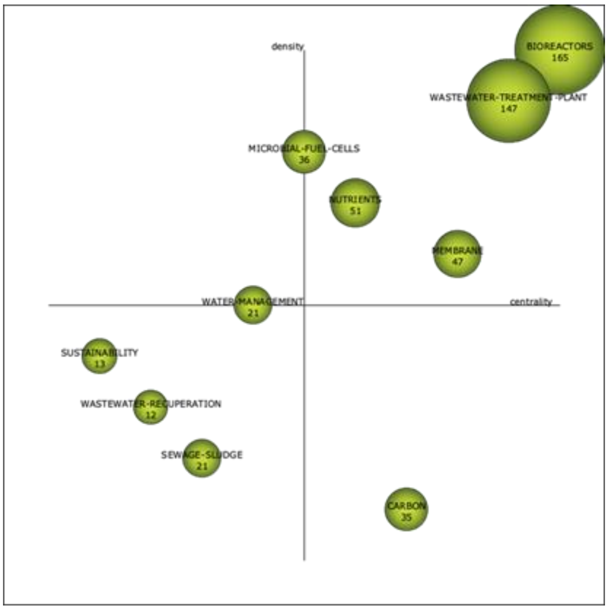
Finally, strategy diagrams for each subperiod were generated with SciMAT to show the performance of the evolution of research topics in terms of the number of documents published. These diagrams are included in Figure 16, which quantitative and impact measures to analyze each one is in Table 8. The number of transversal themes to the scientific field and high developed and isolated themes, located in the lower right and upper left quadrants, respectively, is very low; thus, in the case of transversal themes, only WASTE MANAGEMENT appears in the 1st subperiod, ACTIVATED SLUDGE and RECOVERY appear in the 2nd one, and CARBON appears in the last one; on the other hand, only four isolated themes were identified along the horizon time, specifically, GREEN HOUSE GASES in 1979 to 2008 and BIOELECTRIC ENERGY SOURCES and METABOLISM in 2009 to 2015. None of the themes are included in the last subperiod. Motors themes, in the upper right quadrant, shows well-developed and essential themes in the field and includes RECYCLING and ENERGY RECOVERY in the 1st subperiod; WASTEWATER, NITROGEN, and ANAEROBICS, in the 2nd subperiod, and finally BIOREACTORS, NITROGEN, MEMBRANE, and WASTEWATER TREATMENT PLANT in the 3rd subperiod. In the case of WASTEWATER TREATMENT PLANT, which is just in the border of the four quadrants in the 2nd period, it has evolved to the motor theme in the last one. The lower-left quadrant includes emerging or declining research topics, which lack development and relevance, although they may evolve and be relevant or disappear; in this sense, the theme WASTEWATER MANAGEMENT is the only one that appears in the 1st subperiod; MEMBRANE and BOD are in the 2nd one, and SUSTAINABILITY, WASTEWATER RECUPERATION, and SEWAGE SLUDGE are in the 3rd subperiod. In this case, the theme MEMBRANE has evolved from the emerging theme to motors from the 2nd to the 3rd subperiod. Finally, in the last subperiod, both MICROBIAL FUEL CELLS and WATER MANAGEMENT are in the border of the two quadrants, specifically, between isolated and motors one and emerging to isolated, respectively.





1979 - 2008

2009 - 2015



2016 - 2021

Figure 16. Strategy diagrams in terms of the number of documents published.

Finally, Table 8 summarizes the performance measures of the themes by subperiod, and it complements that displayed in Figure 16. A higher centrality value shows a higher importance of a theme in the development of the entire research field analyzed (Callon et al., 1991). WASTEWATER, in the 2nd subperiod, shows the higher centrality value (949.03), followed by BIOREACTORS and WASTEWATER TREATMENT, both in the 3rd subperiod, with centrality values of 708.8 and 457.24, respectively. High centrality values are related to the importance of these themes with the global development of the analyzed scientific field, as well as the higher degree of their external cohesion. On the other hand, and according to the concept of density, RECYCLING is the theme with the stronger strength of internal ties among all keywords describing the research theme, understanding that they have the higher development (Callon et al., 1991). These results are related to the fact that anaerobic digestion of municipal wastewater sludge, which results in the production of a biogas and of a digestate and is regarded by Directive 98/2008 for waste as a recycling operation, is a widely applied technology to produce energy from waste produced in wastewater plants.

Table 8. Performance measures for the themes for subperiods of the time horizon.

Theme name	Number of documents	h-index	Average citations	Number of citations	Centrality	Density
1979-2008						
RECYCLING	3	3	71	213	192.43	169.31
GREENHOUSE-GASES	5	5	81.4	407	98.53	84.67
ENERGY-RECOVERY	15	14	97.13	1457	296.44	79.32
WASTEWATER-MANAGEMENT	8	8	88	704	103.73	73.53
WASTE-MANAGEMENT	6	5	62.17	373	117.7	35.37
2009-2015						
WASTEWATER	86	43	80.5	6923	949.03	124.56
NITROGEN	29	26	105.38	3056	309.45	68.04
ANAEROBICS	27	21	84.93	2293	276.42	51.33
BIOELECTRIC-ENERGY-SOURCES	24	21	107.08	2570	151.5	54.11
WASTEWATER-TREATMENT-PLANT	30	21	60.77	1823	223.87	40.49
ACTIVATED-SLUDGE	23	20	123.22	2834	252.22	20.92
RECOVERY	30	23	88.07	2642	230.09	19.89
METABOLISM	8	8	58.38	467	39.59	42.44
MEMBRANE	11	9	83.45	918	56.25	16.96
BOD	8	8	62.5	500	47.18	16.53
2016-2021						
BIOREACTORS	165	39	25.61	4226	708.8	134.33
MICROBIAL-FUEL-CELLS	36	20	31.5	134	86.24	41.01
WASTEWATER-TREATMENT-PLANT	147	36	25.24	3711	457.24	60.28
MEMBRANE	47	20	24.85	1168	188.78	29.95
NUTRIENTS	51	22	26.82	1368	128.45	35.54
CARBON	35	17	24.89	871	150.82	9.63
WATER-MANAGEMENT	21	12	26.95	566	56.93	25.4
SEWAGE-SLUDGE	21	13	27.43	576	53.02	12.4
WASTEWATER-RECUPERATION	12	9	32.42	389	40.12	15.48
SUSTAINABILITY	13	10	22.31	290	10.76	23.11

4 CONCLUSIONS

Research trends in WtE from municipal wastewater treatment plants were analyzed from the first paper in the research field, published in 1979, until 2021, obtaining the following conclusions:

- The results show an exponential increase in the number of papers that, although has been aborded from years, it has not yet reached a stage of maturity because of the important roles that the CE paradigm and the energy sustainability are playing a very nowadays in the transformation of wastewater treatment plants toward biorefineries concept.
- A low concentration of documents per source, country, and author was observed, which indicates the great interest in the field. About categories of leading journals are included in environmental science and environmental engineering categories, meaning that the analysis of the research field both from a scientific and technological perspective is necessary to address challenges in sustainability and CE. In fact, the research field presents a significant cohesion in topics related to the technologies to produce energy from the waste of wastewater treatment plants. Finally, although China leads the rankings in the research field, co-authorship analysis showed important collaborations between countries, as well as small collaboration groups of authors.
- The analysis of the evolution of themes shows the novelty of the topics with an important incoming and outcoming of keywords since 1979, which is also reflected in the different topics of the thematic evolution maps. In this sense, studies on energy from wastewater treatment have focused on energy from sludge, mainly in anaerobic digestion processes. However, no significant presence of thermochemical processes applicable to sludge, such as combustion, gasification or pyrolysis, has been detected. Emerging technologies such as microalgae, microbial fuel cells, or the use of membrane technologies, all of them directed to the consolidation of the concept of biorefinery according to CE principles, are at an early stage. So far, they can only be considered as potential alternatives, as their implementation at the industrial level has yet to be studied. Research should

focus on the cost reduction of these processes, along with improving efficiency in producing biofuel and bioelectricity.

- Scientific mapping shows how other waste fractions, such as primary screening waste, have yet to be considered in the literature. In the detailed analysis of the publications on this waste, only anaerobic digestion processes were found, which, unlike their sludge application, could not be successfully developed due to the exact composition of the screening. To achieve zero waste in municipal wastewater facilities, it is considered necessary to open new lines of research to valorize the screening waste.

CHAPTER 2

CHARACTERIZATION OF SCREENING WASTE FROM BIOFACTORÍA SUR[2]

1 INTRODUCTION

Waste fractions in WWTPs include screening waste generated in the first treatment stage of the facility. This waste has been an unrepresented part of the waste generated in WWTPs. Scientific mapping developed in previous Chapter shows how the screening waste, have yet to be considered strongly in the literature, especially compared to other wastes, such as sludge from WWTP or MSW (Boni et al., 2021; Cadavid-Rodriguez and Horan, 2012; Tsiakiri et al., 2021).

In water treatment, CE is focused on water recycling and sludge treatment, which is generated in vast quantities at WWTPs. Many studies have shown the effectiveness of obtaining energy from sludge, assessing the environmental impact of fuel production (Xu et al., 2014), including evaluating its use in clinker production (Donatello and Cheeseman, 2013; Lin and Ma, 2012; Mills et al., 2014; Valderrama et al., 2013; Xu et al., 2014), as well as its possible use in agriculture (Cieślik et al., 2015; Houillon and Jolliet, 2005; Roig et al., 2012; Suthar, 2010). The wastewater treatment

[2] The results shown in this Chapter were presented in: De la Torre-Bayo, J. J.; Martín-Pascual, J.; Torres-Rojo, J. C.; Zamorano, M. Characterization of Screenings from Urban Wastewater Treatment Plants: Alternative Approaches to Landfill Disposal. J. Clean. Prod. 2022, 380, 134884. https://doi.org/10.1016/j.jclepro.2022.134884

plant market is expected to grow by 6.12% through 2027, owing to industry expansion, population growth, and rapid urbanization. This growth will increase the amount of screening waste produced (Paulsrud et al., 2013). In Spain, the destiny for this waste is landfilling. However, this disposal is bound to disappear in the coming years (Kehrein et al., 2020; Razafimanantsoa et al., 2014), so it is considered necessary to seek alternatives to the current landfill disposal of screening waste from WWTP.

Landfilling generates significant environmental problems due to the screening waste high organic matter and moisture content (Cadavid-Rodriguez and Horan, 2012) and substantial transportation costs (Le Hyaric et al., 2010). It is the most common destination for screening disposal (Tsiakiri et al., 2021; Wid and Horan, 2018). Nevertheless, depending on the location, screening waste is incinerated in municipal facilities (Todt and Jenssen, 2015; Tsiakiri et al., 2021), which is considered a good alternative. The high water content is also an unfavourable characteristic that may jeopardize the operating conditions of the incineration plant in terms of combustion temperature and gaseous emissions (Le Hyaric et al., 2009). Both landfill and incineration are expensive options and carry a large carbon footprint (Cadavid-Rodriguez and Horan, 2012).

Regarding alternatives, in the detailed analysis of the publications on this waste through bibliometric analysis exposed in Chapter 1, only anaerobic digestion processes were found. Considering the high organic matter content, mainly suggests the possibility of including screening waste in anaerobic digestion as a viable method of energy recovery (Boni et al., 2021; Cadavid-Rodriguez and Horan, 2012; Mizanur Rahman et al., 2018; Tsiakiri et al., 2021; Wid and Horan, 2018). Methane production was studied based on conditions such as reactor, presence of solids, or retention time. The results obtained were 0.19 to 0.27 L CH_4/g VS (Le Hyaric et al., 2010), 0.42 L CH_4/g VS (Cadavid-Rodriguez and Horan, 2012), 0.36 L CH_4/g VS (Wid and Horan, 2018), 0.20 to 0.74 L CH_4/g VS (Tsiakiri et al., 2021), 0.58 to 1.04 L

CH_4/g VS (Boni et al., 2021). These methane values suggest a potential energy source in the screening waste (Wid and Horan, 2018). Nevertheless, difficulties with the process due to the screening waste composition (Cadavid-Rodriguez and Horan, 2012) and a low methane production yield have been reported (Boni et al., 2021). The literature also includes a study proposing the utilization of cellulosic rejections accumulated in WWTPs to produce free sugars that could be transformed into different products, including bioethanol (Ballesteros et al., 2022).

This Chapter presents a complete characterization of the screening waste, analyzing its properties and their daily, weekly and seasonal variability. According to EC, alternatives to the current elimination are sought, proposing different possibilities to those found in the literature. Characteristics leading to evaluating the screening as a possible source for producing SRF and its subsequent energy recovery in thermochemical processes have been studied.

2 MATERIALS AND METHODS

2.1 CHARACTERISTICS OF THE WASTEWATER TREATMENT PLANT

This research was conducted in Biofactoría Sur of Granada. This facility is managed by EMASAGRA, the municipal water and wastewater company of Granada, and treats more than 18 M m^3 per year of urban wastewater from the approximate equivalent of 425000 inhabitants. The process includes pre-treatment, primary or decanting, biological, secondary, and disposal. This facility is a European benchmark for CE in the sector, seeking energy self-sufficiency, zero waste and 100% reuse of treated water. Different measures have been implemented, such as reusing 100% of treated water and its use for agricultural irrigation, transforming 100% of sludge into compost and recovering 100% of sand and grease in agriculture. 442 t/y of screening waste produced from the screen and with a 3 mm pitch is deposited in the landfill; in consequence, to fulfil the zero-waste objective, it is necessary to seek a way to recycle this waste fraction. Sections 2.2, 2.3 and 2.4 summarize a description of the sample collection and classification process and the techniques applied to their characterization.

2.2 Sample collection

According to the UNE-EN 14899:2007 (Ambiente, 2007), the samples collected had an approximate as-received mass between 6 and 14 kg and an average of 8.70 kg. Collection and sampling of screening waste was conducted collaborating with the technical staff of the plant and taking into account the effect of seasonal variations in the amount of residue produced (Canler and Perret, 2004), as well as on their composition (Cadavid-Rodriguez and Horan, 2012). To this end, sampling included two different weeks of the year coinciding with the winter and summer seasons. The winter sampling campaign was conducted between 3 and 9 March 2021, and the summer one between 14 and 20 July 2020. To detect possible effects of temporal variation in human habits (Kaless et al., 2016), two daily samples were taken. The nighttime sample was collected at 6 am, while the daytime sample was taken at 6 pm. This scheme ran for all seven days of the week, from Monday to Sunday. Each sample was collected from the screening container right at the exit of the compactor. Ultimately, 28 raw screening samples were collected in two campaigns, 14 for each season. To characterize the five main fractions, present in the screening it was necessary to collect additional random samples, both in summer and winter.

2.3 Classification of fractions

To determine the component fractions of screening waste, we initially divided them into ten fractions according to the classification developed by Le Hyaric et al. (2009). However, due to the absence of components such as glass, metals or wood and the difficulty in separating tiny fractions, the following five fractions were finally considered:

- Sanitary textiles: tampons, sanitary towels, wipes etc.
- Paper and cardboard: newspapers, brown corrugated cardboard, package paper rolls, office paper.
- Vegetal: leaves, flowers, plant parts, food scraps etc.
- Plastics: plastic film, bottles, rigid plastic, packaging, condoms, wrapping and bags.
- Other: fractions that are very costly to separate, including inert debris, hair, organic matter and fine particulates (<20 mm).

Fractionation was performed on a triage table, manual separating the fractions in different containers. Each fraction was weighed separately and compared to the total weight of the original screening sample, obtaining its percentage.

2.4 CHARACTERIZATION TESTING METHODS

Following fraction classification, a series of representative samples were reserved for characterizing screening waste from wastewater treatment plants. The physical and chemical parameters included in this study were selected from a literature review developed to treat this type of waste, including anaerobic digestion and energy valorization alternatives; chemical parameters used to characterize potential SRF were included. Table 9 summarizes the parameters applied in the characterization, and the testing methods are in section 2.4.1. and 2.4.2.

Table 9. Standards applied to parameter assessment.

Parameters		Standard
Physical	Moisture	UNE-EN 15414-3:2011 Solid recovered fuels. Determination of moisture content by the oven drying method. Part 3: Moisture content of the sample for general analysis (AENOR, 2011a)

Parameters			Standard
	Ash		UNE-EN 15403:2011 Solid recovered fuels. Determination of ash content (AENOR, 2011b)
	Volatile solids		UNE-EN 15402:2011 Solid recovered fuels. Determination of volatile matter content (AENOR, 2011c)
Chemical	Organic matter		Tyurin method
	C H N		UNE-EN 15407:2011 Solid recovered fuels - Methods for the determination of carbon (C), hydrogen (H) and nitrogen (N) content (Obse, 2011)
	Standardized parameters for SRF classification	Lower Heating Value (LHV)$^{ar,\ d}$	UNE-EN 15400:2011 Solid recovered fuels. Determination of the calorific value (AENOR, 2011d)
		Cl^{d}	UNE-EN ISO 10304-1:2009 Water quality: Determination of dissolved anions by liquid phase ion chromatography. Part 1: Determination of bromide, chloride, fluoride, nitrate, nitrite, phosphate and sulphate (AENOR, 2009)
		Hg^{ar}	UNE-EN 15411:2012 Solid recovered fuels. Method for determining trace element content (As, Ba, Be, Cd, Co, Cr, Cu, Hg, Mo, Mn, Ni, Pb, Sb, Se, Tl, V and Zn) (Be, 2012)

ar As received. d Dry

2.4.1 Physical analysis

The physical parameters as moisture, ash and volatile solid content were determined under the provisions set out in Table 9. Determinations were conducted in triplicate for the 14 raw waste samples and the 5 corresponding to the fractions, for a total of 19 for each sampling season. All these parameters are crucial to the action of microorganisms in anaerobic digestion processes or energy valorization. Volatile solids and ash in solid waste correspond to combustible and non-combustible materials, respectively; their determination is a reasonable way to approximate the degree of combustibility of waste.

2.4.2 Chemical analysis

- Elemental analysis and organic matter

In the anaerobic digestion process, organic matter is degraded by the action of microorganisms in the absence of oxygen. To the physical requirements described bacteria inside the digester must have adequate organic matter, carbon (C) and nitrogen (N) for proper growth. Based on this framework, some chemical parameters have been calculated.

The amount of organic matter present in the residue was calculated using the Tyurin method. It consists of reverse redox volumetry; the organic matter is oxidized by adding an excess of oxidant under acidic conditions and applying heat and a catalyst; after oxidation, the excess dichromate is titrated using Mohr's salt ($Fe(NH_4)_2(SO_4)_2$) as a reducing agent. Elemental analysis was conducted at the University's Scientific Instrumentation Center (CIC) according to UNE-EN 15407:2011 (Table 9). The method was performed with the THERMO SCIENTIFIC Flash 2000 Model elemental analyzer to determine the carbon, nitrogen, and hydrogen contents. Elemental analysis, determined by unit analysis for each sample, and organic matter analysis were conducted in triplicate for each of the 19 samples of each phase.

- Parameters for quality standards for SRF production

Among the materials accepted as a source for the production of SRF, the UNE-EN ISO 21640 standard specifies 'other solid waste from urban wastewater treatment' as a source (AENOR 2021). To consider the possibility of energy valorization, the following parameters described for SRF, which are included in the standard UNE-EN 15359 (AENOR 2012), have been determined: LHV (as received and dry), chlorine (Cl) content (dry) and mercury (Hg) content (as received). The calorific value was determined in triplicate for each of the 19 samples from each phase; Cl content and Hg content were determined by unit analysis for each sample. Table 10 summarizes the quality standards for SRF according to the values of the classification parameters.

Table 10. Quality standards for SRF production

Classification parameters	Statistical measure	Unit	Classes				
			1	2	3	4	5
Lower Heating Value (LHV)	Mean	MJ/kg (ar)	≥25	≥20	≥15	≥10	≥3
Chlorine (Cl)	Mean	% (d)	≤0.2	≤0.6	≤1	≤1.5	≤3
Mercury (Hg)	Median	mg/MJ (ar)	≤0.02	≤0.03	≤0.08	≤0.15	≤0.50
	80^{th} percentile	mg/MJ (ar)	≤0.04	≤0.06	≤0.16	≤0.30	≤1.00

ar As received. d Dry

LHV is included for its economic effect, as it describes the energy generated in its combustion. It has been determined based on the UNE-EN 15400:2011 (Table 9). The 6100 Parr oxygen combustion calorimeter has been used. A weighed portion of the test sample is burned at high oxygen pressure under specified conditions. The results provided by the calorimeter are those determined for the higher heating value (HHV) on a dry basis. Using Eq. 1 and Eq. 2, the lower calorific value (LHV) on a dry basis and the HHV and LHV on a wet basis, were obtained. The equations consider the heat of condensation of water (597), the hydrogen content (H) and the moisture content of the residue (w), these last two parameters in percent out of one.

$$LHV_d\left(\frac{cal}{g}\right) = HHV_d\left(\frac{cal}{g}\right) - 597 \times (9 \times H_d) \qquad \text{Eq. 1}$$

$$LHV_w\left(\frac{cal}{g}\right) = HHV_d\left(\frac{cal}{g}\right) \times (1 - w) - 597 \times (9 \times H_w + w) \qquad \text{Eq. 2}$$

Chlorine content is a limiting factor regarding screening waste as fuel as it measures the potential effects of corrosion, slagging and fouling in boilers (Rotter et al., 2011). A sample of ash derived from the procedure of calorific value determination was diluted in distilled water, and the Cl content was determined from there. For this purpose, ion-exchange chromatography was applied at the CIC of the University of Granada, based on the UNE-EN ISO 10304-1:2009 (Table 9).

Hg content is a measure of the toxicity released into the environment as a consequence of the combustion of the material (Iacovidou et al., 2018). It was determined on a sample ground to a particle size similar to that of dust UNE-EN 15411:2012 (Table 9). This analytical procedure was performed using a mass spectrometer with a plasma torch ionization source and a NexION 2000 B quadrupole ion filter with an NWR 213 Laser Ablation system.

3 RESULTS AND DISCUSSION

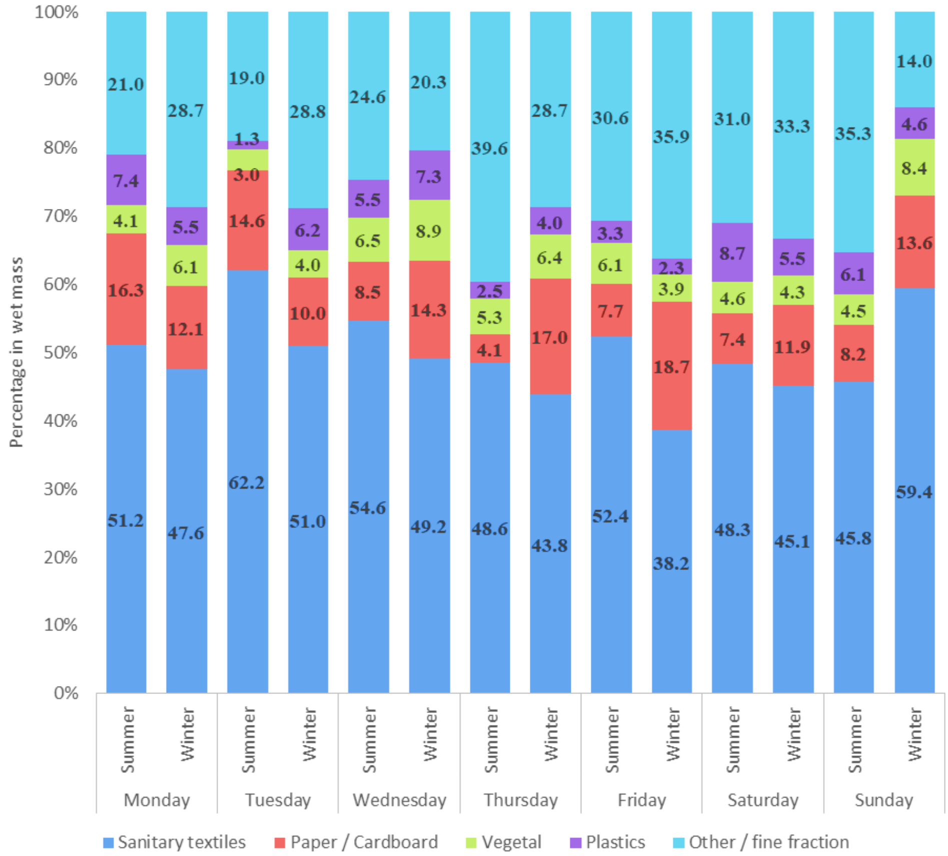
Collected samples were analyzed according to the laboratory standards and parameters described. The results of fraction classification are summarized in Table 11 and Figure 17. The mean values of screening waste characterization are shown in Table 12 for the seasonal and hourly variables. The daily variability of some of the parameters analyzed is shown in Figure 18 and Figure 19. Some of the results obtained in previous tests are contextualized by comparison with other types of waste, such as sludge and MSW, as well as with SRF.

3.1 CLASSIFICATION OF FRACTIONS

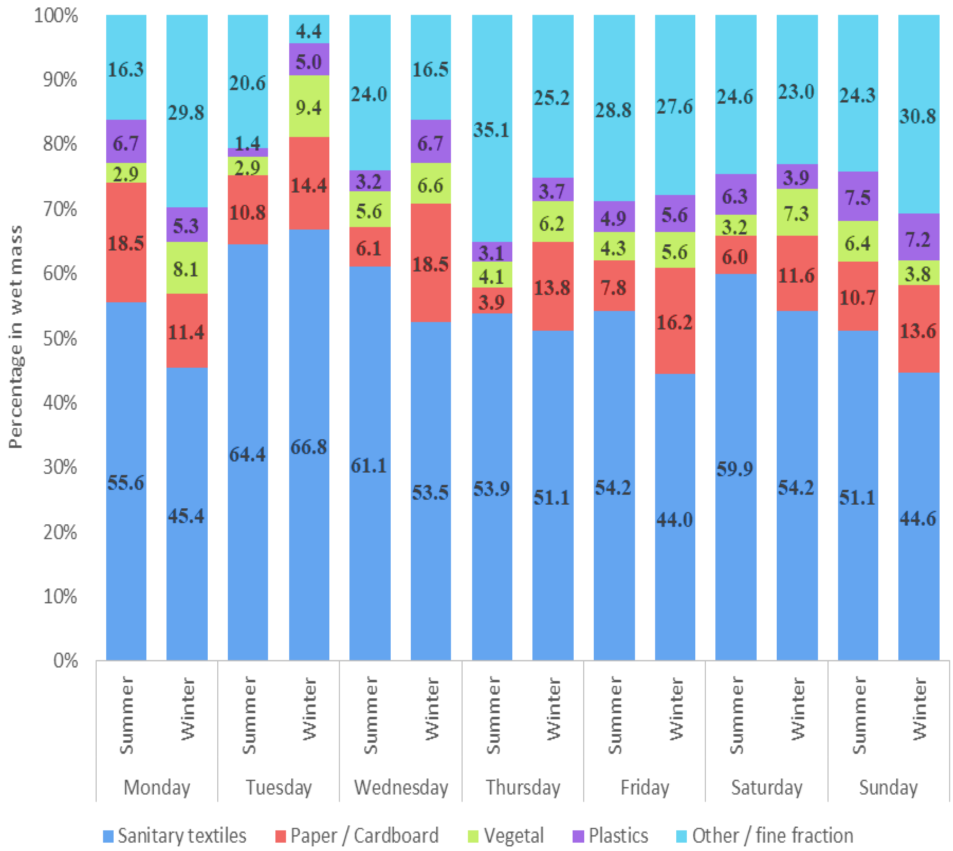
Table 11 summarizes the mean, standard deviation, and coefficient of variation values of the contents of different fractions. Taking into account the annual period, season (winter and summer) and time of day (diurnal and night) for the as-received samples; the annual average value for the classification of screening waste for dry samples was determined from the results of the as-received samples and the moisture content of each fraction.

Table 11. Average values for the classification of screening waste.

Sampling			Statistical parameters	Sanitary textiles	Paper/cardboard	Vegetal	Plastics	Other/fine fraction
				Screening waste fractions				
As received	Year season	Summer	Average value (%)	54.50	9.30	4.50	4.80	26.80
			Deviation	5.60	4.42	1.29	2.40	6.84
			Coefficient of Variation	0.10	0.47	0.29	0.50	0.25
		Winter	Average value (%)	49.60	14.30	6.50	5.10	24.90
			Deviation	7.31	2.68	1.90	1.41	8.48
			Coefficient of Variation	0.15	0.19	0.29	0.27	0.34
		Winter difference over summer (%)		8.99	-53.76	-44.44	-6.25	7.08
	Day time	Diurnal	Average value (%)	49.80	11.70	5.40	5.00	27.90
			Deviation	6.18	4.24	1.72	2.14	7.30
			Coefficient of Variation	0.12	0.36	0.32	0.42	0.26
		Night	Average value (%)	54.30	11.70	5.50	5.00	23.60
			Deviation	7.01	4.56	2.02	1.80	7.60
			Coefficient of Variation	0.13	0.39	0.37	0.36	0.32
		Night difference over diurnal (%)		-9.04	0.00	-1.85	0.00	15.41
	Annual		Average value (%)	52.10	11.80	5.50	5.00	25.90
			Deviation	6.87	4.39	1.84	1.94	7.63
			Coefficient of Variation	0.13	0.37	0.33	0.38	0.29
Dry	Annual		Average value (%)	48.28	10.52	7.57	9.10	24.83
			Deviation	6.73	3.92	2.50	3.42	7.41
			Coefficient of Variation	0.14	0.37	0.33	0.37	0.29
Dry samples differences over as received simples (%)				7.33	10.87	-37.72	-82.14	4.11



a)



b)

Figure 17. Classification of screening waste fractions. Daily variation: a) Daytime sample. b) Nocturnal sample.

A comparison of annual samples showed more significant differences between as received and dry samples in the case of plastic. These results are related to the lower moisture content observed in the plastic fraction due to low-density polyethylene, with a low field capacity (Qi et al., 2020).

Content of the various as-received and dry fractions, the highest value corresponded to sanitary textiles, with an annual average of 52.10% and 48.28%. This fraction showed the lowest coefficients of variation, 13.18% and 13.95%, for as-received and dry samples. Comparing with others screening waste fractions, the rest of values were higher or very close to 30%. These results showed a homogeneous data set for sanitary textiles but not for the rest of the screening fractions. Analysis of the coefficients of variation of the rest of the samples showed values higher than 30% in two or three samplings for all screening fractions except sanitary textiles. These results outcome from the heterogeneity of this type of waste, a characteristic of other types, such as urban waste (Colomer, F.J., Gallardo, 2007). These values show that the sanitary textile fraction is not only the predominant but also the one with the lowest variation, so it is considered the mainstay of screening waste. Sanitary textiles are composed of wipes and other hygiene products, including cellulose and several synthetic fibres such as high-density polyethylene, polyethylene-vinyl acetate, polypropylene and polystyrene (Marques et al., 2020). Its disposal in a landfill causes leachate emissions, causing water and air pollution (Zhang et al., 2021). The values obtained for this fraction were lower than those determined by Le Hyaric et al., (2009) who reported a range from 54.7% to 72.9% on a dry basis, but higher than those of Wid and Horan, (2018) who obtained results of 27.2% for the wet mass of sanitary textiles. The differences detected show the heterogeneity of the composition of this waste, which will depend on the citizen's habits, the meteorology of the area of influence and the characteristics of the pre-treatment, more specifically, the gap size of the screens (Cadavid-Rodriguez and Horan, 2012).

The following most dominant fraction was the so-called other fraction, composed of subfractions that are very costly to separate, including inert debris, hair, organic matter and fine particulates (<20 mm), comprising 25.9% and 24.83% of as-received and dry samples. The coefficients of variation were around 0.30, indicating a high degree of heterogeneity in the data. Winter and night samples, in particular, with values of 0.34 and 0.32, corresponded to heterogeneous data sets.

The fraction, including paper and cardboard, showed annual averages of 11.80% and 10.52% in as-received and dry samples, corresponding to heterogeneous data sets. Only in the case of winter samples could the data set be considered homogeneous, with a coefficient of variation of 0.19 for this period. Both remaining fractions, including vegetables and plastics, showed similar average values in the as-received annual samples, reaching 5.50% and 5.00%. These fractions also presented high heterogeneity, reaching coefficients of variation of 0.37 for night samples of vegetables and 0.50 for summer samples of plastics.

To analyze the effect of the season, we summarized the average values for each fraction of summer and winter sampling, as well as the percentage variation of the winter sampling compared to that of the summer one (Table 11). Paper and cardboard, followed by vegetal, were the fractions that showed the greatest differences, with an increase in winter of 53.76% and 44.44% concerning the summer percentage. The remaining fractions presented a variation below 10%, showing a higher percentage of plastic in winter than in summer but a reduction in the case of sanitary textiles and other fractions. Lower percentages of paper/cardboard in summer sampling could be related to higher water consumption this season due to higher temperatures, which would dissolve the paper fraction (Hussien et al., 2017). It could indirectly impact the percentage represented by the sanitary textile fraction that was reported to be lower in winter. The increase in vegetables in the case of winter sampling (4.44% higher in winter than in summer) may be explained by vegetables carryover from rainy days and the fact that the waste source is a combined sewage network (Michielssen et al., 2016). Rains increase dissolution, explaining the decrease in the so-called other fraction from 26.80% in summer sampling to 24.90% in the case of winter sampling.

Regarding the time of day, differences between diurnal and night samples were lower. Only the so-called other and sanitary fractions displayed noteworthy changes; specifically, night samples showed a 15.41% reduction over diurnal ones in the case of the other fraction and a 9.04% increase in the case of sanitary textiles. These differences could be explained by the higher cleanliness of the night-time sample due to lower activity during the night, as waste disposal into the sewage system is related to personal hygiene activities (Bernal et al., 2020).

Figure 17 summarizes the percentage of screening waste fractions sampled on each day of the week. The daily variation shows the heterogeneity of the waste across days of the week as well as the effect of rainy days; in fact, there were coincidences in the slight increase in the amount of textiles on both Tuesdays (summer and winter sampling) and increased rainfall, corresponding to days where registered 7.2 and 9 mm/d of rain. The influence of heavy rain was evident in the winter Tuesday night period, resulting the highest percentage of sanitary textiles recorded in the sampling (66.80%) and the lowest for the other fraction (4.40%) dissolved by the rain. Besides, for the same sample, the high value for vegetables (9.80%) was also notable, a result favoured by the dragging capacity of rainfall.

The comparison of screening waste fractions discussed revealed similar compositional characteristics to those of MBT reject waste. It is not only due to of its heterogeneity (Edo-Alcon et al., 2016) but also because of the difficulty separating, recycling or valorising it from a technical, economic and environmental perspective (Edo-Alcón et al., 2016). Regarding the fractions present, MBT rejects are mainly composed of paper and cardboard, plastics and organic matter, together accounted for 71.3% of the total(Ramos Casado et al., 2016) and 76.1% reported by Montejo et al., (2011). As Dong et al. (2010) pointed out, the fractions found and the composition of the scrap waste are similar to those present in MSW rejects, except for the high percentage of sanitary textiles. The composition of sanitary textiles includes cellulose and synthetic fibres. It could be compared to the paper and plastic fractions mostly present in MBT rejects. So, the composition of screening waste could be similar to that of the rejects, and matching options for converting waste-to-energy could be studied.

3.2 WASTEWATER TREATMENT PLANT SCREENING WASTE CHARACTERIZATION

Following classification of the fractions, their physical and chemical parameters were analyzed according to established methodologies. The results were analyzed in terms of physical and chemical parameters, and the chemical parameters that characterized SRF. Table 12 summarizes the results, which are discussed in section 3.2.1.

3.2.1 Physical parameters

Moisture, ash and volatile solid contents by season and time of day, as well as those in the annual sample, are summarized in Table 12.

The average annual water content of the residue was 77.30%, with a statistically significant difference between a higher value in winter sampling (78.80%) and a lower one in the summer season (75.80%). The T-value about seasonal variability is 0.001, showing a minimal possibility that both data sets underlay each other. In the case of daytime sampling, a 1% difference was observed between the daytime and night-time samples, obtaining a higher T-value (0.225). It is essential to highlight that the samples were homogeneous regarding moisture content, as indicated by coefficients of variation lower than 0.06 in all cases. The daily evolution is shown in Figure 18. No trend was observed throughout the week. In samples from the summer period, the average moisture content was one percentage point lower

Table 12. Average values for screenings parameters analyzed.

Sampling		Statistical parameter	Physical parameters			Chemical parameters							
			Moisture (%)	Ash (% TS)	Volatile solid (VS) (% TS)	C (% TS)	H (% TS)	N (% TS)	Organic matter (% TS)	LHV_d (MJ/kg)	LHV_{ar} (MJ/kg)	Cl (mg/g)	Hg (mg/MJ)
Year season	Summer	Average value (%)	75.80	10.90	89.60	47.71	7.59	2.55	61.50	24.53	4.05	0.28	4.3*10-5
		Deviation	4.20	4.10	3.40	3.11	0.66	0.28	9.10	3.05	1.40	0.14	3.9*10-5
		Coefficient of Variation	0.06	0.38	0.04	0.07	0.09	0.11	0.15	0.12	0.35	0.50	0.91
	Winter	Average value (%)	78.80	8.70	91.30	48.39	7.82	3.23	61.60	24.05	3.12	0.33	3.1*10-5
		Deviation	2.40	1.00	0.70	5.42	1.10	0.51	7.80	2.14	0.47	0.17	1.2*10-5
		Coefficient of Variation	0.03	0.11	0.01	0.11	0.14	0.16	0.13	0.09	0.15	0.52	0.39
	T-Test Value Summer - Winter		0.001	0.022	0.021	0.624	0.450	0.001	0.972	0.340	0.001	0.399	0.288
Day time	Diurnal	Average value (%)	77.80	10.30	90.00	48.51	7.70	2.95	60.60	24.00	3.40	0.27	3.4*10^{-5}
		Deviation	3.10	5.70	4.60	4.94	0.63	1.24	7.70	2.79	1.46	0.13	3.7*10^{-5}
		Coefficient of Variation	0.04	0.55	0.05	0.10	0.08	0.42	0.13	0.12	0.43	0.48	1.09

Sampling	Statistical parameter	Physical parameters			Chemical parameters							
		Moisture (%)	Ash (% TS)	Volatile solid (VS) (% TS)	C (% TS)	H (% TS)	N (% TS)	Organic matter (% TS)	LHV_d (MJ/kg)	LHV_{ar} (MJ/kg)	Cl (mg/g)	Hg (mg/MJ)
Night	Average value (%)	76.80	9.20	91.00	47.59	7.71	2.82	62.60	24.58	3.77	0.32	$4.1*10^{-5}$
	Deviation	3.40	2.80	1.10	4.03	0.80	1.28	9.00	3.84	1.37	0.13	$2.2*10^{-5}$
	Coefficient of Variation	0.04	0.30	0.01	0.08	0.10	0.45	0.14	0.16	0.36	0.41	0.54
T-Test Value Diurnal - Night		0.225	0.118	0.061	0.334	0.970	0.213	0.250	0.200	0.176	0.286	0.375
Annual	Average value (%)	77.30	9.40	91.00	48.05	7.71	2.89	61.60	24.29	3.59	0.31	3.8*10-5
	Deviation	3.40	3.40	2.80	4.02	0.75	0.53	8.30	2.60	1.15	0.18	2.9*10-5
	Coefficient of Variation	0.04	0.36	0.03	0.08	0.10	0.18	0.13	0.10	0.32	0.58	0.76

than that of the remainder of the year at 64.50%, but this was the only seasonal difference in the data. This value could not be related to the percentages of the fractions mentioned in Figure 17.

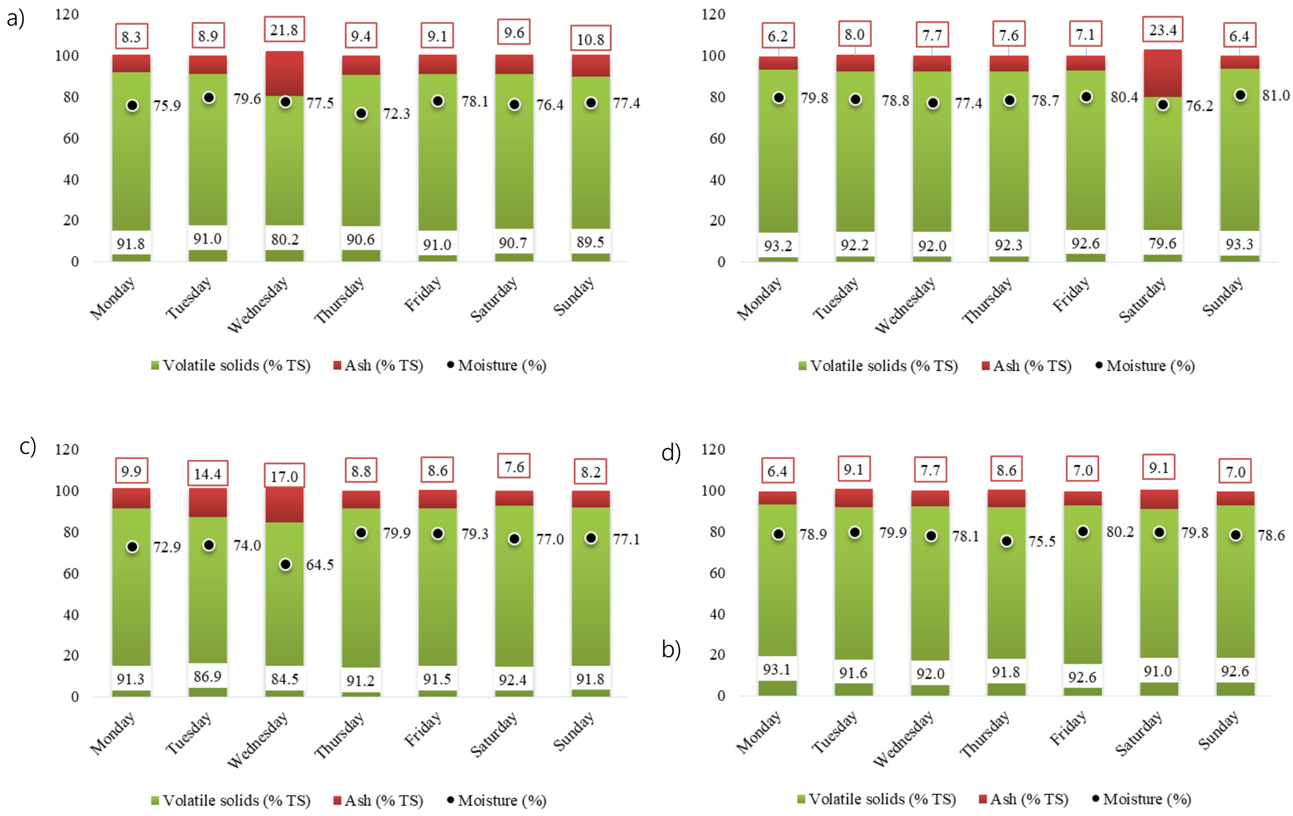


Figure 18. Daily evolution for physical parameters. a) Summer daytime. b) Winter daytime. c) Summer night-time. D) Winter night-time.

The average ash value of the residue was 9.40%, with a very high coefficient of variation of 36%, indicating the heterogeneity of the results. A relevant difference was found in the seasonal comparison of ash values, decreasing from 10.90% in summer to 8.70% in winter, possibly due to the lower proportion of fine particulates smaller than 20 mm in winter. The difference between the mean values concerning daytime is minor, with 10.30% for diurnal and 9.20% for night time. These results would be consistent with the T-Value obtained, being lower in the seasonal comparison (0.022) than in the daytime (0.118). The ash values were well above the average for point samples, which were collected in the daytime in both summer

(21.80%) and winter (23.40%). Table 11 shows how these anomalous values could be related to the higher prevalence of inert substances in these samples.

Volatile solids showed a mean value of 91.00% and no significant variation between daytime (90.00%) and night-time (91.00%) sampling. As with ash content, there was a slight seasonal variation in volatile solids, which were slightly higher in winter than in summer (91.30% vs 89.60%). Deviation (2.80%) and coefficient of variation (0.03) are low, providing a T-Value of 0.021 for summer-winter and 0.061 for diurnal-night, also concerning the differences between mean values. The punctual discordance between values for ash content corresponds to those of volatile matter, which was around 80% for these specific samples, also much lower than average.

The average values obtained for moisture content agreed with those of other screening studies, ranging from 67.80% (Cadavid-Rodriguez and Horan, 2012) to 85.90% (Kaless et al., 2016). In another study, the screening waste moisture varied from 73.30% to 85.00%, depending on the presence of a compactor in the WWTP pretreatment (Le Hyaric et al., 2009). The only data found in the literature on ash from screening is from (Cadavid-Rodriguez and Horan, 2012), who reported an ash content of 2.10% for screening waste from a WWTP in the Yorkshire region, a lower value than reported in this manuscript. Regarding volatile solids, the present results agree with those found in published characterizations of screening waste, ranging from 89.40% (Le Hyaric et al., 2009) to 94.00% (Wid and Horan, 2018). Cadavid-Rodriguez and Horan (2012), obtained a range of 91.40 and 94.00% values in the Yorkshire trials.

The same parameters were also calculated for each fraction present in the waste (

Table 13). Plastic showed the lowest moisture content because it has a lower field capacity (Qi et al., 2020); the remaining fractions had a similar water content to that of the general sample. Results can be considered homogeneous for the tests performed, with a coefficient of variation of less than 10% for all fractions. Regarding ash content, the 'other' fraction presented the highest value of 16.5% since it included a complex mixture that was difficult to separate and had a higher content of inert substances such as stones, glass and metals. This fraction contained the lowest percentage of volatile solids (84.2%).

Moisture content calculated for screening waste was close to that of sludge from sewage treatment plants, around 93% (Granados González, 2015). This value exceeds those obtained for MSW and rejects from MBT, with moisture contents of around 40% (Colomer, F.J. and Gallardo, 2007) and 30% (Ranieri et al., 2017a). Regarding the ash content of the sludge, its value was well below those presented by Gil-Lalaguna et al. (2014) for sludge from WWTP (39.04%). MSW presents a lower ash content, around 6% (Montiel Bohórquez and Pérez, 2019). By comparison, slightly higher values were obtained for the rejected fraction of MBT in investigations such as those of (Gallardo et al., 2014) and (Edo-Alcón et al., 2016) (10.69% and 10.31%, respectively). According to Gil-Lalaguna et al. (2014), sludge from WWTPs has a volatile solid content of 50.09%, considerably less than that of screening waste. MSW presents a volatile matter content between 60% and 80% (Adani et al., 2004).

To study the production of SRF from screening waste, some of the physical parameters for SRF produced from rejects were tested. In addition to the quality requirements for its classification in the UNE-EN 15359 (AENOR, 2012a) standard, it is necessary to consider others that are also important in the assessment of SRF quality, including moisture, volatile matter or ash content (Lorber et al., 2012; Rotter et al., 2011). Referenced moisture values for SRF range between 6.2% (Dunnu et al., 2009) and 15% Nasrullah et al., 2015), much lower than those obtained for screening waste. This fuel produced from MBT rejects has already undergone a drying treatment. Likewise, the ash content present in the SRF from rejects of MSW treatment plants is similar to that of the screening waste, presenting values around

Table 13. Average values of parameters analyzed for the fractions present in screening waste.

Fraction	Statistical parameter	Physical parameters				Chemical parameters						
		Moisture (%)	Ash content (% TS)	Volatile matter (% TS)	Organic matter (% TS)	C (% TS)	H (% TS)	N (% TS)	LHVd (MJ/kg)	LHVar (MJ/kg)	Cl (mg/g)	Hg (mg/MJ)
Sanitary Textiles	Average value	81.10	8.10	92.10	48.90	47.41	7.54	2.63	22.71	2.29	0.41	$4.60*10^{-5}$
	Deviation	2.10	2.60	2.40	7.93	0.91	0.42	0.19	0.29	0.36	0.04	$1.30*10^{-5}$
	Coefficient of Variation	0.03	0.32	0.03	0.16	0.02	0.06	0.07	0.01	0.16	0.10	0.28
Paper / Cardboard	Average value	81.70	5.60	94.30	61.30	44.42	7.12	2.50	22.36	2.06	0.16	$3.50*10^{-5}$
	Deviation	2.00	0.40	0.60	2.58	1.83	0.14	0.06	0.15	0.66	0.05	$0.70*10^{-5}$
	Coefficient of Variation	0.02	0.07	0.01	0.04	0.04	0.02	0.02	0.01	0.32	0.31	0.20
Vegetables	Average value	71.60	10.40	90.30	72.05	47.97	7.67	2.90	24.27	5.12	0.57	$4.00*10^{-5}$
	Deviation	3.60	0.90	0.30	6.62	0.67	0.28	0.93	2.33	0.97	0.22	$4.00*10^{-5}$
	Coefficient of Variation	0.05	0.09	0.00	0.09	0.01	0.04	0.32	0.10	0.19	0.39	1.00
Plastics	Average value	62.80	10.00	89.90	65.90	52.59	8.69	2.59	33.53	10.88	0.98	$1.60*10^{-5}$
	Deviation	6.30	0.90	1.20	10.22	1.07	0.78	0.72	1.90	0.11	0.16	$0.60*10^{-5}$
	Coefficient of Variation	0.10	0.09	0.01	0.16	0.02	0.09	0.28	0.06	0.01	0.16	0.38

Fraction	Statistical parameter	Physical parameters				Chemical parameters						
		Moisture (%)	Ash content (% TS)	Volatile matter (% TS)	Organic matter (% TS)	C (% TS)	H (% TS)	N (% TS)	LHVd (MJ/kg)	LHVar (MJ/kg)	Cl (mg/g)	Hg (mg/MJ)
Others	Average value	80.60	16.50	84.20	64.90	46.18	7.54	3.00	24.13	1.42	0.12	$6.00*10^{-5}$
	Deviation	7.50	1.80	1.80	5.25	5.23	1.31	0.12	1.57	0.42	0.04	$4.00*10^{-5}$
	Coefficient of Variation	0.09	0.11	0.02	0.08	0.11	0.17	0.04	0.07	0.30	0.33	0.67

ar As received. d Dry

9.8%, since inert fractions such as glass, metals or stones are eliminated in the biofuel generation process (Nasrullah et al., 2015). There are also references to much higher ash contents, around 15%, in SRF (Dunnu et al., 2009), as the ash content depends on the composition of the original waste. The volatile matter content of SRF is around 80% (Edo-Alcón et al., 2016; Nasrullah et al., 2014a).

3.2.2 Chemical parameters

- Elemental analysis and organic matter

The mean values and standard deviations obtained in the elemental analysis are shown in Table 12. The global average showed a composition of 48.05% of C, 7.71% of H and 2.89% of N, with coefficients of variation of 0.08, 0.10 and 0.18. Similar values were obtained for day and night samples. Regarding the seasonal variation, the C and H contents were statically similar, with a T-Value of 0.624 and 0.450. In the case of N, there was a significant decrease from winter (3.23%) to summer (2.55%), highlighted with a T-Value of 0.001 that exposes the minimal coincidence between the two subsets of data. This difference was due to high temperatures and lower humidity (Celaya and Castellanos, 2017). Night and diurnal data are statistically similar for C, H, and N, distinguishing the high T-Value for H, which at 0.970 shows high equality between the data referring to daytime.

Similar values were obtained throughout the sampling week (Figure 19). Four exceptional data points were among the samples cited for ash and volatile solid contents in section 3.2.1. In these cases, the average C and H contents were 40.35 ± 0.82% and 6.53 ± 0.16%, slightly lower than the average values. Results for N were maintained. These results imply the presence of less organic matter and, in turn, may be related to the increase observed in the ash content of the same sample, which has been previously justified by the possible presence of more inert materials.

There were no significant variations in the elemental analysis of the different fractions (

Table 13), although the higher percentage of H (8.69%) in plastics stands out due to the presence of hydrocarbons in its production (Barbarias et al., 2018).

Table 12 shows the values of organic matter content obtained by the Tyurin method. An annual average of 61.6% was observed, with no significant differences between values reported for summer and winter, obtaining a high T-Value (0.972). If diurnal time and night-time average are compared, there is a slight difference of 2%, consistent with a lower T-value (0.250). The resulting coefficient of variation, with values exceeding 10% of the mean value, highlights the variability observed in this parameter throughout the week.

The minimum organic matter content of all samples (48.87%) coincides with a minimum value of C and with the highest ash content. This organic matter and total organic carbon content (35.7%) indicate that anaerobic digestion would be a viable alternative. The obtained C/N ratio (16.67) makes this process possible (Ward et al., 2008). Cadavid-Rodríguez and Horan (2014) obtained similar results for C (50.2%) and N (2.6%). Boni et al. (2021) also achieved similar values, although with a higher range for C (32-49%). Suppose the results of organic matter content are compared with the characterization performed by Le Hyaric et al. (2009), obtaining values for organic matter between 37% and 51% for the different WWTPs, it is observed that the results in this report are slightly higher.

Hla and Roberts (2015), in a study on MSW in Australia, obtained a value of 45.8% carbon in the residue—only two points lower than that obtained in screening waste. In the same study, the hydrogen content (5.3%) was lower than that obtained in screening waste; in the case of nitrogen, the situation was similar, 0.92% for MSW and 2.89% for screening waste. Compared to the reject fraction, the reported results were higher for carbon, at 61.20% and lower for nitrogen (1.25%) (Edo-Alcón et al., 2016). The SRF obtained by Dunnu et al. (2009) contained 46.20% carbon, 7.65% hydrogen and 1.71% nitrogen. Nasrullah et al., (2015) obtained very similar data in elemental composition: 47.0%, 7.4% and 0.5%, respectively.

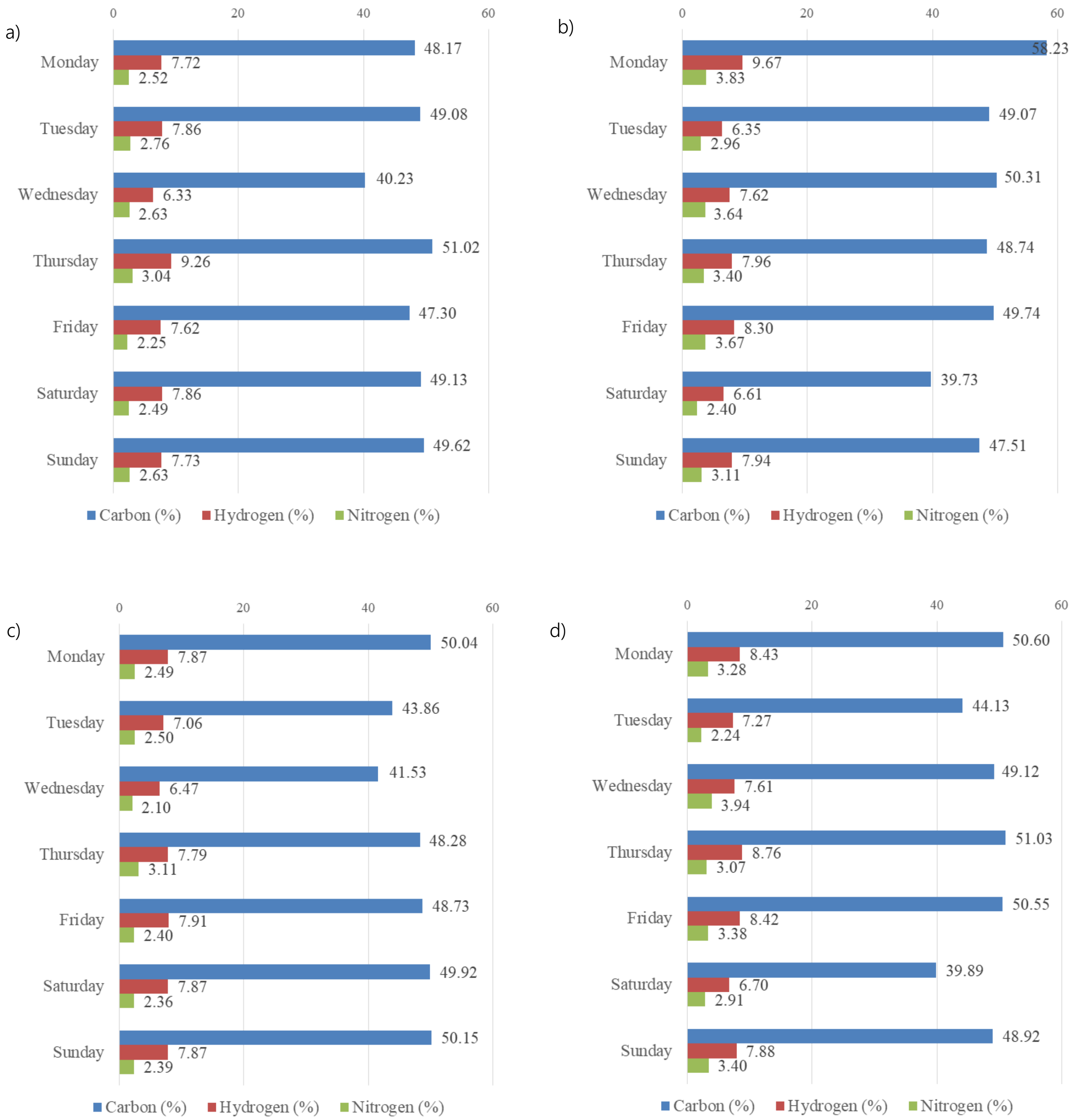


Figure 19. Daily evolution for elemental analyses. a) Summer daytime. b) Winter daytime. c) Summer night-time. d) Winter night-time.

The volatile solids values of 91.00% indicate a high level of biodegradability and, consequently, good possibilities for biogas production. The results obtained for organic matter (61.6%) are higher than some of those reported in studies on screening waste, and the C/N ratio (16.67) is considered suitable for biomethanization processes. Based on values and their comparison with the literature reviewed on the anaerobic digestion of screening, this process is considered a viable option for its valorization.

- Parameters for quality standards for SRF production

Regarding the possibility of producing SRF from screening waste, the following are the parameters listed in the standard UNE-EN 15359 (AENOR, 2012a).

The calorific values are summarized in Table 4. The calculations of the LHV for dry and wet mass were based on the moisture content and give a more realistic idea of the energy capacity of the raw waste. The resulting average value of the LHV on a dry basis was 24.29 MJ/kg, showing a coefficient of variation of 10.7% of the calculated average value. This high deviation is again due to the heterogeneity of the waste. The annual average result for LHV on a wet basis was 3.59 MJ/kg, a low value that reflecting the residue's high moisture content. In this case, the results show a high deviation of 32% since the calculation of this parameter already includes the variables moisture and hydrogen content.

The LHV value on a dry basis for the night-time mean (24.58 MJ/kg) was slightly higher than that obtained for the daytime (24.00 MJ/kg) because of the lower proportion represented by the 'other' fraction in the night-time samples. The value corresponding to the LHV on a dry basis for summer (24.59 MJ/kg) was slightly higher than that for winter (24.05 MJ/kg). The T-values obtained in the statistical analysis were 0.340 for seasonal and 0.200 for the diurnal-night variability. LHV results on a wet basis for summer (4.05 MJ/kg) were significantly higher than for winter (3.12 MJ/kg). The statistical analysis for the seasonal variable shows a T-Value of 0.001, which coincides with the result for humidity in the same comparison, being able to justify statistically the difference in LHV between summer and winter.

Regarding the analysis of LHV for the different fractions (

Table 13), the average value on a dry basis (33.53 MJ/kg) was higher for plastics, in agreement with similar characterizations, such as those of Tchobanoglous (1994).

The Cl content of the residue is also shown in Table 12 and gives mean values of 0.31 mg Cl/g residue, assuming 0.031% Cl in the residue. No significant differences were recorded for the seasonal (T-Value of 0.399) and hourly (0.286) variability, although slightly higher Cl percentages were shown for winter and night. Results are highly variable throughout the week, with standard deviations higher than 50%. This variation may again be due to the residue's heterogeneity and because samples for this test are small and could vary in composition.

According to the tests conducted on fractions (

Table 13), the highest values for Cl correspond mainly to the plastics fraction, with 1.16 mg Cl/g and to vegetables, with 0.68 mg Cl/g. The first case can be explained by the presence of Cl in the composition of the plastics, mainly those of the polyvinyl chloride (PVC) type (Xu et al., 2020). Regarding vegetables, the Cl content is influenced by irrigation water, which can be characterized by a high salt content (NaCl) and the presence of chlorophyll (Themelis, 2010). These results could explain the higher Cl content of the waste in winter when the presence of the fractions corresponding to vegetables and plastics was higher.

The Hg content, also shown in Table 12 and measured by mass spectrometry, gives results around 0.150 ppb. It is also important to note that several of the samples analyzed showed a value below the measurement threshold of the equipment (values marked as ND). To adapt these results to the quality requirements for SRF production, a unit conversion was conducted using the LCV on a wet basis as a reference. The average annual value obtained was 3.8 * 10^{-5} mg/MJ, considering the ND values as 0. The standard deviation represents 76% of the average, but given the low values, it is impossible to interpret significant variations between days of the week, daytime and night-time samples or seasons. According to this statement, the T-Values obtained are 0.288 for summer-winter and 0.375 for daytime.

The only LHV data for screening waste collected in the studies consulted in the literature show values varying between 16 MJ/kg (Canler and Perret, 2004) and 17.72 MJ/kg (Sidwick, 1991), both on a dry basis. These are lower than the average value of this study and any of the results of the 28 samples analyzed. Values obtained for Cl and Hg content could not be compared with those of other studies since no quantifications of these elements were found for this type of waste.

The LHV of the screening waste presents a similar value to those collected in the MSW reject fraction characterization, with maximum values of 21.36 MJ/kg (Gallardo et al., 2014) or 19.06 MJ/kg (Ramos Casado et al., 2016) were observed. SRF produced from MSW also presented values ranging from 14.78 MJ/kg (Vounatsos et al., 2015) to 23.56 MJ/kg (Vainikka et al., 2012). Regarding the Cl content, a characterization of the MSW residual fraction reported values ranging from 0.30% (Bessi et al., 2016) to 1.21% (Ramos Casado et al., 2016). In the case of SRF, the product obtained by Nasrullah et al. (2015) showed a value of 0.6%, similar to those reported by Velis et al. (2012), with 0.69% in a study on the influence of inlet streams for SRF production. Ramos Casado et al. (2016) obtained a Cl content of 0.4% in a characterization of SRF produced in Navarra. All the values indicated are much higher than those obtained for screening waste, whose maximum and average were 0.07% and 0.03%. The remaining MSW fractions analyzed by Edo Alcón et al. (2016) present, as a minor result, a Hg content of 0.05 ppm, which was not comparable to the value obtained for screening waste.

The results obtained for these parameters were compared with the requirements for the SRF (AENOR, 2012a). Regarding Cl and Hg contents, the potential SRF produced from screening waste could meet class 1. LHV, as received, would comply with class 5 of the classification. This calorific value corresponds to a moisture content of 77.3%, so there is much scope for improvement in producing a higher-quality SRF with lower moisture content. This margin of improvement could be achieved by implementing novel screening treatment systems, such as low-temperature dryers or natural greenhouses.

4 CONCLUSIONS

A characterization campaign was conducted on Biofactoría Sur (Granada) screening waste. Composition of the waste and a series of chemical and physical parameters were determined, providing the following conclusions:

- It is a very heterogeneous waste that does not show significant variability between days of the week or between daytime and night-time. Neither there are seasonal differences. Consequently, to implement a new process at the wastewater treatment plant, it will not be necessary to consider time as a variable.
- Determination of the parameters shows a high average of organic matter content (61.6%) and a carbon/nitrogen ratio (16.67). These results together, with the percentage in volatile solids (91.0%), would be adequate to consider anaerobic digestion as an alternative to landfilling.
- The predominant fraction present in screening waste consists of sanitary textiles (52.10%), followed by paper/cardboard (11.80%), vegetables (5.50%) and plastics (5.00%). The screening waste composition is comparable to rejects from mechanical biological treatment processes, so similar WtE alternatives could be investigated, opening up the possibility for SRF production.
- According to the standards, SRF production from screening waste could be an alternative. Screening waste meet the quality standards for lower heating value (3.59 MJ/kg on wet basis) as an economic parameter, chlorine content (0.031%) as a technical requirement, and mercury content (3.8×10^{-5} mg/MJ)

as an environmental impact factor. It is necessary to explore the optimal conditions for producing densified and non-densified SRF and its subsequent characterization for use as fuel.

CHAPTER 3

PRODUCTION AND CHARACTERIZATION OF SOLID RECOVERED FUEL FROM SCREENING WASTE[3]

1 INTRODUCTION

WWTP management companies need to look for alternatives to the current disposal of screening wastes in landfills, thus contributing to circularity in their way to zero-waste. Among the possible alternatives to be considered is energy recovery, through the biofuel production (Shehata et al., 2022). In this sense, and as explained in the previous Chapter of this report, the screening waste would satisfy the requirements of ISO 21640:2021 to produce SRF. The results show suitable values for LHV as an economic parameter, chlorine content as a technical requirement and mercury content as an environmental factor. In addition, the characterization of the screening waste showed results similar to those present in the reject from mechanical biological treatment of MSW.

[3] The results shown in this Chapter were presented in: De la Torre-Bayo, J. J.; Zamorano, M.; Torres-Rojo, J. C.; Rodríguez, M. L.; Martín-Pascual, J. Analyzing the Production, Quality, and Potential Uses of Solid Recovered Fuel from Screening Waste of Municipal Wastewater Treatment Plants. Process Saf. Environ. Prot. 2023, 172, 950–970. https://doi.org/10.1016/j.psep.2023.02.083

Producing biofuel from waste should play a relevant role as an alternative to the use of fossil fuels (Yan et al., 2021). The objective of SRF production is to decrease the reliance on fossil fuels in combustion, gasification, and pyrolysis processes (Nasrullah et al., 2014b). By doing so, the densification of the final product not only lowers the environmental footprint associated with managing waste (Hettiarachchi et al., 2019), but also cuts the expenses of handling, transporting, and storing wood-based products along the supply chain (Whittaker and Shield, 2017). Furthermore, pelletizing in agro-biowaste compost has the potential to reduce the environmental impact by over 63% (Sarlaki et al., 2021). The feasibility of the SRF production and utilization process must be studied in technical, economic, social, and environmental terms. For improved decision-making, cost/benefit analyses have been developed for SRF from MSW for use in cement plants (Iacovidou et al., 2018) or gasification processes (Arena et al., 2015). In the environmental and social aspects, an analysis with more variables of the environmental impact derived from the exposed processes is necessary (Aghbashlo et al., 2022). Life Cycle Assessment (LCA) is one of the most useful and established methodologies (Ferrari et al., 2021) being a powerful computerized tool that, in the case of SRF, analyzes impacts derived from its production (Grosso et al., 2016) and use (Breckel et al., 2013).

No studies have been reported that analyse the possible use of WWTP screening waste for energy recovery through the production of SRF. Thus, among the wastes that Sarc et al. (2014) consider suitable for SRF production, as the most commonly used, are rejects from biological treatment of municipal waste (Jędrczak and Suchowska-Kisielewicz, 2018) and construction and demolition by-products (Nasrullah et al., 2015) with EWC codes 19 12 12 and 17 09 04, respectively. Screening waste (EWC code 19 08 01) do not appear among them. However, ISO 21640:2021 (AENOR, 2021a), which in 2021 updated the specifications and classes of EWCs, already considers "*solid waste from urban wastewater treatment*" as a possible origin.

The present Chapter aim to study the technical feasibility of producing non-densified and densified SRF from screening waste as an alternative to its problematic disposal in landfills. In addition, the determination of the properties of the SRF generated, and the evaluation of its quality has been developed based on an exhaustive study of the existing regulations on SRF and densified biofuels for evaluating the feasibility of using SRF as an alternative to fossil fuels in combustion or gasification processes.

2 MATERIALS AND METHODS

The work developed to achieve this set of objectives includes the following four stages (Figure 20) which are described in the following sections: (i) production of SRF at a laboratory scale; (ii) basis for establishing the quality of the SRF produced; (iii) determination of the quality of the SRF; (iv) determination of the potential uses of the SRF.

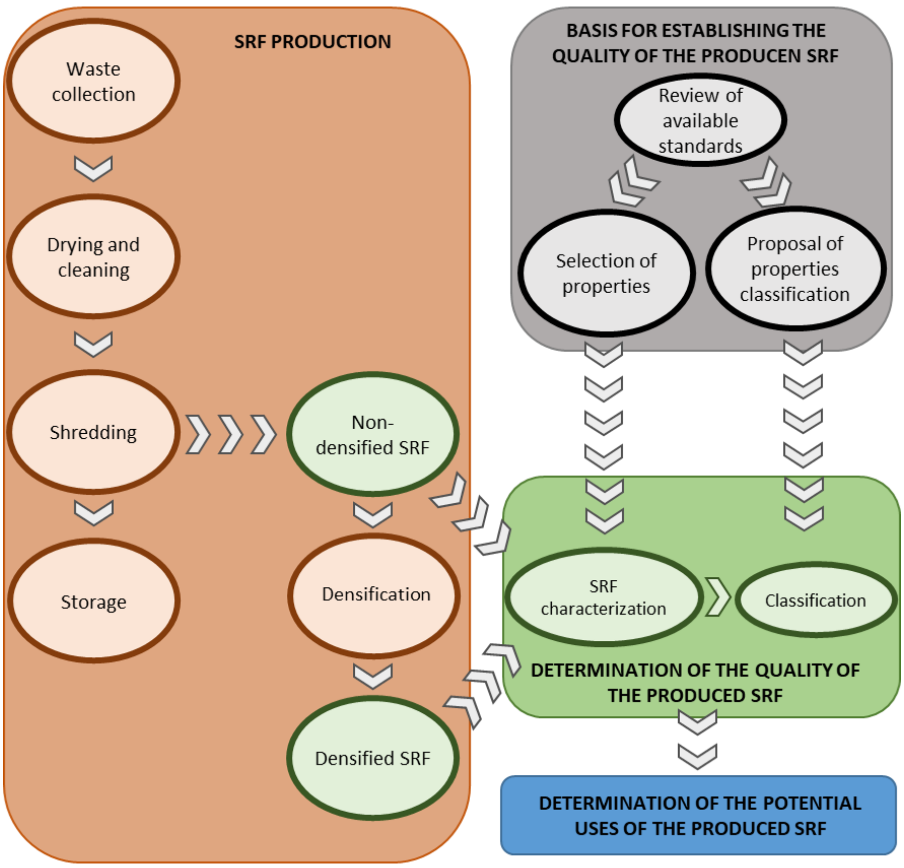


Figure 20. Study phases for Solid Recovered Fuel (SRF).

2.1 SRF PRODUCTION AT A LABORATORY SCALE

In this study, laboratory-scale production of SRF from screening waste has been carried out. Both non-densified and densified SRF were produced. For this purpose, the production process shown in Figure 20, described below, was followed.

2.1.1 Collection of material

The screening waste used came from the Biofactoría Sur of Granada. To work with the most representative material possible, several samples were taken. Specifically, 16 samples of approximately 8 kg were taken from the output of the screen compactor. Two pieces per week, throughout October and November 2021, were collected on random days of the week and during daytime and night-time hours (Figure 21a).

2.1.2 Drying and cleaning

Once each sample was received in the laboratory, it was dried. For this purpose, the sample was spread on metal trays, and after 24 hours in an oven at 105 °C, it was mixed to be introduced again in the oven at the same temperature for another 24 hours. Once dry, the undesirable fractions that could affect the process, especially those of an inert nature, were removed and prepared for crushing (Figure 21b).

2.1.3 Shredding

The dry waste was shredded using a Viking GE450 garden bio-shredder with a power of 2500 W (Figure 21e), which yielded the non-densified SRF, shown in Figure 21c, characterized by a light matrix and a cottony appearance, due to the high content of sanitary textiles.

2.1.4 Storage

A portion of the SRF produced was stored at room temperature for characterization. The rest was used for the production of densified SRF.

2.1.5 Densification

Finally, to produce the densified SRF, non-densified SRF was quartered to obtain a homogeneous sample and pelletized using a flat die type press, KAHL 14-175, with a drive power of 3 kW and a feed capacity of 50kg/h (Figure 21f). The pelletizing process is subject to input variables including particle size, moisture, the diameter and compression length of the die, temperature (Garcia-Maraver et al., 2015), and the presence of additives (Said et al., 2015). After preliminary tests, and because of the low density of the residue due to the content of sanitary textiles, work was carried out at intensities lower than 7 A and temperatures that did not exceed 29 °C. Likewise, the homogeneity in the particle size of the sample was not considered a variable. Therefore, three operating variables were considered for the pelletizing process: moisture of the inlet stream and the die`s diameter and compression length. In the case of moisture, studies of the pelletization of rejects from biological and mechanical treatment of municipal waste were taken as a reference, with maximum moisture percentages of 45% (Zafari and Kianmehr, 2014), which allowed establishing four operation values, 10, 20, 30 and 40%. These values were achieved by spraying the dry sample, obtained after the drying and shredding processes, with water until reaching the values required for each test, taking into account for this purpose the moisture value of the stored non-densified SRF, obtained in its characterization at the time of its use. In terms of diameter (Dd) and compression length (Lc) of the pelletizing dies, which determines their compression ratio (Dd/Lc), five available dies were used with diameters of 6 or 8 mm and compression lengths of 16, 20-, 24-, 32- or 48-mm. Table 14 shows the designation and characteristics of the five dies used. Finally, to lower production costs, and given that studies of pellet production from urban waste showed the possibility of manufacturing them without the need to add additives (Rezaei et al., 2020), it was decided not to use additives for the densification of the material. As a result, 20 pellet samples were obtained, whose designations are given in Table 14, which were stored at room temperature for characterization.

Figure 21. Non-densified and densified Solid Recovered Fuel (SRF) production process. a) Waste after collection. b) Dry waste. c) Non-densified SRF. d) Densified SRF. e) Bio shredder Viking GE450. f) KAHL 14-175 Pelletizer

Table 14. Denomination of pellet samples produced

Pelletizing die identification	Diameter of pelletizing die (Dd) (mm)	Compression length of pelletizing die (Lc) (mm)	Compression ratio (Dd/Lc)	Pelletizer inlet stream moisture (%)			
				10	20	30	40
D6L20	6	20	6/20	P-10-D6L20 	P-20-D6L20 	P-30-D6L20 	P-40-D6L20 
D6L24	6	24	6/24	P-10-D6L24 	P-20-D6L24 	P-30-D6L24 	P-40-D6L24 
D8L16	8	16	8/16	P-10-D8L16 	P-20-D8L16 	P-30-D8L16 	P-40-D8L16 
D8L32	8	32	8/32	P-10-D8L32 	P-20-D8L32 	P-30-D8L32 	P-40-D8L32 
D8L48	8	48	8/48	P-10-D8L48 	P-20-D8L48 	P-30-D8L48 	P-40-D8L48 

2.2 Basis for establishing the quality of the SRF produced

The quality of the SRF produced was established based on the classification of the set of properties that characterize it. These characteristics are related. On one hand, is its use as fuel, as well as its final use, regardless of its presentation in densified form or not, taking into account economic, technical, and environmental aspects. On the other hand, in the case of densified SRF, it is necessary to consider other properties directly related to its densified form and which affect its storage, transport, and feeding in the thermochemical processes in which it can be used.

Given the diversity of existing reference standards, as well as the absence of specific standards to classify pellets generated from screening waste or similar wastes (e.g., MSW), it was decided to develop our proposal to organize the identified properties based on a set of existing standards which were used to determine the optimal conditions to produce densified SRF in the form of pellets. For this purpose, the following stages were followed Figure 20: (i) review of the available standards; (ii) selection of properties for the characterization of the produced SRF; (iii) proposal of properties classification. These stages are described below.

2.2.1 Review of available standards

In the first place, the review of standards applicable to manufactured SRF included those properties that evaluate its quality as a fuel. For densified fuel, this was completed with a set of properties to assess the quality of the densified form as pellets. Table 15 and Table 16 list the standards used as a reference to evaluate the quality of the SRF produced as fuel and pellets, respectively, indicating end uses, categories, and requirements concerning their properties.

Table 15. Quality of Solid Recovered Fuel (SRF) as fuel. Reference standards

Standard	Application area	Final use	Included properties							
			Classes	LHV	Cl	Hg	Moisture	Ash	Particle size/ density	Heavy metal
ISO 21640:2021. Solid recovered fuels — Specifications and classes (AENOR, 2021a).	International	N.S.	✓	✓	✓	✓				
UNI 9903-1:2004. Non mineral refuse derived fuels - Specifications and classification (Ente Nazionale Italiano di Unificazione (UNI), 2004).	Italy	Cement plants, EfW	✓	✓	✓	✓	✓	✓		✓
Arrêté du 23 mai 2016 relatif à la préparation des combustibles solides de (Ministère de l'environnement, 2016).	France	EfW		✓	✓	✓				✓
RAL-GZ 724 (2008) Quality and test instructions Solid Recovered Fuels (Gütegemeinschaft Sekundärbrennstoffe und and e. V., 2008).	Germany	Cement plants, lime kilns, EfW	✓	✓	✓	✓	✓	✓		✓
WRAP. A classification scheme to define the quality of waste derived fuels **(Waste & Resources Action Programme, 2013)**.	United Kingdom	EfW	✓	✓	✓	✓	✓	✓	✓	✓
No. 389/2002 in the Incineration Waste, BGBI (BMLFUW, 2002).	Austria	Cement plant, EfW, Co-incineration	✓	✓	✓	✓		✓		✓
Limit values set by authorities for individual permits for cement plants in Spain (Schorcht et al., 2013).	Spain	Cement plants			✓	✓				✓
Limit values set by authorities for individual permits for cement plants in Belgium (Schorcht et al., 2013).	Belgium	Cement plants			✓	✓				✓

Standard	Application area	Final use	Classes	Included properties						
				LHV	Cl	Hg	Moisture	Ash	Particle size/ density	Heavy metal
SFS 5875 (2000) Solid Recovered Fuel - Quality Control System (General Industry Federation, 2008).	Finland	Incineration, Co-incineration	✓		✓	✓				✓
Guidelines on Usage of Refuse Derived Fuel in Various Industries. Draft of July 2018 (Health et al., 2018).	India	Cement plants	✓	✓	✓		✓	✓	✓	
Act on the Promotion of Saving and Recycling of Resources Enforcement Regulation (Addendum 7) (Korea, 2002)	South Korea	N.S.	✓	✓	✓	✓	✓	✓	✓	✓

Table 16. Quality of Solid Recovered Fuel (SRF) as pellet. Reference standards

Standard	Application area	Extraction	Final Use	Classes	Included properties								
					Dp	Lp	PD	BD	DU	Mp	Ash	LHV	Chemical elements
ISO 17225:2021. Biocombustibles sólidos. Especificaciones y clases de combustibles (AENOR, 2021b)	International	Wood or herbaceous biomass	Commercial and residential applications. Industrial.	✓	✓	✓		✓	✓	✓	✓	✓	✓
O NORM M7135 ***(O NORM M7135 - Compressed***	Austria	Wood or herbaceous biomass	Industrial		✓	✓	✓	✓	✓	✓	✓	✓	✓

Standard	Application area	Extraction	Final Use	Classes	Included properties								
					Dp	Lp	PD	BD	DU	Mp	Ash	LHV	Chemical elements
***wood or compressed bark in natural state, pellets and briquettes. Requirements and test specifications*, 2002)**													
DIN 51731 and DIN PLUS (Norm, 2002)	Germany	Wood or herbaceous biomass	Specific boilers for pellets. Industrial	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Agro and Agro+ (Narra et al., 2012)	France	Agricultural origin	Incineration, boilers or furnaces.	✓	✓	✓		✓	✓	✓	✓	✓	
SS187120	Sweden	N.S.	N.S.	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Pelet Fuel Institute Standards ***(SS187120 Pelet Fuel Institute Standards*, 2014)**	United States of America	Wood	N.S.	✓	✓	✓		✓	✓	✓	✓	✓	✓
NY/T 1878-2010 (Ministry of Agriculture, 2010)	China	Wood or herbaceous biomass	N.S.	✓	✓	✓	✓			✓			

Standard	Application area	Extraction	Final Use	Classes	Included properties								
					Dp	Lp	PD	BD	DU	Mp	Ash	LHV	Chemical elements
JAS Standards for Wood Pellets for Non-Industrial Use (Ministry of Agriculture, 2021)	Japan	Wood	Non industrial	✓	✓	✓		✓	✓	✓	✓	✓	✓

2.2.2 Selection of properties for the characterization of the produced SRF

Based on the review of available standards, the properties considered to characterize the produced SRF were selected and are shown in Table 17, including the standards analytical methods.

Table 17. Properties analyzed to determine the quality of the manufactured Solid Recovered Fuel (SRF).

SRF type		Properties	Unit	Standard analytical method
Densified SRF	Non-densified SRF	Lower Heating Value (LHV)	MJ/kg	UNE-EN 15400:2011 (AENOR, 2011d)
		Cl Content	%	UNE-EN ISO 10304-1:2009 (AENOR, 2009)
		Hg Content	mg/MJ	UNE-EN 15411:2012 (AENOR, 2012b)
		Ash Content	%	UNE-EN 15403:2011 (AENOR, 2011b)
		Moisture (M, for non densified SRF). (Mp, for densified SRF)	%	UNE-EN 15414-3:2011 (AENOR, 2011a)
		Pellet Diameter (Dp)	mm	UNE-EN 16127:2012 (AENOR, 2012c)
		Pellet Length (Lp)	mm	UNE-EN 16127:2012 (AENOR, 2012c)
		Pellet Density (PD)	kg/m3	UNE-EN 15150:2012 (AENOR, 2012d)
		Bulk Density (BD)	kg/m3	UNE-EN 15103:2010 (AENOR, 2010a)
		Durability (DU)	%	UNE-EN 15210-1:2010 (AENOR, 2010b)
		Hardness (HD)	kgf	Determination made by using a manual hardness tester (Amandus Khal mod. 21465)

In the non-densified SRF, the following were selected: LHV, Cl content, Hg content, ash, and moisture. The first three, which have great relevance from an economic, technical, and environmental point of view, were selected because they are included in ISO 21640:2021 (AENOR, 2021a). Moisture and ash content, although not limited to ISO 21640:2021 (AENOR, 2021a), were incorporated because they are present in most of the standards included in the ISO/TR 21916:2021 report (ISO/TC 300, 2021). The report mentioned above consists of an extensive study on the quality of SRFs based on a literature review and consultations with producers, concluding that in 99% of the exposed cases, the moisture and ash content of the SRF produced is evaluated. On the other hand, other properties incorporated in some of the standards (Table 15), such as particle size and density, were considered of little relevance for this work since, in addition to being present in only three of the eleven standards reviewed, the product obtained, as indicated above, is difficult to break down into particles. Finally, the content of other heavy metals, in addition to Hg, was not considered due to the nature of the waste obtained.

In the case of densified SRF, a total of eleven properties were used, five of them were included in the characterization of non-densified SRF and six additional ones are related to the conditioning of SRF in pellet form, including diameter, length, pellet density, bulk density, durability, and hardness. Diameter and length are present in all standards; bulk density and durability are in all but one, and pellet density is in half of the revised standards. Finally, hardness was included, despite not being included in any of the standards reviewed (García-Maraver et al., 2011), because it is a property linked to pellet handling and storage (García-Maraver et al., 2011; Gilbert et al., 2009; Said et al., 2015), and it has been extensively analyzed in numerous studies such as torrefaction analysis (Haykiri-Acma and Yaman, 2022) or the effect of additives in pellets (Nursani et al., 2020) , including manuscripts that focus on the relation of hardness to other fuel parameters (Suryawan et al., 2022).

As a result of the above analysis, Table 18 and Table 19 show the standards, properties, and the included values applied to the SRF without densification and SRF pellets, respectively. The reference units were taken from ISO 21640 (AENOR, 2021a). It was necessary to convert units for some standards. In the case of Hg content, and given the impossibility of unit conversion, it was decided to include the two units that appear in the standards. On the other hand, it was observed that the reference values used for moisture content for non-densified and densified SRF are different; this is because moisture, unlike LHV, and ash, Cl, and Hg contents, is a property that affects the logistics of SRF, so it was evaluated based on the pellet standards, in addition to being conditioned by the manufacturing process. Finally, hardness was analyzed as a relevant property of SRF, but it does not appear in any standard. So, it was compared, based on the literature which considers it a pertinent parameter in pellet quality.

2.2.3 Proposed classification of properties

Given the absence of an SRF property classification applicable to the specific case of the waste under consideration and based on the values established in the standards analyzed for the selected properties (Table 17), a proposal will be prepared to lead to a classification that includes four categories, according to the levels indicated below:

- Class 1 (C1). It will correspond to the range of optimum values for the property under consideration and includes those values met in 100% of the standards selected for this study.
- Class 2 (C2). It will correspond to average quality values for the property under consideration and includes a range that met at least 50% of the standards selected for this study without reaching 100%.
- Class 3 (C3). It will correspond to low-quality values for the property under consideration and includes a range that met at least 25% of the standards selected for this study without reaching 50%.

Table 18. Standards and recommended values for the fuel properties of Solid recovered fuel (SRF) [4]. Lower heating value (LHV). Chlorine content (Cl). Mercury content (Hg). Ash content (Ash). Moisture (M).

	Standard	LHV	Cl	Hg		Ash	M
		MJ/kg	%	mg/MJ	mg/kg	%	%
1	ISO 21640:2021. Solid recovered fuels — Specifications and classes (AENOR, 2021a).	≥3 ≥25	≤3 ≤0.2	≤0.15 ≤0.02			
2	UNI 9903-1:2004. Non mineral refuse derived fuels - Specifications and classification (Ente Nazionale Italiano di Unificazione (UNI), 2004).	≥15 ≥25	≤1		≤3	≤20 ≤15	≤25 ≤15
3	Arrêté du 23 mai 2016 relatif à la préparation des combustibles solides de récupération (Ministère de l'environnement, 2016).	≥12	≤1.5		≤3		
4	RAL-GZ 724 (2008) Quality and test instructions Solid Recovered Fuels (Gütegemeinschaft Sekundärbrennstoffe und and e. V., 2008).	≥13 ≥27	≤1 ≤0.7		≤1 ≤0.5	≤20 ≤9	≤35 ≤12.5
5	WRAP. A classification scheme to define the quality of waste derived fuels **(Waste & Resources Action Programme, 2013)**.	≥6.5 ≥25	≤0.8 ≤0.2	≤0.12 ≤0.04		≤50 ≤10	≤40 ≤10
6	No. 389/2002 in the Incineration Waste, BGBI (BMLFUW, 2002).	≥11 ≥25	≤1.5 ≤0.8	≤0.075		≤35 ≤10	
7	Limit values set by authorities for individual permits for cement plants in Spain (Schorcht et al., 2013).		≤2		≤10		
8	Limit values set by authorities for individual permits for cement plants in Belgium (Schorcht et al., 2013).		≤2		≤5		
9	SFS 5875 (2000) Solid Recovered Fuel - Quality Control System (General Industry Federation, 2008).		≤1.5 ≤0.15		≤0.5 ≤0.1		
10	Guidelines on Usage of Refuse Derived Fuel in Various Industries. Draft of July 2018 (Health et al., 2018).	≥12.5 ≥18.5	≤1 ≤0.5			≤15 ≤10	≤20 ≤10

11	Act on the Promotion of Saving and Recycling of Resources Enforcement Regulation (Addendum 7) (Korea, 2002)	≥12.5 ≥27.2	≤2 ≤0.3		≤1.2 ≤0.6	≤20 ≤4	≤25 ≤10

[4] The cells with several values show that the standard establishes different classes, so the established limits are included.

Table 19. Standards and recommended values for Solid Recovered Fuel (SRF) pellet properties [5]. Pellet moisture (Mp). Pellet diameter (Dp). Pellet length (Lp). Pellet density (PD). Bulk density (BD). Durability (DU).

	Standard	Mp	Dp	Lp	PD	BD	DU
		%	mm	mm	kg/m^3	kg/m^3	%
1	ISO 17225:2021 Biocombustibles sólidos. Especificaciones y clases de combustibles (AENOR, 2021b)	≤10 ≤12 ≤15	≥ 6 ≤ 25	≥ 3.15 ≤ 50		≥ 550 ≤ 750	≥ 96.0 ≥ 97.7
2	O NORM M7135 (***O NORM M7135 - Compressed wood or compressed bark in natural state, pellets and briquettes. Requirements and test specifications***, 2002)	≤10	≥ 4 ≤ 10	≥ 20 ≤ 50	≥ 1120	≥ 540	≥ 97.7
3	DIN 51731 and DIN PLUS (Norm, 2002)	≤10 ≤12	≥ 4 ≤ 10	≥ 20 ≤ 50	≥ 1000	≥ 540	≥ 97.7
4	Agro and Agro+ (Narra et al., 2012)	≤11 ≤15	≥ 6 ≤ 8	≥ 10 ≤ 30	≥ 1200 ≤ 1400	≥ 580	≥ 92
5	SS187120	≤10 ≤12	≤ 25	≥ 3.15 ≤ 40		≥ 500	≥ 98.5 ≥ 99.2
6	Pelet Fuel Institute Standards (***SS187120 Pelet Fuel Institute Standards***, 2014)	≤8 ≤10	≥ 5.84 ≤ 7.25	≥ 3.15 ≤ 38.1		≥ 609 ≤ 737	≥ 95
7	NY/T 1878-2010 (Ministry of Agriculture, 2010)	≤13	≤ 25	≥ 3.15 ≤ 40	≥ 1000		
8	JAS Standards for Wood Pellets for Non-Industrial Use (Ministry of Agriculture, 2021)	≤10	≥ 6 ≤ 8	≥ 3.15 ≤ 40		≥ 600	≥ 96.5

[5] The cells with several values show that the standard establishes different classes, so the established limits are included.

- Not recommended (NR). Finally, if the value of property results in quality outside the limits to be established in the indicated classes, it will be considered unsuitable or not recommended, corresponding to values included in less than 25% of the consulting standards.

2.3 DETERMINATION OF THE QUALITY OF THE SRF PRODUCED

To determine the quality of the SRF produced, the analytical methods listed in Table 19 were applied. Each determination was performed in triplicate to obtain an average value. In the case of densified SRF, to determine the most suitable production conditions, the correlation between the independent variables (initial humidity, compression length, and die diameter) and pellet properties was studied (Table 17) using R (V. 4.1.1), a free programming environment and language with a focus on statistical analysis.

2.4 DETERMINATION OF THE POTENTIAL USES OF THE SRF PRODUCED

Finally, taking ISO/TR 21916:2021 (ISO/TC 300, 2021) as a reference, three potential uses will be considered for the SRF produced: cement plants, power plants, and gasification. The minimum and maximum values to be taken as reference established for the SRF properties considered in the previous report (LHV, Cl content, Hg content, ash, and moisture) are shown in Table 20.

Table 20. Maximum and minimum values referenced for the use of the Solid Recovered Fuel (SRF).

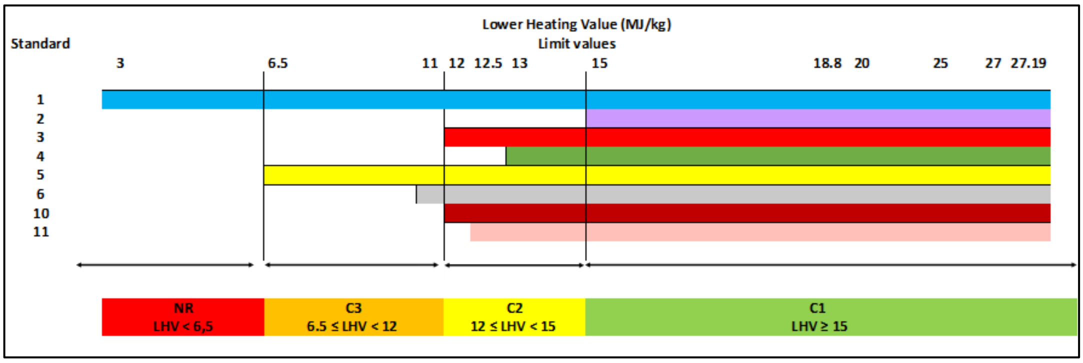
Uses	Properties				
	LHV (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	M or Mp (%)
Cement plants	15.6-32.4	0.05-3.89	N.S.5	5.27-30.60	1.4-35.0
EfW	13.24-32.98	0.10-1.16	0.001-0.209	7.40-23.60	3.8-34.1
Gasification	15.4-25	0.26-0.65	0.02-0.04	6.30-21.20	2.5-15.0

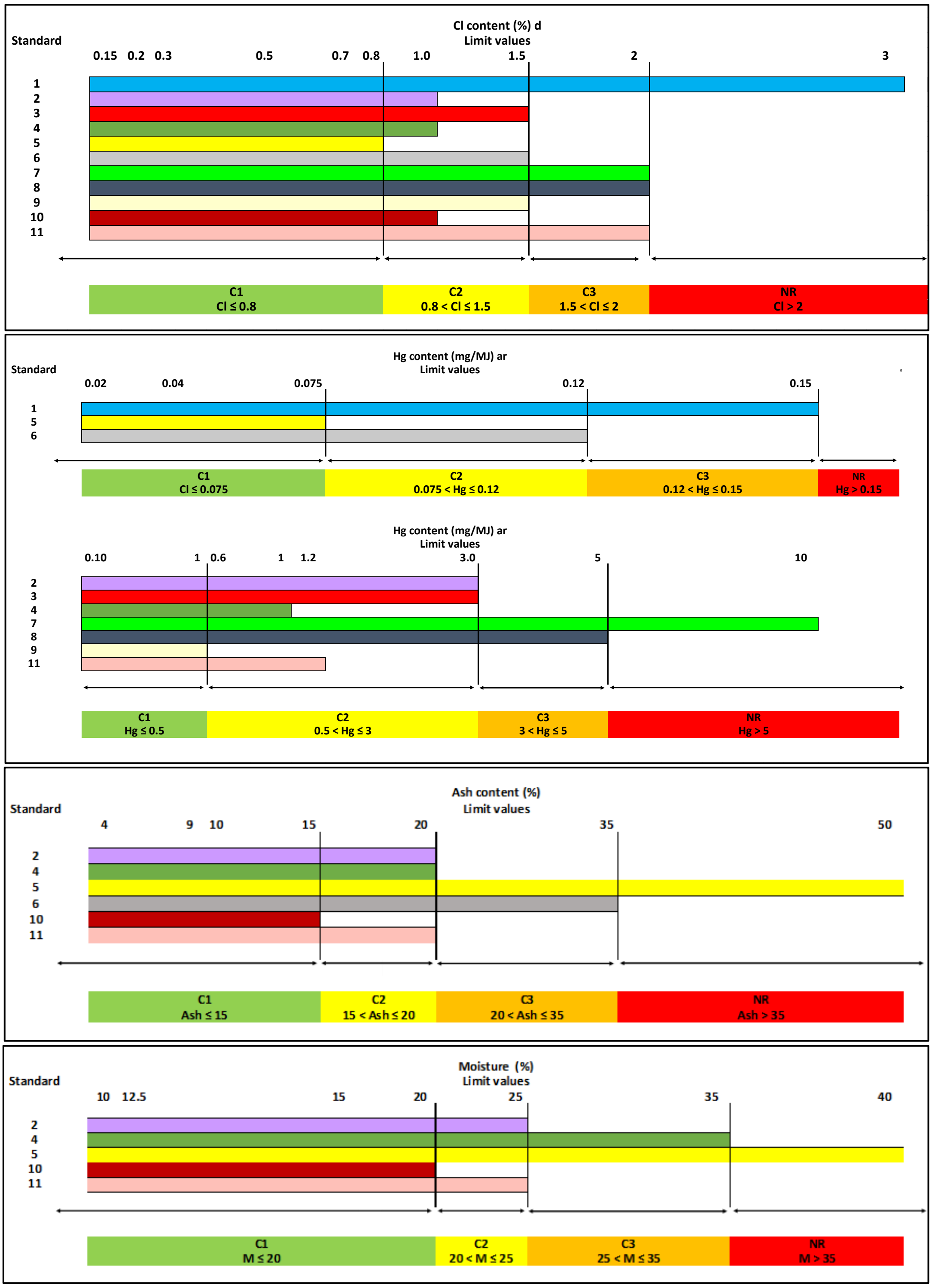
3 RESULTS AND DISCUSSION

The non-densified and densified SRF were produced at a laboratory scale according to the described procedure. Then we proceeded to establish the basis for defining the classification of its properties, based on which samples were characterized and classified, and their potential use was found. The results obtained are presented, analyzed, and discussed below.

3.1 PROPOSED CLASSIFICATION OF SRF PROPERTIES

Taking into account the criteria established in the methodology described above and the properties and values included in the revised standards (Table 18 and Table 19), a classification proposal for these properties was prepared, establishing limit values for the different classes defined (Class 1, Class 3, Class 3, and Not recommended). Figure 22 and Figure 23 show the proposed ranges, with the limit values established for each class and properties considered for the non-densified and densified SRF.





[1] C1: Class 1. C2: Class 2. C3: Class 3. NR: Not recommended

1. ISO 21640:2021. Solid recovered fuels — Specifications and classes.
2. UNI 9903-1:2004. Non mineral refuse derived fuels – Specifications and classification.
3. Arrêté du 23 mai 2016 relatif à la préparation des combustibles solides de récupération en vue de leur utilisation dans des installations relevant de la rubrique 2971 de la omenclatura des installations classées pour la protection de l'environnement.

4. RAL-GZ 724 (2008) Quality and test instructions Solid Recovered Fuels.
5. WRAP. A classification scheme to define the quality of waste derived fuels.
6. No. 389/2022 in the Incineration Waste, BGBI.
7. Limit values set by authorities for individual permits for cement plants in Spain.
8. Limit values set by authorities for individual permits for cement plants in Belgium.
9. SFS 5875 (2000) Solid Recovered Fuel – Quality Control System.
10. Guidelines on Usage of Refuse Derived Fuel in Various Industries. Draft of July 2018.
11. Act on the Promotion of Saving and Recycling of Resources Enforcement Regulation (Addendum 7)

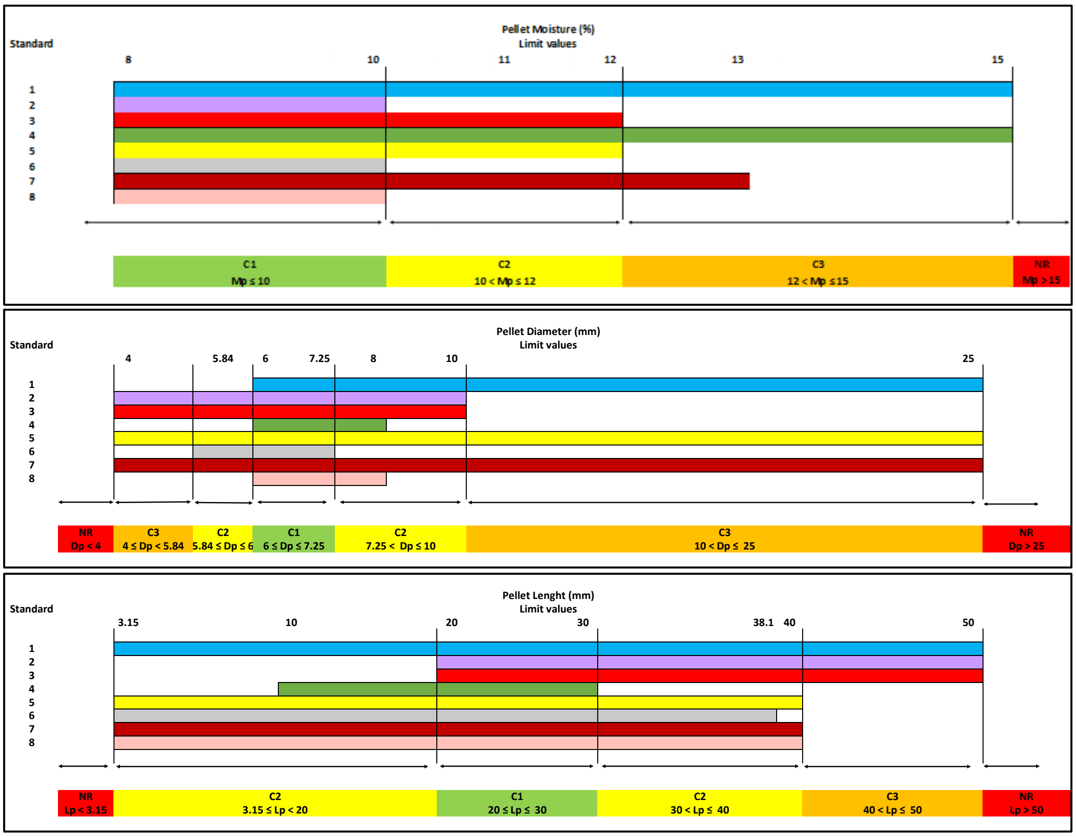
Figure 22. Proposed classification for non-densified Solid Recovered Fuel (SRF) properties

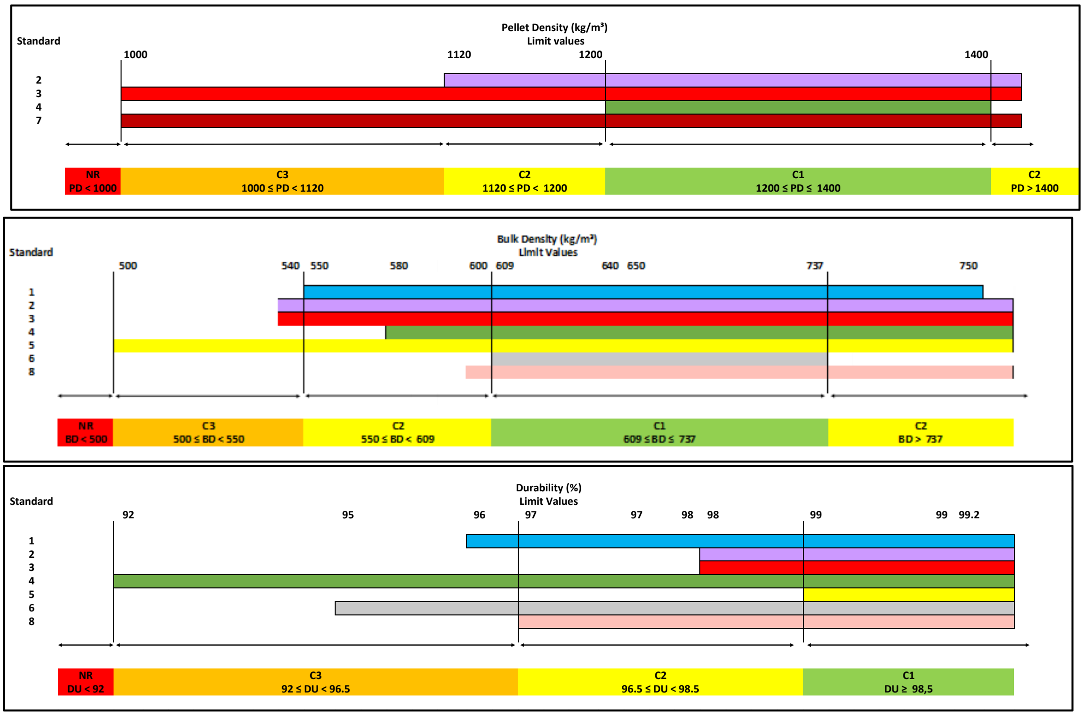
LHV. It corresponds to the classification included in Figure 22

Cl content. It corresponds to the classification included in Figure 22.

Hg content. It corresponds to the classification included in Figure 22.

Ash content. It corresponds to the classification included in Figure 22.





[1]C1: Class 1. C2: Class 2. C3: Class 3. NR: Not recommended

1. ISO 17225:2021. Biocombustibles sólidos. Especificaciones y clases de combustibles.
2. O NORM M7135
3. DIN 51731 and DIN PLUS
4. Agro and Agro+
5. SS187120
6. Pelet Fuel Institute Standards
7. NY/T 1878-2010
8. Japanese Agricultural Standards

Figure 23. Classification proposal for properties of densified Solid Recovered Fuel (SRF) in pellet form

3.2 SRF CHARACTERIZATION

Once the SRF was manufactured, it was characterized by determining the properties and analytical methods shown in Table 17. Results are shown in Table 21 and Table 22, which show the values obtained for each property and its class according to the proposed classification. The results are presented and discussed below.

3.2.1 Characteristics of the non-densified SRF

The values determined for the properties of the non-densified SRF samples and their classification are shown in Table 21. For clarity, a colour code, including green, yellow, orange, and red, was used for classes C1, C2, C3, and NR, respectively. Shredding the residue at the laboratory level required very high drying to obtain an SRF with 4.5% moisture, which cannot be considered a realistic option at an industrial scale. To complete the study, results are included based on moisture levels corresponding to the limits established for this property for each proposed class (20, 25, and 35%). The results obtained are discussed below.

- Lower heating value

Defined as the economic parameter within the requirements for characterization as SRF (Matignon, 2020), this is a standard that measures the total energy content produced as heat when a substance is burned (Etim et al., 2022). The results for SRF produced at a laboratory scale (Table 21) presented a value of 22.93 MJ/kg for a moisture content of 4.5%. This value decreases with increasing the moisture content to 13.37 MJ/kg for 35% water content. The LHV, on a dry basis, was 24.29 MJ/kg, higher than that referenced for SRF generated from waste treatment plant rejects for incineration or co-incineration, with values ranging from 20.06 MJ/kg (Montejo et al., 2011) to 22.13 MJ/kg (Edo-Alcón et al., 2016). The results obtained place the SRF produced in class C1 for three of the four samples, with class C2 corresponding to the sample corresponding to 35% moisture.

- Cl content

From the combustion point of view, low Cl content reduces adverse effects such as corrosion, slagging, and fouling in boilers (Iacovidou et al., 2018; Rotter et al., 2011). In addition, a study on the emission of nanoparticles by conventional and advanced technology noted that the lower presence of Cl suggesting predominately biodegradable salts, but not toxic metals (Panessa-warren et al., 2022). The Cl percentage was determined dryly, so its content does not vary with humidity, reaching a value of 0.031% (Table 21). The values obtained are lower than those referenced in the case of samples generated from urban waste in the studies of Montané et al. (2013), Nasrullah (2015), and Velis (2012) who obtained similar values, specifically 0.65%, 0.60%, and 0.69% respectively. The higher Cl content referenced in the studies above is motivated by the more significant presence of rigid plastics such as PVC (Ma et al., 2008; Rada and Ragazzi, 2014). These plastics are practically non-existent in the SRF produced, with a major presence of sanitary textiles which, even though they include plastics in their composition, are mainly composed of synthetic fibres (Marques et al., 2020). With the value obtained, the Cl content complies with the requirements of class C1 (Table 21).

- Hg content

The Hg content represents the environmental factor of the SRF, measuring the possible toxicity caused by its combustion (Iacovidou et al., 2018). Its characterization is performed on a wet basis, providing contents for laboratory-produced SRF that ranged between 1.0 x 10^{-5} and 5.9 x 10^{-6} mg/MJ (Table 21) for samples SD-35 and SD-4.5, respectively. These values are lower than the 6.9 x 10^{-3} mg/MJ reported by Ranieri et al. (Ranieri et al., 2017b) for SRF produced from municipal waste. If the average Hg content is written concerning the mass of the SRF made, results of 1.3 x 10^{-4} mg/kg are obtained. This is lower than the 9.0 x 10-2 mg/kg found in the literature (Ramos Casado et al., 2016). The Hg content would give the SRF generated from screening class C1 (Table 21) in any of the samples produced.

- Ash content

Determination of the amount of ash quantifies the amount of inert materials present in the SRF, which in this study was 9.4% in all samples (Table 21) since it is determined on a dry basis. If this value is compared with that reported in studies of SRF produced from rejects coming from urban waste, it is observed that they are higher, as in the case of Velis (2012) or Dunnu (2010) whose ash percentage was

17.3% and 15.79%, respectively. The ash content obtained allows classifying this property in the SRF produced, class C1 for all samples (Table 21).

- Moisture

The low moisture content of the SRF allows considerable energy cost savings (Mohammed et al., 2017), as it is directly related to energy value and transportation (Hilber et al., 2007). Due to the need to lower the moisture content as much as possible to facilitate the shredding process, the moisture content of the SRF produced from screening was 4.5% (Table 21), a relatively low value compared to other studies. Studies of SRF made from urban waste have referenced moistures of between 15 (Nasrullah et al., 2015) and 25% (Rada and Ragazzi, 2014). This implies that, on an industrial scale, it could be produced with higher moisture values, thus reducing production costs, taking as a reference the limits considered for this property, i.e., 20, 25, and 35%.

Considering all of the above in the analysis of the different properties that have been included in the characterization of the non-densified SRF, it can be concluded that all the samples produced would comply with the established limits, and none of them would be classified as not recommended. On the other hand, the samples produced with lower moisture values (4.5 and 20%) would have all their properties classified as C1, i.e., they would allow obtaining the fuel with the highest quality. In the case of moisture values of 25%, for the production of SRF, only this property would be affected, and it would be classified in a lower category, C2. Finally, the production of SRF from screening with 35% moisture, classified as C3, would also slightly reduce its quality due to the effect of moisture on its LHV, which would be classified as C2.

Table 21. Values and classification of the properties of the non-densified Solid Recovered Fuel (SRF)

Sample idenfitication	LHV (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	M (%)
ND-4.5	22.93	0.031	5.9 × 10-6	9.4	4.5

Sample idenfitication	LHV (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	M (%)
ND-20	18.15	0.031	7.5 × 10-6	9.4	20
ND-25	16.58	0.031	8.2 × 10-6	9.4	25
ND-35	13.37	0.031	1.0 × 10-5	9.4	35

Class 1 (C1)
Class 2 (C2)
Class 3 (C3)
Not recommended (NR)

3.2.2 Characteristics of the densified SRF

The values and classification of each of the properties analyzed for the 20 pellet samples manufactured, the correlation with the process input variables, and the comparison of the results obtained from other studies are presented below. Average values and standard deviation obtained for the properties determined for samples are shown in Table 22, and their classification according to the colour code is described.

Figure 24 shows the correlation between the input variables (moisture, diameter, and compression length) and the chemical (LHV, Cl, Hg, and ash contents), physical (moisture, diameter, length and density of the pellets, and bulk density), and mechanical (hardness and durability) properties of the pellets produced; the correlation coefficients are shown also. When an increase in one accompanies an increase in the value of another one of the variables, it will be considered a positive or direct correlation, represented in Figure 24 with a range of blue colours. Conversely, if a decrease in one variable accompanies an increase in another, the correlation is negative or inverse, represented in a range of red colours. A correlation coefficient of 1 implies a perfect and positive correlation. On the contrary, the value -1.00 implies an ideal and negative correlation, and finally, the value 0 means that there is no correlation. The diameter of the circles shown in Figure 24 is proportional to the value of the correlation, so a stronger correlation will imply a larger diameter. Finally, Figure 26 represents the LHV, Cl, Hg and ash contents, moisture, diameter, length, pellet density, bulk density, durability, and hardness, concerning the inlet stream moisture, for the different dies used in the pellet production process, in addition to the classification intervals, as well as maximum and minimum values reported in other studies included in the discussion.

Table 22. Average values, standard deviation, and classification of the properties of densified Solid Recovered Fuel (SRF)

Sample	Lower Heating Value (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	Pellet moisture (%)	Pellet diameter (mm)	Pellet length (mm)	Pellet density (kg/m^3)	Bulk density (kg/m^3)	Durability (%)	Hardness (kgf)
P-10-D6L20	24.41 ± 0.39	0.037 ± 0.002	$6.4 \times 10^{-6} \pm 3.4 \times 10^{-6}$	7.73 ± 0.03	9.42 ± 0.43	5.91 ± 0.04	25.57 ± 3.70	1198.03 ± 64.53	504.65 ± 12.70	98.47 ± 0.20	13.00 ± 1.00
P-20- D6L20	21.72 ± 0.21	0.047 ± 0.004	$7.3 \times 10^{-6} \pm 2.9 \times 10^{-6}$	7.78 ± 0.07	18.07 ± 0.24	5.84 ± 0.02	21.87 ± 2.60	1062.91 ± 30.21	419.20 ± 8.30	97.89 ± 0.94	12.67 ± 1.53
P-30- D6L20	18.60 ± 0.10	0.029 ± 0.008	$8.7 \times 10^{-6} \pm 3.7 \times 10^{-6}$	8.02 ± 0.17	27.69 ± 0.18	6.16 ± 0.09	23.39 ± 2.14	810.16 ± 10.27	329.38 ± 4.21	94.39 ± 0.63	8.33 ± 0.58
P-40- D6L20	15.77 ± 0.09	0.033 ± 0.003	$1.0 \times 10^{-5} \pm 3.8 \times 10^{-6}$	8.42 ± 1.03	34.80 ± 0.70	6.02 ± 0.17	20.82 ± 2.71	706.50 ± 62.79	314.87 ± 9.39	93.62 ± 0.96	5.67 ± 1.53
P-10- D6L24	24.54 ± 0.09	0.026 ± 0.003	$6.3 \times 10^{-6} \pm 2.7 \times 10^{-6}$	7.94 ± 0.84	8.81 ± 0.61	6.13 ± 0.10	26.62 ± 3.46	1080.15 ± 25.43	508.10 ± 6.61	98.49 ± 0.34	16.33 ± 2.08
P-20- D6L24	21.47 ± 0.25	0.066 ± 0.003	$7.2 \times 10^{-6} \pm 4.1 \times 10^{-6}$	8.17 ± 0.91	17.93 ± 0.56	6.11 ± 0.12	28.51 ± 2.30	1025.95 ± 60.76	456.55 ± 5.03	97.70 ± 0.72	16.00 ± 1.00
P-30- D6L24	18.90 ± 0.40	0.034 ± 0.010	$8.3 \times 10^{-6} \pm 6.1 \times 10^{-6}$	9.37 ± 0.48	25.64 ± 0.37	6.06 ± 0.09	22.30 ± 3.30	858.14 ± 48.79	408.22 ± 17.65	97.55 ± 0.33	11.33 ± 0.58
P-40- D6L24	15.86 ± 0.67	0.026 ± 0.008	$9.9 \times 10^{-6} \pm 2.5 \times 10^{-6}$	8.63 ± 0.43	33.91 ± 0.96	6.06 ± 0.07	22.37 ± 4.39	753.87 ± 24.45	346.70 ± 17.34	94.95 ± 0.06	9.33 ± 0.58

Sample	Lower Heating Value (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	Pellet moisture (%)	Pellet diameter (mm)	Pellet length (mm)	Pellet density (kg/m^3)	Bulk density (kg/m^3)	Durability (%)	Hardness (kgf)
P-10- D8L16	25.65 ± 0.52	0.041 ± 0.006	$6.2 \times 10^{-6} \pm 3.1 \times 10^{-6}$	9.63 ± 0.44	7.75 ± 0.23	8.33 ± 0.10	36.72 ± 2.11	958.80 ± 37.72	426.28 ± 13.70	99.64 ± 0.22	12.67 ± 1.15
P-20- D8L16	21.84 ± 0.43	0.036 ± 0.004	$7.2 \times 10^{-6} \pm 9.0 \times 10^{-7}$	9.00 ± 0.09	17.66 ± 0.35	8.34 ± 0.05	36.80 ± 2.98	833.55 ± 33.73	380.68 ± 5.10	97.80 ± 0.46	9.67 ± 1.53
P-30- D8L16	18.65 ± 0.18	0.031 ± 0.006	$8.2 \times 10^{-6} \pm 1.5 \times 10^{-6}$	7.73 ± 0.04	25.05 ± 3.17	8.75 ± 0.20	23.95 ± 1.88	691.75 ± 28.43	347.50 ± 10.41	97.72 ± 0.46	7.00 ± 1.00
P-40- D8L16	15.55 ± 0.08	0.033 ± 0.010	$9.4 \times 10^{-6} \pm 1.2 \times 10^{-6}$	7.48 ± 0.64	31.45 ± 0.76	9.04 ± 0.44	29.94 ± 3.69	606.81 ± 71.77	301.07 ± 9.03	88.63 ± 2.06	4.67 ± 0.58
P-10- D8L32	24.80 ± 0.19	0.023 ± 0.003	$6.3 \times 10^{-6} \pm 1.5 \times 10^{-6}$	9.30 ± 0.44	8.55 ± 0.35	8.11 ± 0.09	21.46 ± 0.06	1025.94 ± 20.35	517.53 ± 14.89	97.91 ± 0.89	12.33 ± 1.15
P-20- D8L32	22.01 ± 0.11	0.027 ± 0.001	$7.2 \times 10^{-6} \pm 2.2 \times 10^{-6}$	9.85 ± 1.34	17.18 ± 0.28	8.22 ± 0.08	21.72 ± 1.01	928.46 ± 17.31	457.70 ± 11.64	98.15 ± 0.37	9.67 ± 0.58
P-30- D8L32	18.85 ± 0.33	0.031 ± 0.006	$8.4 \times 10^{-6} \pm 3.8 \times 10^{-6}$	8.02 ± 0.69	26.02 ± 0.20	8.46 ± 0.22	17.79 ± 1.33	650.32 ± 72.77	338.62 ± 2.88	90.08 ± 3.46	5.67 ± 1.53
P-40- D8L32	15.57 ± 0.19	0.028 ± 0.006	$9.8 \times 10^{-6} \pm 4.4 \times 10^{-6}$	8.14 ± 0.22	33.54 ± 0.40	8.66 ± 0.12	19.35 ± 4.33	522.61 ± 35.81	328.17 ± 12.02	62.63 ± 8.81	3.33 ± 0.58
P-10- D8L48	24.87 ± 0.21	0.027 ± 0.005	$6.2 \times 10^{-6} \pm 1.6 \times 10^{-6}$	7.63 ± 0.24	7.91 ± 0.53	8.17 ± 0.29	26.82 ± 4.11	1006.89 ± 72.85	491.37 ± 26.63	99.76 ± 1.42	20.00 ± 1.00

Sample	Lower Heating Value (MJ/kg)	Cl (%)	Hg (mg/MJ)	Ash (%)	Pellet moisture (%)	Pellet diameter (mm)	Pellet length (mm)	Pellet density (kg/m^3)	Bulk density (kg/m^3)	Durability (%)	Hardness (kgf)
P-20- D8L48	21.82 ± 0.22	0.027 ± 0.009	$6.9 \times 10^{-6} \pm 2.0 \times 10^{-6}$	7.54 ± 0.34	14.59 ± 0.17	7.76 ± 0.27	24.50 ± 4.14	911.16 ± 33.82	455.93 ± 14.39	99.22 ± 0.38	14.33 ± 1.53
P-30- D8L48	18.69 ± 0.36	0.029 ± 0.004	$8.1 \times 10^{-6} \pm 4.2 \times 10^{-6}$	7.45 ± 0.46	23.99 ± 0.67	8.14 ± 0.27	24.89 ± 3.11	753.91 ± 52.05	403.65 ± 15.42	98.35 ± 0.21	9.67 ± 1.15
P-40- D8L48	15.47 ± 0.14	0.041 ± 0.011	$9.6 \times 10^{-6} \pm 2.7 \times 10^{-6}$	7.22 ± 0.51	32.36 ± 0.17	8.20 ± 0.15	22.33 ± 2.04	696.12 ± 36.96	366.85 ± 22.52	85.88 ± 4.14	4.33 ± 0.58

Class 1 (C1)

Class 2 (C2)

Class 3 (C3)

Not recommended (NR)

Properties not subject to classification

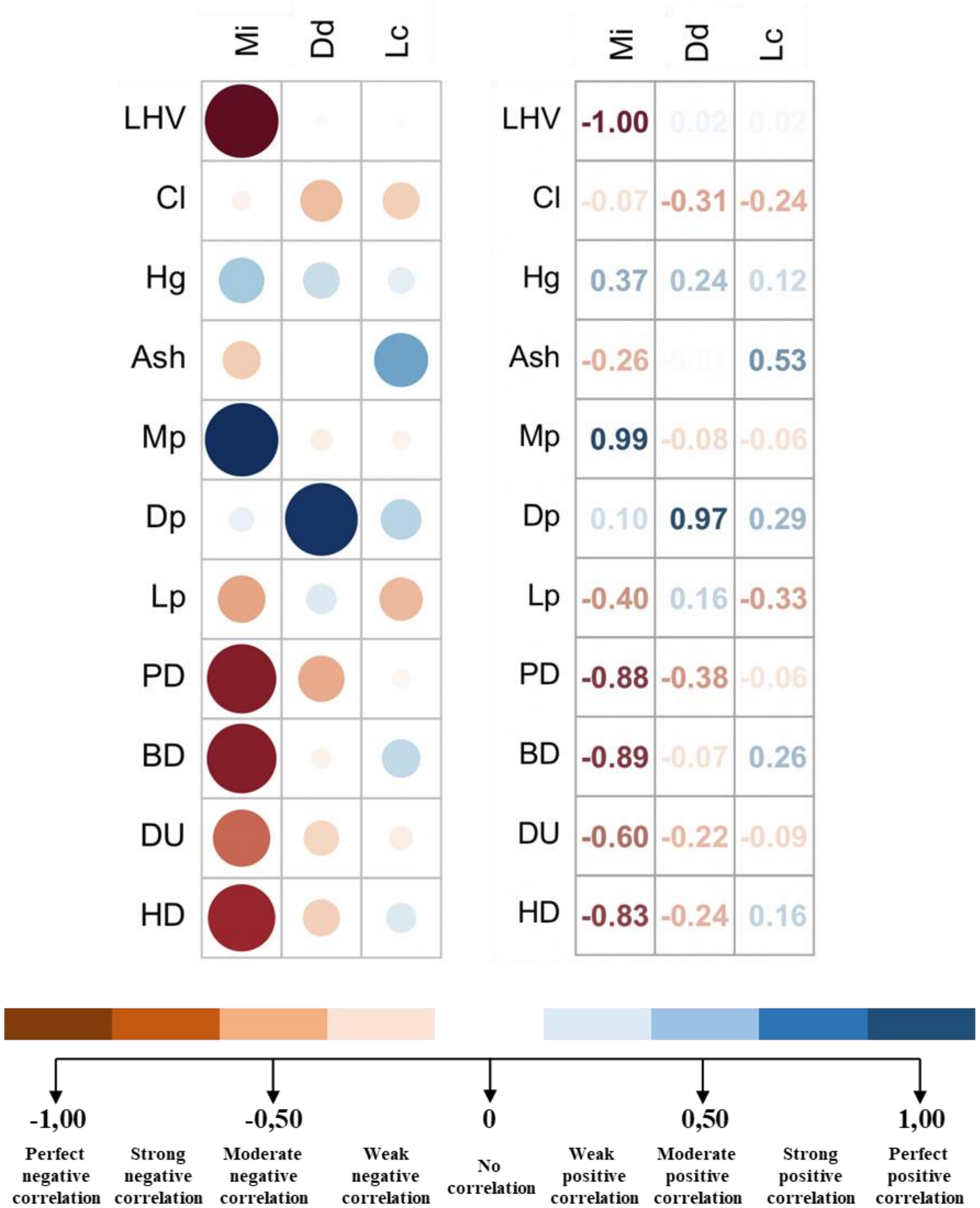


Figure 24. Graphical representation and values of the correlation between production variables and the properties1 of the pellets produced.

- Lower heating value

Table 22 and Figure 26 show the LHV values determined for the densified SRF samples, which ranged from 15.47 to 25.65 MJ/kg for the pellets identified as P-40-D8L48 and P-10-D8L16, respectively, with the value of this property decreasing with that of the input stream moisture. As indicated in Table 22, Figure 24 shows a perfect inverse correlation between LHV and inlet stream moisture, with a regression coefficient of -1.00; however, it is observed that there is practically no correlation between this property with the characteristics of the pellet dies, with correlation coefficient of 0.02 for diameter and length.

Comparing the values obtained with those of other studies (Figure 26), it is observed that for inlet current humidity values of 10%, the LHV obtained varied between 24.41 and 25.65 MJ/kg in the case of samples P-10-D6L20 and P-10-D8L16, respectively. These values are higher than those obtained for MSW pellets in the studies of Nursani (2020) and Ramos Casado et al. (2016) who recorded 18.24 and 20.34 MJ/kg, respectively, or that of Suryawan et al. (2022) in the case of pellets produced from paper, garden and food waste, where the maximum value referenced was 20.41 MJ/kg. The results obtained for inlet humidity values of 20% ranged from 21.47 to 22.01 MJ/kg for samples P-20-D6L24 and P-20-D8L32; these values are still slightly higher than those previously cited (Nursani et al., 2020; Ramos Casado et al., 2016) and also the minimum value of 17.22 MJ/kg recorded by Suryawan et al. (2022) in the case of pellets produced from paper, garden, and food waste. Finally, for inlet humidity values of 30%, the LHV results were similar to the lowest values of the mentioned studies, ranging between 18.60 and 18.90 MJ/kg for samples P-30-D6L20 and P-30-D6L24, respectively; the lowest values were reached for the highest inlet stream humidity values (40%), all of them were below 16 MJ/kg and up to 11% lower than the minimum value referenced by Suryawan et al. (2022). The LHV results allowed classifying this property in class C1, with no pellet samples of class C2, C3, or not recommended (Figure 26). Only the five samples corresponding to an inlet current humidity of 40% are close to the limit established by class C2 (15 MJ/kg), ranging between 15.47 and 15.86 MJ/kg.

- Cl content

As indicated in the non-densified SRF, low Cl content reduces adverse effects such as corrosion, slagging, and fouling in the boilers (Iacovidou et al., 2018; Rotter et al., 2011). Since Cl determination is done on a dry weight basis, densification does not influence Cl content, as evidenced by a correlation coefficient of -0.07 for moisture. (Figure 24) and values that varied between 0.023 and 0.066% for P-10-D8L32 and P-20- D6L24 respectively (Table 22) An average of 0.034 % was obtained for the twenty samples, similar to the results obtained in the case of the SRF without densification.

Comparing these results with other pellet studies, the percentage obtained is similar to those produced with olive wood, which was 0.03% (Garcia-Maraver et al., 2015). This value is lower than that referenced by Ramos Casado (2016) and Garcia et al. (2021) for SRF produced from household waste, with values of 0.76% in Cl, but higher than the 0.016% determined for those produced from a mixture of sewage sludge and herbaceous biomass (Kliopova and Makarskiene, 2015) or 0.01% for those made from olive leaf (Garcia-Maraver et al., 2015). As a result, this property would be classified in all cases with the highest quality, i.e., with category C1 (Figure 26).

- Hg content

As in the non-densified SRF, the Hg content must be considered because of its toxicity in the combustion process (Iacovidou et al., 2018). Table 22 and Figure 26 show the Hg content values determined for the densified SRF samples, ranging from a minimum of 6.2×10^{-6} to a maximum of 1.0×10^{-5} mg/MJ for samples P-10-D8L16 and P-40-D6L20, respectively. Figure 24 shows a positive correlation of Hg content for the three input variables. Although, as shown in Figure 24, Pearson's Coefficient values (0.37, 0.24, and 0.24 for Mi, Dd, and Lc, respectively) are rated as weak in all three cases; they are higher in the case of moisture.

The values obtained have results lower than those referenced in other studies in which the Hg content reached values of 0.005 mg/MJ (Ramos Casado et al., 2016) and 0.042 mg/MJ (Kliopova and Makarskiene, 2015) in pellets produced from the rejection of biological treatment of waste and sewage sludge, respectively. This result has allowed classifying all the pellet samples within the limits established for class C1 for this property (Figure 26).

- Ash content

Since the ash content expresses its results on a dry basis, densification does not influence the results of this property, whose values varied between 7.22 and 9.85% for samples P-40-D8L48 and P-20-D8L32 (Table 11), similar to those obtained in the case of the non-densified SRF (Table 8). Figure 24 shows that ash content exhibits a weak negative correlation (-0.26) with initial moisture, being null for its relationship with diameter. In contrast, a Pearson's coefficient of 0.53 concerning input length exhibits a moderate positive correlation.

This result is below the 21% referenced in other studies for pellets manufactured from municipal waste treatment plant reject (Ramos Casado et al., 2016), sludge, and biomass waste (Kliopova and Makarskiene, 2015), or 30.5% for SRF from municipal waste (Santamaría et al., 2021). On the contrary, the values obtained are higher than those referenced for pellets produced only from wood or herbaceous biomass which varied between 0.4% for pellets from pine wood (García et al., 2021) and 1.43% for olive wood (García-Maraver et al., 2015). However, the ash content reported by Said et al. (2015) for pellets produced from rice straw was higher, with values between 13.25 and 18.66%. A value of 6.49% was obtained from wheat straw, slightly lower than the SRF produced (Carroll and Finnan, 2012). In any case, the values obtained allow classifying this property for all pellet samples produced as class C1 (Figure 26).

- Moisture

The water content of the produced SRF was evaluated within the framework of quality standards regarding pellet conditioning since safe storage must be ensured to avoid bacterial growth (Lehtikangas, 2001) and degradation of the produced material. Table 22 and Figure 26 show the moisture values determined for the densified SRF samples, which ranged from 7.75 to 34.80% for samples P-10-D8L16 and P-40-D6L20, respectively. This demonstrated a reduction of 15.27 ± 5.32% concerning the moisture content of the incoming residue, with pelletizing temperatures below 29 °C. Figure 24 shows a perfect direct correlation between the moisture of the pellet produced and the moisture of the incoming stream, with a correlation coefficient of 1 (Figure 24). However, in the case of the variables related to the characteristics of the pelletizing dies, the correlation coefficients obtained were -0.08 and -0.06 for the diameter and compression length, respectively, which shows that there is practically no correlation between them and the moisture of the pellets produced.

Studies on pellets made from agricultural residues have been consulted to analyse the values obtained. In them, values ranging from 18.2 (García-Maraver et al., 2011) to 27.17% were achieved (Nursani et al., 2020). In a work by Wang (2018), who produced pellets from MSW, moisture values between 10.3 and 18.9% were obtained. It was observed that for samples produced with an input stream moisture content of 10%, the resulting moisture was always lower than the referenced values. For samples produced with input stream moisture of 20 and 30%, all values, except for sample P-30-D6L20 with an output moisture content of 27.69%, were in the range of the referenced studies. Specifically, in the case of an input moisture of 20%, all pellet samples obtained a moisture value within the limits referenced by Wang (2018) for densified MSW. Finally, pellet production with input stream moistures of 40% resulted in pellets with moistures above 31%, these values are well above the referenced studies.

The moisture results obtained allowed us to classify this property in the different established classes (Table 22), showing that inlet stream moisture values higher than 10% resulted in pellets with moisture levels that were considered not recommendable. In fact, 14 of the 20 samples manufactured reached moisture values higher than 15%. Five of the samples produced were classified as class C1 concerning this property, all corresponding to samples pelletized with 10% moisture. In addition, none of the samples were classified as class C2, and only one of them, pelletized with an inlet moisture content of 20% with the M5 die, was ranked as class C3.

- Pellet size: diameter and length

Pellet size is a relevant factor in the use phase because combustion is more uniform with smaller diameter pellets, and a high length can hinder the continuous feeding of the plant (Lehtikangas, 2001) and block the hoppers (Grootjes et al., 2015). At the same time, a long pellet is easier to break than a shorter one (Said et al., 2015; Tarasov et al., 2013), affecting the storage and transport phases. In the case of the results obtained in the pellet diameter, the eight samples produced with the 6 mm diameter die present values between 5.84 (P-20-D6L20) and 6.16 mm (P-30-D6L20). On the other hand, the remaining twelve samples, manufactured with 8 mm dies, resulted in pellets with diameters between 7.76 (P-20-D8L48) and 9.04 mm (P-40-D8L16). There is an almost perfect positive correlation between the die inlet diameter and the pellet outlet diameter (Figure 24), which reaches a correlation coefficient of 0.97 (Figure 24). However, it is practically not affected by the die compression length and the moisture content of the inlet stream, with coefficients of 0.29 and 0.10, respectively. Regarding the most commonly used diameter in other studies in which pellets were manufactured from urban waste, the use of 6 mm diameter dies predominates, although some studies with larger diameters of 16, 18, and 20 mm have also been referenced (Jewiarz et al., 2020).

The values obtained in the case of length have shown large variability, with a minimum of 17.79 mm (P-30-D8L32) and a maximum of 36.80 mm (P-20-D8L16), as shown in Table 22. Also, there was a weak positive correlation with die diameter, a weak negative correlation with compression length, with a correlation coefficient of -0.33, and close to moderately negative, with a coefficient of -0.4, in the case of input moisture (Figure 24). The effect of inlet moisture is higher, as shown in Table 22, in the case of pellets manufactured with a diameter of 8 mm. Although, in some cases, they are higher, the length values obtained could be considered similar to those of other studies, with values that have varied between 20 (Wang et al., 2018) and 24 mm (Nursani et al., 2020).

The results obtained for the properties that define the size of the pellets made it possible to classify them into several of the established classes. By diameter (Table 22), six of the samples would be classified as class C1, all of them produced by 6 mm diameter dies, while the remaining 14, corresponding to pellets manufactured with 8 mm diameter dies, would be included in class C2. Regarding the length (Table 22), 16 of the 20 samples correspond to Class C1, while the remaining four fall into Class C2. With the M1, M2 and M5 dies, pellets classified as Class C1 were produced. In the case of the M3 and M4 dies, Class C2 pellets were produced with 10 and 20% and 30 and 40% moisture content, respectively.

- Pellet density

Pellet density is a fundamental parameter because low-density pellets are more easily broken and decomposed (García-Maraver et al., 2015; Lehtikangas, 2001). According to the literature reviewed, the use of high-density biofuels generally improves combustion (Jewiarz et al., 2020), gasification (Nixon et al., 2013), and pyrolysis processes (Chen et al., 2015), although there are studies that argue that a very high pellet density could generate combustion problems (Tarasov et al., 2013).

Table 22 and Figure 26 show the density values of the manufactured pellets, which varied between 522.61 and 1198.03 kg/m^3 for samples P-40-M4 and P-10-M1, observing the effect that moisture has on the production process. In fact, Figure 24 shows a high inverse correlation between pellet density and moisture of the input stream, with a correlation coefficient of -0.88 (Figure 24). Also, in the case of the variables related to pellet die characteristics, an inverse correlation is observed in both cases, but it is very weak in the case of compression length and weak for die diameter, with correlation coefficients of -0.06 and -0.38, respectively (Figure 24).

Comparing these results with those referenced in other studies, it becomes clear that they are comparable to those of other studies in the case of pellets produced with low feed stream humidity values (10 and 20%). Thus, Ramos Casado produced pellets from the rejects of mechanical biological treatment plants of municipal waste, reaching a density of 1050 kg/m^3 (Ramos Casado et al., 2016). After torrefaction of such waste, Ma et al. (Ma et al., 2022) produced pellets with densities that varied between 994.78 and 1208.86 kg/m^3, depending on the torrefaction and pelletization temperature. Meanwhile, mixtures of different waste fractions present in MSW were pelletized, reaching values that varied between 1040 and 1199.5 kg/m^3. The minimum value corresponded to the composition with a lower percentage of paper (Rezaei et al., 2020). Finally, studies of densification of sewage sludge mixed with biomass allowed obtaining pellets with densities ranging from 851.2 to 1270.3 kg/m^3 (Jiang et al., 2014). With increasing inlet stream moisture up to 30 and 40%, pellet density values were reduced to values below the minimum value referenced by Jiang et al. for sewage sludge with biomass (Jiang et al., 2014).

The pellet density results obtained allowed classifying this property in the different established classes (Figure 26), with a predominance (fourteen of the twenty samples analyzed, 75%) of those being classified as not recommended. Furthermore, none of the samples produced was classified as class C1, only one as C2 (pelleted at 10% moisture), and five as C3 (pelleted at 10 and 20% moisture). The low-density values are explained by the use of reference values to establish the classification based on standards applicable to agricultural waste, which usually report higher values such as 1327 kg/m^3 for olive wood (Garcia-Maraver et al., 2015), 1260 kg/m^3 for rice straw (Said et al., 2015), or 1198 kg/m^3 for pellets produced from alfalfa (Sarker et al., 2015). However, the composition of the initial screening is characterized by the high presence of low-density fractions (52.10% of sanitary textiles and 11.70% paper and cardboard). These values could be increased by adding a binder, as reported in the study by Nursani et al. (2020) which produced pellets with a density between 988 - 1009 kg/ m^3 from urban waste.

- Bulk density

Some of the problems derived from the low bulk density of SRF is the need for high storage volumes, increased transportation costs, as well as difficulties in feeding (Lomas Esteban et al., 2001), hence the importance of the analysis of this property, which in this study reached values between 301.07 and 517.53 kg/m^3 for samples P-40-D8L16 and P-10-D8L32 (Table 22), respectively. The effect of inlet stream moisture was observed (Figure 26). This translates into a strong inverse correlation between bulk density and inlet stream moisture, with a correlation coefficient of -0.89 (Figure 24). In the case of compression length and die diameter, the observed correlations were weakly positive and very weakly negative, with correlation coefficient values of 0.26 and -0.07, respectively (Figure 24).

Comparing these values with those obtained in other MSW pelletizing studies, it becomes clear that they are comparable to those referenced in other studies of those produced with low feed stream moistures (10 and 20%). MSW pelletizing studies show bulk densities that varied between 383.9 (Nursani et al., 2020) and 540 kg/m^3 (Ramos Casado et al., 2016). In another study, Rivera (2018) reported values between 420 and 510 kg/m^3.

The results for this property, shown in Table 22, allow the samples to be classified into the established classes. None of the samples produced was classified as class C1 and C2 for this property. Three of them, corresponding to an inlet stream moisture content of 10%, were classified as C3, while the remaining 17 (85% of the samples produced) reached bulk densities below 500 kg/m^3, and therefore, were classified as not recommended. Again, the low-density values are explained by the presence of low-density fractions in the waste and the use of reference values to establish the classification based on standards applicable to agricultural waste, which usually report higher values. In fact, in the case of pellets produced from various biomasses, according to a review, the density values were found to be higher than 600 kg/m^3 (Miranda et al., 2015). These values were reported for pellets from olive pomace with 780 kg/m^3 (Miranda et al., 2012) and oak and scots pine wood with 678 (Miranda et al., 2009) and 675 kg/m^3 (Filbakk et al., 2011), respectively, or wheat straw with a bulk density of 620 kg/m^3 (Verma et al., 2012). The values for bulk density could be increased by adding a binder, as reported in some studies certifying that binders strengthen the cohesion between particles and increase the density, both particle and bulk (Ju et al., 2020; Zdanowicz and Chojnacki, 2017).

- Durability

Durability is an essential parameter concerning transportation and logistics (Jewiarz et al., 2020). It can be considered a reference property for SRF pellet conditioning (Said et al., 2015). High durability is synonymous with high-quality (Zafari and Kianmehr, 2014) as it avoids the generation of fine particles that could increase pollutant emissions and even health risks (Miranda et al., 2015). Table 22 and Figure 26 show the durability values of the manufactured pellets, which ranged from 62.63% to 99.76%, for samples P-40-D8L32 and P-10-48, respectively. Figure 24 shows a strong inverse correlation between durability and inlet stream moisture, with a coefficient of -0.60. The effect of die diameter and compression length is not significant, with observed correlation coefficient values of -0.22 and -0.09, respectively (Figure 24), also implying an inverse correlation but in this case weak and very weak.

The analyses of durability values obtained in other studies of densified SRF production from MSW show similar values, such as those obtained from MSW rejects with durability of 96.8% (Ramos Casado et al., 2016). In a work where water content was analyzed as a variable, 93.10 and 98.72% durabilities were obtained with input stream moisture values of 30 and 15%, respectively (Rezaei et al., 2020). Considering pelletizing temperature as a variable, the maximum durability (96%) was reached at 120 °C (Jewiarz et al., 2020). Higher durability values have been found in the case of torrefied biodegradable products from MSW, with 99.67% (Ma et al., 2022) and up to 99% improving the pellet by adding 6% binder (Nursani et al., 2020). The pellets produced from rubber wood and waste derivative mixtures presented high durability levels (98.27 to 99.07%) (Laosena et al., 2022).

The durability results concerning the input variables (Figure 26) allow placing the samples in the established classes, predominantly class C2, which includes 50% of them, followed by C3 with four samples, and C1 with 3. Finally, only 3 of the samples produced were considered not recommended, with values lower than 92%, all of these corresponded to inlet current humidity values of 40%.

- Hardness

As mentioned in the methodology, the importance of this property lies in handling and storage, as well as in the combustion process itself, where adequate hardness is required to avoid crushing and deforming the pellets (García-Maraver et al., 2011; Gilbert et al., 2009; Said et al., 2015), which causes difficulties in the boiler operation due to occasional blocking of the screw conveyor, regardless of the thermal boiler load (García-Maraver et al., 2014).

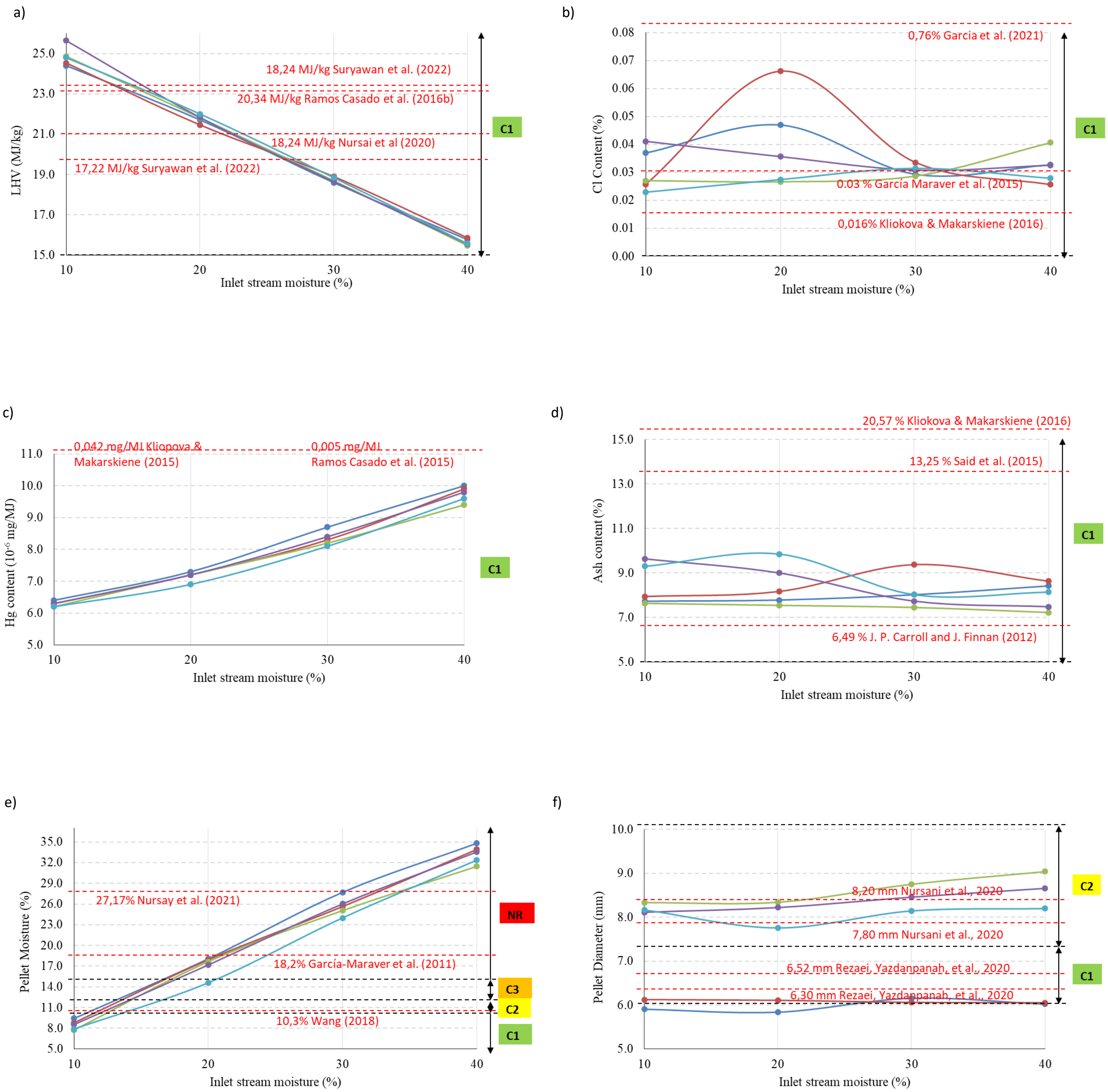
The results obtained for this property in the SRF screening cover a broad spectrum of values, with a minimum of 3.33 kgf and a maximum of 20 kgf for samples P-40-D8L32 and P-10-D8L48, respectively (Table 22). Regarding the relationship of this property with the input variables, a strong inverse correlation was observed between hardness and input moisture, with a correlation coefficient of -0.3, which is also evident in Figure 24. The correlation was also harmful in the case of diameter, although relatively weak (-0.24), while for compression length, it is also soft but positive (0.16) (Figure 24).

If the results are compared with other studies, the maximum values are similar. However, the minimum value is much lower. Thus, in a study developed to improve the properties of pellets from municipal waste by hydrothermal treatment, hardness values between 7.37 and 13.34 kgf were obtained (Phasee and Areeprasert, 2018). In the case of the pellets produced by Rezaei (Rezaei et al., 2020), the hardness varied between 11.11 and 17.13 kgf for water contents of 15% and 30% respectively. The addition of binder in the pellet manufacturing process increased the hardness up to 17.68-21.37 kgf (Nursani et al., 2020). In any case, it was observed that for moisture below 30%, SRF hardness values can be similar to those found in the literature.

No classes have been established in this case because the property is not contemplated in the reference standards. However, the values obtained are below those recommended in studies for biomass pellets such as wood (Arshadi et al., 2008) and herbaceous or agricultural residues (Carroll and Finnan, 2012; Zamorano et al., 2011), whose optimum hardness, according to the literature, would be 22 kgf (Arshadi et al., 2008; Carroll and Finnan, 2012; Said et al., 2015; Zamorano et al., 2011).

Taking into account the classifications of the properties considered to establish the quality of the densified SRF in pellet form, shown in Table 22 with the colour code, it is observed that only 3 of the 20 samples produced comply with the limits established for all of the properties, specifically samples P-10-D6L20, P-10-D6L24, and P-10-D8L32. In the rest of the samples, some of the properties were not recommendable, so the quality of the pellets would not be suitable according to the classification proposal. On the other hand, for all the samples, it was observed that the specific properties conducive to evaluating the quality of SRF as fuel reach classes C1 and C2. Moisture, pellet density, and bulk density are the ones that reach values with a lower rate (C3), not even recommendable in most of the density determinations. This result is explained by the use of standards for pellets produced from agricultural residues, with higher density than screening waste, which has a lower density due to its high content of sanitary textiles. For this reason, it is considered that the proposal for quality standards in the future should consider this aspect, as well as the incorporation of hardness, which is not incorporated in the current standards for other types of waste since it is viewed as a property to be included in future pellet quality standards.

On the other hand, concerning the operating variables of the pelletizing process for this residue, the most favourable results were obtained with 10% moisture in the inlet stream, which would imply the need to subject the screening waste, characterized by high moisture, to an intense drying process, which would mean higher production costs. Regarding the dies, the most suitable option would be the 6 mm inlet diameter. With the two compression lengths tested (20 and 24 mm), this 6 mm die makes it possible to obtain convenient pellets (classes C1, C2, or C3). It is also possible to produce pellets with a larger diameter, 8 mm, with a compression ratio of 8/32.


a)
LHV (MJ/kg)
Inlet stream moisture (%)
18,24 MJ/kg Suryawan et al. (2022)
20,34 MJ/kg Ramos Casado et al. (2016b)
18,24 MJ/kg Nursai et al (2020)
17,22 MJ/kg Suryawan et al. (2022)
C1
b)
Cl Content (%)
Inlet stream moisture (%)
0,76% Garcia et al. (2021)
0.03 % García Maraver et al. (2015)
0,016% Kliokova & Makarskiene (2016)
C1
c)
Hg content (10^-6 mg/MJ)
Inlet stream moisture (%)
0,042 mg/MJ Kliopova & Makarskiene (2015)
0,005 mg/MJ Ramos Casado et al. (2015)
C1
d)
Ash content (%)
Inlet stream moisture (%)
20,57 % Kliokova & Makarskiene (2016)
13,25 % Said et al. (2015)
6,49 % J. P. Carroll and J. Finnan (2012)
C1
e)
Pellet Moisture (%)
Inlet stream moisture (%)
27,17% Nursay et al. (2021)
18,2% García-Maraver et al. (2011)
10,3% Wang (2018)
NR
C3
C2
C1
f)
Pellet Diameter (mm)
Inlet stream moisture (%)
8,20 mm Nursani et al., 2020
7,80 mm Nursani et al., 2020
6,52 mm Rezaei, Yazdanpanah, et al., 2020
6,30 mm Rezaei, Yazdanpanah, et al., 2020
C2
C1

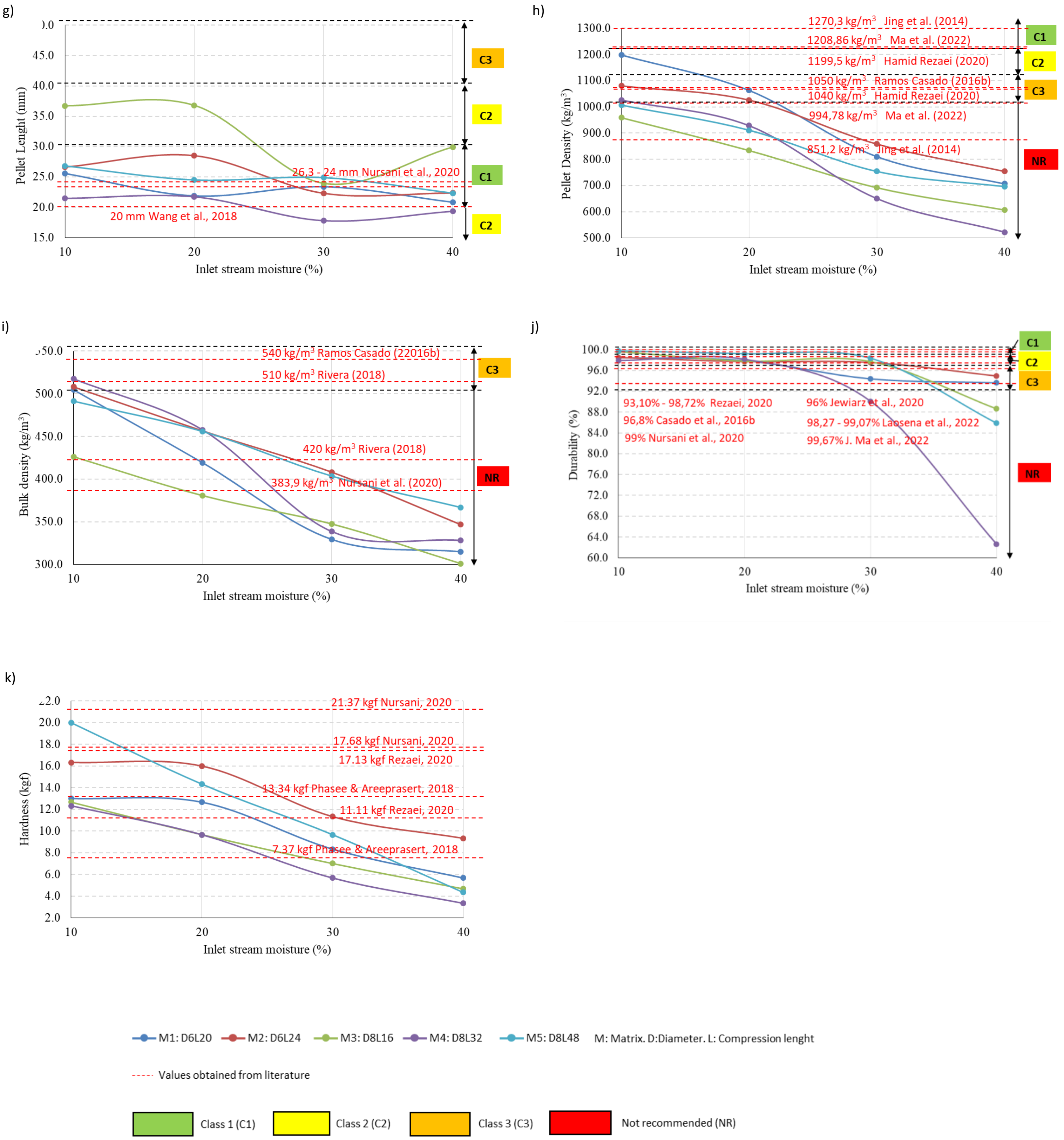


Figure 25. Characteristics of densified solid recovered fuel (SRF) in relation to inlet stream moistures (10, 20, 30 and 40%) and for the different matrix used in the pellet production process (M1, M2, M3, M4 and M5). a) Lower Heating Value (LHV). b) Cl content. c) Hg content. d) Ash content. e) Pellet moisture. f) Pellet diameter. g) Pellet length. h) Pellet density. i) Bulk density. j) Durability. k) Hardness.

3.3 Determination of SRF uses

The results of the properties produced from screening both densified and non-densified SRF were compared with the reference ranges shown in Table 20 for its uses in cement plants, power plants, and gasification, resulting in the degree of compliance by properties shown overall in Table 23.

Firstly, it can be seen that the Cl, Hg, and ash content of the SRF manufactured was limiting for any of the uses analyzed, while LHV limits its application in cement works and LHV and moisture for gasification.

On the other hand, it was observed that the application of manufactured SRF in plants to produce energy from waste does not pose any limitation. In the case of cement plants, the possibility of application is high, since three of the four and fifteen of the twenty samples of non-densified and densified SRF, respectively, comply with all the established limits, which represents 75% of the same. In both cases, the limiting property was the LHV, which shows the difficulty of using SRF produced at humidity values above 35%. Finally, gasification turned out to be the application with the lowest number of samples suitable for use, with only one of the five applicable non-densified SRF samples and six of the twenty in the case of densified, representing 25% and 30%, respectively. In this case, the applicability is limited to SRF produced at humidity values below 20%. The moisture content of the manufactured SRF was the most limiting property since 17 samples did not meet the requirements. These corresponded to the samples of non-densified SRF manufactured with a moisture content of 35% and to all the samples of densified SRF produced with moisture content equal to or higher than 20%, except in the case of one that used a longer compression length (P-20-D8L48), which failed with moisture content values equal to or higher than 30%. In the case of LHV, given its relationship with humidity, six samples were added to the non-compliance list, one in non-densified SRF and five in densified SRF. These cases also corresponded with samples of SRF manufactured with humidity values equal to or higher than 35%.

Table 23. Comparative properties of Solid Recovered Fuel (SRF) produced for use in cement, waste to energy, and gasification plants. ✓. Meet requirements. ⊗. Does not meet requirements

SRF type	**Samples**	**Uses**																	
		Cement plants						Energy from waste plants						Gasification					
		LHV	Cl	Hg	Ash	M	Suitable	LHV	Cl	Hg	Ash	M	Suitable	LHV	Cl	Hg	Ash	M	Suitable
Non densified SRF	ND-4.5	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	ND-20	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	ND-25	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	ND-35	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗
Densified SRF	P-10-D6L20	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-20- D6L20	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-30- D6L20	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-40- D6L20	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗
	P-10- D6L24	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-20- D6L24	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-30- D6L24	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-40- D6L24	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗
	P-10- D8L16	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-20- D8L16	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-30- D8L16	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-40- D8L16	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗
	P-10- D8L32	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-20- D8L32	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-30- D8L32	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-40- D8L32	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗
	P-10- D8L48	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-20- D8L48	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	P-30- D8L48	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	⊗	⊗
	P-40- D8L48	⊗	✓	✓	✓	✓	⊗	✓	✓	✓	✓	✓	✓	⊗	✓	✓	✓	⊗	⊗

4 CONCLUSIONS

Energy recovery from the screening waste would be a definitive step towards achieving the zero-waste objective in wastewater treatment, avoiding the economic and environmental costs derived from landfill disposal. Conclusions obtained have been grouped into 3 categories.

In relation to the proposed classification of the SRF produced, based on existing regulations on the quality of SRF as a fuel, and on those relating to the quality of the pellet according to its mechanical properties.

- The proposed classification of the properties that affect the quality of non-densified SRF is a reference framework to be considered for future quality standards since it brings together the diversity of existing standards and can be used as a common framework.
- Due to its benefits in handling and use, the densification of SRF from sources such as MSW or screening waste requires the development of quality standards applicable to pellets, depending on its uses, which are currently non-existent. In this sense, the proposed classification of the properties that affect the quality of densified SRF is a frame of reference to be taken into account for future quality standards that can be based on the current standards for agricultural and forestry waste or similar waste. However, it will be necessary to uniquely analyse values that limit properties linked to the characteristics of these wastes derived from their composition and the incorporation of hardness values due to their effects on the handling and use of the pellets.

Concerning the production of SRF at laboratory scale required some conditions, mainly for the experimental design of densified SRF. The input moisture content for densification varied between 10 and 40%, and the compression ratios of the matrices used were 6/20, 6/24, 8/16, 8/32 and 8/48.

- The production of SRF, both densified and non-densified, is a viable option for screening waste that meets the requirements of the European standard ISO 21640:2021.
- In the case of the production of the non-densified SRF, taking into account the classification proposed for the properties selected to determine its quality, it would be desirable to produce it from screening waste with a maximum of 20% moisture; it is possible to do so up to moisture content of 35%, even if there is a loss in its LHV.
- In the case of the production of densified SRF in the form of pellets, taking into account the classification proposed for the properties selected to determine its quality, it would be desirable to produce it with a residual moisture content of 10%, using a die with a 6 mm inlet diameter with compression lengths of 20 or 24 mm, or a larger diameter, 8 mm, with a compression ratio of 8/32.
- The moisture content of the residue used for the production of SRF is the variable that will condition the process the most since it is necessary to reduce it to values of 35% in the case of non-densified and 10% for the manufacture of pellets, which could affect the economic viability of the product.

In relation with the potential uses proposed by the standard ISO/TR 21916:2021 for applying the SRF produced:

- The Cl, Hg, and ash content of the SRF manufactured did not limit any of the uses analyzed, while the LHV limits its application in cement works, and LHV and moisture were limiting in the case of gasification.
- The use of the SRF produced is not limited in the case of power plants. Cement plants would require production processes with humidity values below 35%, both for non-densified and densified SRF. The major limitation for SRF use is observed in its application for gasification. Non-densified SRF

could be used when manufactured with humidity values lower than 35%, reducing this limit to values lower than 20% in the case of densified SRF for samples densified with a high compression ratio die (8/48).

CHAPTER 4

ANALYZING THE ECONOMICAL FEASIBILITY OF THE PRODUCTION OF SOLID RECOVERED FUEL

1 INTRODUCTION

During last years, wastewater management has been developing toward sustainability and a CE through energy self-sufficiency and zero waste (Gherghel et al., 2019). As discussed in Chapter 1, screening waste from the WWTP pre-treatment generally has no energy recovery within the CE guidelines (Boni et al., 2021) and is mainly disposed of in landfills, generating economic and environmental problems (Tsiakiri et al., 2021).

As an alternative to avoid landfill disposal, an analysis of screening waste from Biofactoría Sur in Granada concluded in the Chapter 2 of this thesis report that the properties of the waste were suitable for transformation into SRF. In addition, Chapter 3 of this report exhibited the technical feasibility of producing densified and non-densified SRF.

Although the production of SRF does not follow a specific preparation technology (Di Lonardo et al., 2016), the process generally includes the stages of shredding, removal of unsuitable fractions (e.g., metals or inerts), drying and conditioning of the product (Kaartinen et al., 2013). Drying is relevant for the screening waste treatment as moisture levels of up to 77.3% (Chapter 2) would have to be lowered to achieve optimal values concerning the calorific energy of the fuel obtained. Solar drying is generally carried out in a greenhouse containing a scarification roller and an air movement system and can be applied to dry sludge from WWTPs (Đurđević et al., 2019). Another, more established, alternative is thermal drying (Juchelková, 2019). The shredding of the dry screening is a complicated task due to the high percentage of sanitary textiles and their resistance to grinding (Le Hyaric et al., 2009). On a technical level, SRF densification improves boiler feed for combustion (Gilbert et al., 2009), and processes such as gasification (Nixon et al., 2013) are more suited for densified fuel. From an economic point of view, and motivated by the decrease in volume, the transport phase is a much more efficient process (Jewiarz et al., 2020).

Solutions for screening waste management must be environmentally viable and acceptable in social and economic terms (Fernández González et al., 2017). At this point, the techno-economic analysis of SRF production should be a focus of research. Most studies in this area analyse the economic feasibility of using SRF as a substitute for fossil fuels; however, the SRF production chain has yet to be studied in economic terms. The co-firing of SRF with biomass and coal was subjected to a cost impact study in cement plants (Iacovidou et al., 2018). In Metro Vancouver (Canada), using a cost/benefit analysis, four scenarios involving the use of a fuel produced from MSW were compared (Reza et al., 2013). As an alternative to the use of SRF, gasification was analyzed in terms of economic viability, considering the initial and operating costs of a plant with a capacity of 5000 tons/year (Arena et al., 2015).

Based on the results obtained in the Chapter 3, SRF was produced without densification and densified from this waste; however the economic implications associated with scaling-up this process were not studied. The main objective of this study was to carry out an economic feasibility study of implementing SRF production, including drying, shredding and densification stages, from screening waste from a WWTP. The financial tool used was the Net Present Value (NPV), which was combined with Monte Carlo (MC) analysis, obtaining results that will facilitate decision making.

2 MATERIALS AND METHODS

The feasibility analysis was carried out based on four SRF production scenarios. This section presents the economic and environmental evaluation methods. In addition, MC simulation was proposed as a risk analysis.

2.1 DESCRIPTION OF SCENARIOS

As an alternative to landfill disposal of screening waste and based on the production of SRF, four scenarios were developed (Figure 26): i) Scenario 1 (S1) (landfill disposal); ii) Scenario 2 (S2) (non-densified SRF production with solar drying); iii) Scenario 3 (S3) (non-densified SRF production with thermal drying); iv) Scenario 4 (S5) (densified SRF production with solar drying); v) Scenario 5 (S5) (densified SRF production with thermal drying).

The proposed alternatives develop production processes for non-densified (S2 and S3) and densified (S4 and S5) SRF. The drying stage of screening waste with a moisture level of 77.3% was defined for two processes to compare the alternatives' potential economic and environmental impact: solar drying (S2 and S4) and thermal drying (S3 and S5). The objective of this process would be to achieve a moisture content in the screening waste of approximately 15%, which would meet the moisture requirements for the use of the waste as fuel in some thermochemical processes (ISO/TC 300, 2021). Considering the characteristics of screening waste, all four scenarios have shredding processes in common to reduce and homogenise the particle size of the waste. The output stream of the previous process, the SRF without densification, would be the input stream of the latter process, densification, only present in Scenarios S4 and S5. This is the conditioning of the SRF as pellets through compaction, which generates notable benefits.

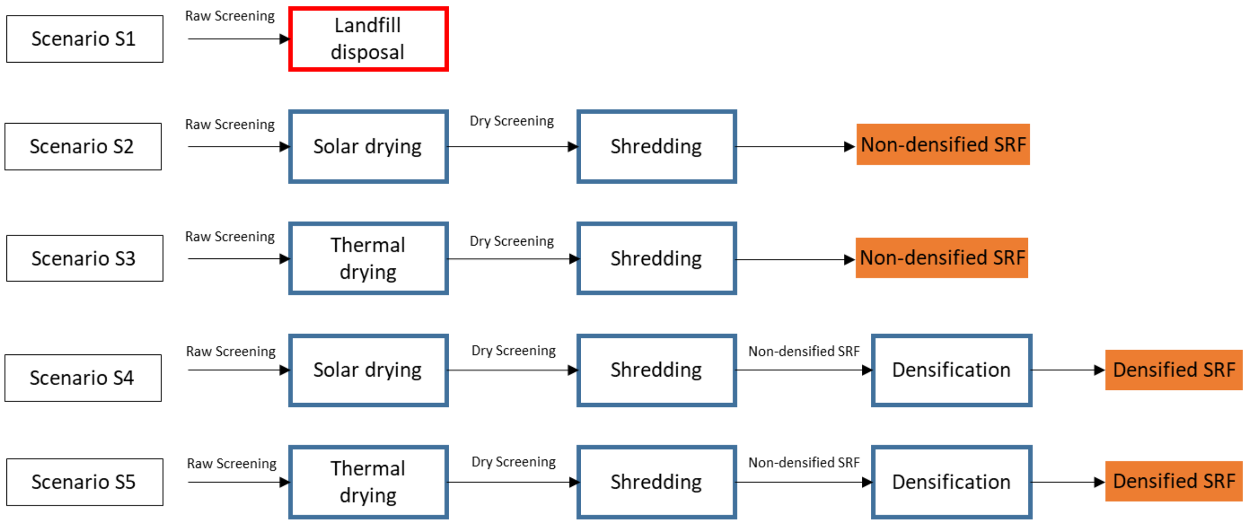


Figure 26. Proposed scenarios, landfill and production of solid recovered fuel (SRF), for screening waste treatment.

2.2 Cost/Benefit Assessment

Costs and benefits were identified to facilitate an economic evaluation for the treatment of screening waste. The main costs studied were assigned to two macro-categories: investment costs required to start a new business, representing a one-time cost, and operating and maintenance costs (OMC), which incur periodically, usually annually.

The profit resulting from the potential sale of the final SRF would represent a specific income for the company. According to the alternative scenarios, there are two output products, the non-densified and the densified SRF. Although the production of SRF is on an increasing trend (Matignon, 2020), the market selling price is still a variable to be established as it depends on a wide range of factors, such as the cost of production, the environmental impact or the quality of the SRF (Chavando et al., 2022). Furthermore, depending on the WWTP, the SRF produced could be valorised energetically through gasification, pyrolysis or gasification processes within the WWTP itself (Shen et al., 2016), without the need for sale.

In addition, the costs derived from the CO_2 emissions generated in the SRF production process are also analyzed. These costs are attributable to all the phases established in the four alternative scenarios and will also be accounted for in the landfill disposal scenario. Although CO_2 emissions do not present specific monetised costs for the company, they offer a broader view of the social and environmental cost of the processes developed (Ni et al., 2022). The amount of CO_2 was measured economically using SendeCO2 (https://www.sendeco2.com/es/precios-co2), a European CO_2 trading system. An average conversion factor of 80.87 €/t of CO_2 was established for 2022, which will be used for this study, considering it to be stable throughout the defined lifetime project.

2.3 Economical analysis; Financial Net Present Value (NPV_F)

There are several methods to evaluate the economic efficiency of the implementation of a process (Alwaeli, 2011). Of these, the NPV_f study was for this study, which has already been used in decision making in the field of energy recovery (Codignole et al., 2015; Cucchiella et al., 2017; Xin-Gang et al., 2016). The economic return and profitability of the potential investments of the company producing the waste were studied for a payback period, after which a neutral NPV_f should be obtained from the cost-benefit ratio. The NPV_f, which is presented in Eq. 5, is the result of cash flows that contain the annual revenues (RE) obtained from the sale of the SRF, the initial investment costs (Io), the operation and maintenance costs (OMC) and the industrial benefit (BE). In this analysis, the starting point was an NPV_f equal to zero over a project lifetime (N) of 10 years, where each cash flow (n = 0,...,N) is discounted from its time n to the present time (n = 0) by the discount rate (r) of 12% (Alao et al., 2022). The Industrial Benefit considered was 6%, according to Spanish Royal Decree 1098/2001 (Ministerio de Hacienda, 2001).

$$NPV_F = -I_0 + \sum_{n=1}^{N} \frac{1}{(1+r)^n}(RE - BE - OMC) \qquad \text{Eq. 3.}$$

The volume of SRF obtained, considering the yields of all production phases, is 37.7% with respect to the input residue, the raw screening waste, mainly due to drying. In addition, a 5% loss of SRF is considered during the production process to obtain the potential SRF to be sold (95% of dry screening). These two percentages are included in the term "SRF relation". Based on the assumption of NPV_f being equal to zero and considering the cash flow, the final simulated sale price (SP) is obtained from Eq. 4.

$$SP = \frac{\dfrac{I_0}{\sum_{n=1}^{N} \dfrac{1}{(1+r)^n}} + OMC}{SRF\ relation * (1 - BE)} \qquad \text{Eq. 4.}$$

2.4 ENVIRONMENTAL ANALYSIS. SOCIAL NPV

The social NPV (NPV_s) is the financial tool that considers all costs challenging to monetize because they do not represent an annual cash flow. The NPV_s includes all economic elements that make up the NPV_f and assumes the social cost resulting from the CO_2 emissions generated during the proposed scenarios. The NPV_s is calculated using Eq. 5, which presents the value of the NPV_f. From this, the economic cost of the CO_2 emissions generated is subtracted and calculated for an r = 12% and a 10-year project lifetime.

$$NPV_S = -I_0 + \sum_{n=1}^{N} \frac{1}{(1+r)^n}(RE - BE - OMC - CO_2) = NPV_F - \sum_{n=1}^{N} \frac{CO_2}{(1+r)^n} \qquad \text{Eq. 5.}$$

2.5 MONTE CARLO SIMULATION

The MC risk analysis technique, applied in quantitative studies in a wide range of areas, including project management, energy, engineering, research and development (Martín-Pascual et al., 2020), was performed to determine the trend, variability and performance under uncertainty. The simulation, performed over a period of 10 years and using the costs and benefits found in the literature, determined a simulated sale price of the SRF for an NPV_f equal to zero. The methodology was applied to 5000 iterations among the different variables, guided by random items of costs and benefits as inputs. The same procedure was also used to determine the NPV_s.

3 RESULTS AND DISCUSSION

3.1 ECONOMIC ANALYSIS

A literature review was carried out with the aim of obtaining a global view of SRF production and a broad spectrum of costs for each scenario. The unit of reference for comparing the different studies analyzed was €/t. The initial as well as operation and maintenance costs are presented for each of the phases present in the scenarios. Table 24 shows the minimum and maximum values found in the literature reviewed. The wide range corresponds to the variability of the studies analyzed regarding the process, location, types of materials or the volume treated.

3.1.1 Initial costs

Solar drying is becoming an alternative to the established thermal drying for processes applicable to municipal and agricultural waste (Bennamoun et al., 2013), using renewable energy and applicable in many parts of the world (Mathioudakis et al., 2013). In the field of wastewater, greenhouses for sludge drying are already being established (Đurđević et al., 2019; Mathioudakis et al., 2013), with mainly environmental advantages (Boguniewicz-Zablocka et al., 2021). However, the investment costs are higher than those of other types of drying and are mainly a factor of the cost of the site, civil works and machinery (Kamarulzaman et al., 2021). For example, the construction and commissioning of a solar drying system for fruit and vegetables in Thailand involved an initial cost of 200.90 €/t, with a drying capacity of 1000 kg every 2–3 days (Janjai, 2012). Literature reviewed on the applicability for sludge from WWTPs showed values in a similar order of magnitude. Four greenhouse sheds for drying a daily production of 48.84 tons of sludge, with a surface area of more than 6000 m^2, involved an initial cost of 230.33 €/t (Risueño, 2020). In another study on several WWTPs of different sizes, a model was established to optimise the possible costs of the implementation of solar sludge drying. The results for the construction of greenhouses, combined with the installation of solar panels, were similar, regardless of the plant size, with initial costs of 116.26 and 134.56 €/t, for sludge productions of 226884.00 and 35.04 tons per year, respectively (Kurt, 2015). According to the data reviewed, the initial cost for the implementation of solar drying as the first phase of SRF production would range from 116.26 to 230.33 €/t of wet waste.

Thermal drying is the most established method in waste management for MSW (Li et al., 2013), sludge (Kelessidis and Stasinakis, 2012) or biomass transformation processes (Del Giudice et al., 2019). However, in most cases, this is neither very cost-effective nor environmentally friendly (Juchelková, 2019). In one study, for a wood pellet production process, the investment cost of a dryer with a feed of 6 t/hour was 397543.60 €, equivalent to 2.45 €/t of wet wood (Mobini et al., 2013). The initial costs for five dryers and a capacity of 75000 tons per year were 22.93 €/t (Pirraglia et al., 2010). In a study comparing the framework conditions for pellet production between Austria and Sweden, the wet waste was dried with different types of dryers, which impacted its initial cost. The tube bundle dryer had an investment cost of 7.92 €/t, for Austria, whereas the drum dryer implementation doubled the cost to 14.07 €/t, for Sweden (Thek and Obernberger, 2004). Considering all the literature reviewed, the minimum value for this phase is 2.45 €/t, whereas the maximum one is 22.93 €/t.

Regarding economic data related to shredding, Zakrisson (2002), comparing the economic costs of pellet production, presented investment costs of 0.94 and 0.74 €/t for plants with a capacity of 10 and 3 t/h, respectively. In the same study, pelletizing had investment costs of 1.58 and 4.67 €/t for 10 and 3 t/h, respectively (Urbanowski, 2005). In the work cited above, the total cost of shredding and pelletizing for the Austrian model was 11.58 €/t. At the same time, for Sweden, it was lower, with a total of 3.5 €/t (Thek and Obernberger, 2004). The maximum initial cost for both processes was derived from the same study, with 3.56 €/t for shredding and 18.74 €/t for pelleting (Pirraglia et al., 2010). Consequently, the possible price range for shredding is 0.74 and 3.56 €/t, whereas that for pelletizing is 1.58 to 18.74 €/t.

3.1.2 Operation and maintenance costs

The OMC data for solar drying were derived from greenhouse studies. The greenhouse built in Thailand (Janjai, 2012), intended for fruit and vegetable drying, had an OMC of 13.63 €/t, corresponding to repair and maintenance costs as well as gas and electricity demand. This value is similar to that reported by Lapuerta and Fonseca (2020), with an OMC of 14.22 €/t. Based on experimental work at laboratory scale to study sludge drying, it was concluded that drying using transparent covers is more effective than conventional drying. Extrapolating the results of (Khanlari and Gungor, 2020) would mean an OMC of 28.51 €/t. At industrial level, data were found on implementing a greenhouse for solar sludge drying in New Zealand. This installation, which allows obtaining 500 tons of sludge with 18% moisture per year, has an OMC of 38.23 €/t of wet sludge (Mwh et al., n.d.). The most economic values found in the literature correspond to the study of the implementation of drying greenhouses, which are complemented with the installation of solar panels, reducing the OMC to 0.97 €/t (Kurt, 2015). These data indicate that the range of OMC for solar drying is between a minimum of 0.97 €/t and a maximum of 38.23 €/t.

In the study on pellet production in Austria and Sweden (Thek and Obernberger, 2004), the OMC of thermal drying was 25.1 and 13.0 €/t, respectively, contrary to the initial installation costs, for which the most significant investment was found for Sweden. Similar values were found in a study on pellet production in Canada (Reza et al., 2013), with an OMC of 20.73 €/t. In the United States, a value of 61.45 €/t for the thermal drying of a pellet production plant was obtained, which can be explained by the high energy consumption of drying, accounting for 70% of the energy consumption of the entire process. Thus, the minimum OMC obtained was 9.54 €/t, with a maximum of 61.45 €/t.

In the comparison proposed by Zakrisson (2002) for pellet plants with different production capacities, the OMC values for crushing and pelletizing are 3.5 and 5.5 €/t, respectively. These results did not vary with the production volume of the plants and were similar for both 10 and 3 tons of pellets. In a comparative study between countries (Thek and Obernberger, 2004), the OMC values and the initial costs were also higher for Austria, both for shredding (2.70 vs 2.30 €/t) and pelletizing (7.60 vs 4.10 €/t). In conclusion, and based on all results found, the OMC values for shredding range between 2.30 and 5.07 €/t, whereas those for pelletizing range from 3.63–13.00 €/t.

Table 24. Value ranges for the initial cost as well as operation and maintenance cost (OMC) of solar and thermal drying, shredding and pelletizing.

Process	Initial Cost (€/t)		OMC (€/t)	
	Min	Max	Min	Max
Solar drying	116.26	230.33	0.97	38.23
Thermal drying	2.45	22.93	9.54	61.45
Shredding	0.74	3.56	2.30	5.07
Pelletizing	1.58	18.74	3.63	13.00

3.1.3 Scenario costs

From the combination of the minimum and maximum costs for each process, both the initial costs and OMC values were defined for each scenario. The values are also shown in €/t of the treated material. For the first drying stage, for both solar and thermal drying, the input stream is the raw screening waste, with 77.3%

moisture, and the costs are therefore relative to the weight of this input. The crushing process has the dry screening waste, containing 15% moisture, as input material. Thus, the costs for this phase were defined according to the material to be shredded, which, after drying, corresponds to 37.7% of the gross input waste. The values for the last stage, concerning pelletizing, were specified for non-densified SRF obtained after shredding, considering that there are no losses. The results for the defined alternative scenarios can be found in Table 25.

Table 25. Value ranges for initial cost as well as operation and maintenance cost (OMC) of the proposed scenarios.

Items		Scenario S2		Scenario S3		Scenario S4		Scenario S5	
		Min	Max	Min	Max	Min	Max	Min	Max
Cost	Initial cost	116.54	231.67	2.73	24.27	117.13	238.74	3.32	31.34
	OMC	1.84	40.14	10.41	63.36	3.21	45.11	11.78	68.33

Regarding the investment costs, there is an evident difference between the scenarios that use solar drying, S2 and S4, with ranges of 116.54–231.67 €/t and 117.13–238.74 €/t, respectively, and those that processed the waste by thermal drying, S3 and S5, with values between 2.73–24.27 €/t and 3.32–31.34 €/t. Solar drying is the phase with the highest investment cost, representing, in average values, 99.53% and 97.39% of the total cost in S2 and S4, respectively. Thermal drying, with lower investment costs, represents a substantial reduction in the total costs, with 94.00% for S3 and 73.35% for S5. These results highlight the significant importance of drying in SRF production processes (Velis et al., 2009). In financial terms of the initial cost, thermal drying should be selected as the best option. The next phase of the SRF production, related to shredding, is common to all four alternative scenarios and therefore does not present any change in the total investment costs. The last process, leading to the conditioning of the final product as pellets, is common to S$ and S5, with increased initial costs. The return on this added cost should be evaluated according to a possible price of the SRF produced, which, once densified, would be higher (Štofová et al., 2021). Considering the above, the scenarios with the highest investment cost would be S2 and S4, mainly due to solar drying. S1, which is currently being performed, does not involve any initial cost since the waste is being disposed of in an external landfill.

The OMC of S1 is composed of the transport and treatment and includes the fees for landfill disposal, depending on the country and the location (Tan et al., 2015). For this research, the OMC of S1 (disposal in landfill) corresponds to the actual values of waste management in the municipality where the primary research for this thesis was carried out. The cost was set at 115 €/t, double the maximum values defined for the most expensive scenarios, S3 and S5. Regarding the proposed alternatives, the presence of the drying process in the OMC, as for the initial costs, continues to be the reference process, with percentages of 93.38%, 96.23%, 81.12% and 88.61% for the four scenarios. This relevance of drying is also present in the production of wood pellets (Pirraglia et al., 2010), where it accounts for 70% of the costs of the entire process. In this case, the trend changes with respect to the type of drying, with thermal drying contributing more OMC to the total than solar drying. Therefore, the optimal option for this phase would be thermal drying. Shredding, accounting between 9.14% and 17.55%, is common to all scenarios, and therefore, its OMC has no impact on the decision making process. Pelletization represents an increase of approximately 5 €/t for S4 and S5, which, as with the initial costs, would theoretically be made profitable by the better quality of the SRF (Kliopova and Makarskiene, 2015). In final terms, the OMC values are higher for S4 and S5, largely because of the expense related to thermal drying, as noted by Thirugnanasambandam et al., (2010).

It is concluded that the drying process, regarding both initial costs and OMC, governs the remaining processes. However, by comparing each drying type's economic advantages and disadvantages, it should be possible to determine the choice that would optimise the SRF production process in monetary terms.

Under the financial conditions of this study, the NPV_f for landfill disposal (S1) is -649.78 €/t. To compare the remaining alternatives and considering the hypothesis of $NPV_f = 0$, an SP of SRF was determined to find the most effective scenario in financial terms. Table 26 shows a summary of the results obtained, with minimum (Min), maximum (Max) and average (Av) values for each scenario, applying MC analysis. S2 and S4, with solar drying, cause the SP of SRF to be the highest, with average values of 159.96 and 172.57 €/t, respectively. Therefore, it is considered that the scenarios with thermal drying (S3 and S5), with average prices of 123.25 and 133.71 €/t, are most economically efficient. For densification, S4 and S5 would mean an increase in the SP of 7.88% and 8.48% compared to S2 and S3. At this point, the potential market for both non-densified and densified SRF should be evaluated to determine the inclusion of pelletizing in fuel production.

Optimising transport as a subsequent step in the production of SRF is crucial in environmental and economic aspects (Hilber et al., 2007). The pelletization of the product substantially increases its density (Ramos Casado et al., 2016) and according to the results obtained in the study exposed in Chapter 3, bulk density increased from 58.16 kg/m^3 for non-densified SRF to 461.78 kg/m^3 for densified one. This decreases the transportation cost of the final product, which is another variable in the choice of a suitable scenario.

Table 26. Value ranges for initial cost as well as operation and maintenance cost (OMC) of the proposed scenarios.

Item	Value	Scenario S2	Scenario S3	Scenario S4	Scenario S5
	Min	74.30	39.49	79.63	44.06
SP (€/t SRF)	Max	245.42	207.01	264.10	224.77
	Av	159.96	123.25	172.57	133.71

To include all results from the MC analysis simulation, graphs of the density function and the price distribution for each scenario are presented in Figure 27. The range class was defined between 0 and 300 €/t to cover the whole set of values assumed by the MC simulation rates in all scenarios.

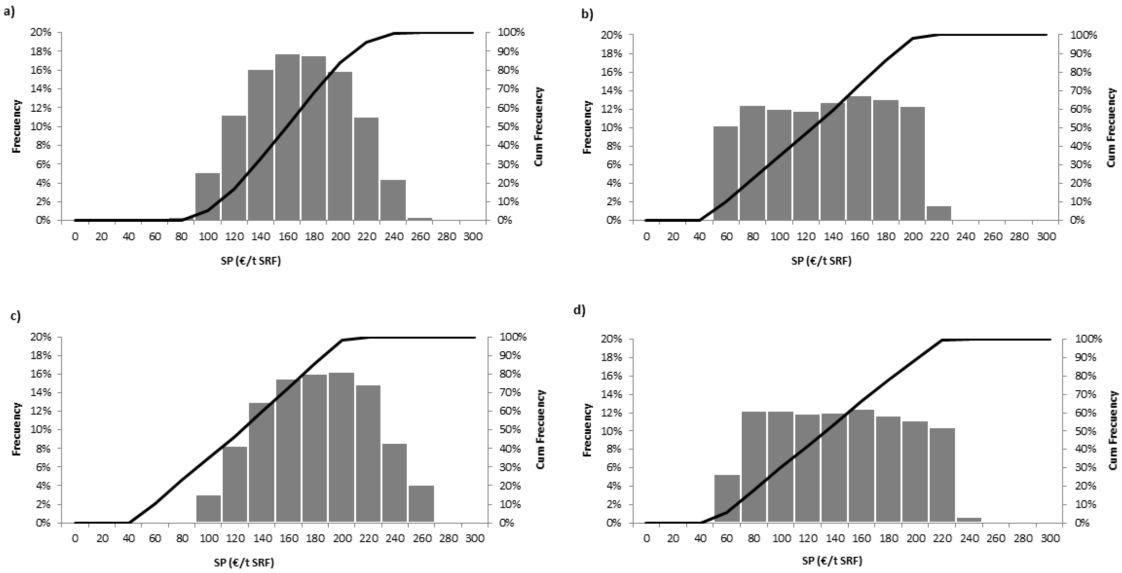


Figure 27. Simulated Price (SP) per ton of solid recovered fuel (SRF) distribution and density function. a) Scenario S2. Non-densified SRF production with solar drying; b) Scenario S3. Non-densified SRF production with thermal drying; c) Scenario S4. Densified SRF production with solar drying; d) Scenario S5. Densified SRF production with thermal drying.

Following the comparison of the scenarios according to the type of drying used, solar drying (S2 and S4) reaches the highest percentages for several classes, obtaining 17.82% and 17.30%, respectively, both for the 180–200 €/t class (Figure 27a and 27c). The two scenarios with thermal drying (S3 and S5) did not reach a 14% frequency for any of the classes considered (Figure 27b and Figure 27d.). Based on these data, the possibilities for each of the established classes would be more distributed in the scenarios with thermal drying, with their distribution being more homogeneous and covering more range classes.

As a reference, the SP of 100 and 200 €/t, present in all scenarios, a probability (P) of the SRF price being below 100 €/t or above 200 €/t can be observed. For S2 (Figure 27a.), P (SP ≤ 100 €/t) was 4.90%, with a further decrease when the pelletization phase is included, resulting in a P (SP ≤ 100 €/t) of 2.64% for S4 (Figure 27c.). Regarding thermal drying, the probability increases substantially with a P (SP ≤ 100 €/t) of 37.10% and 28.64% for S3 (Figure 27b.) and S5 (Figure 27d.), respectively. The results for P (SP > 200 €/t) are in agreement with the financial advantages of the scenarios with thermal drying. For S3, the probability was 1.34%, whereas S5, due to the inclusion of pelletization, presented a result of 11.70%. Solar drying, as the primary source of variation in the scenarios, would generate a P (SP > 200 €/t) of 16.10% for S2 and of 26.40% for S4.

Thus, considering the financial analysis performed, S3 (non-densified SRF with thermal drying) is the most viable one, with the lowest simulated price. In contrast, S4 (densified SRF with solar drying) is the least feasible one.

3.2 Environmental analysis

3.2.1 CO_2 emission

The literature provides data for the CO_2 emissions associated with each process in the different scenarios of this study, including S1. Table 27 shows the amounts of CO_2 (minimum and maximum) generated by each of the processes in the different scenarios.

Table 27. Value ranges for CO_2 emissions of landfill, solar and thermal drying, shredding and pelletizing.

Process	CO_2 emissions (kg CO_2/t)	
	Min	Max
Landfilling	145.00	1,610.00
Solar drying	12.16	141.73
Thermal drying	62.58	137.56
Shredding	0.75	39.30
Pelletizing	1.22	56.90

Emissions generated by landfill disposal have been a relevant issue for years (Gollapalli and Kota, 2018). Concerning S1, according to a study on waste disposal in South Africa, (Friedrich and Trois, 2013) concluded that greenhouse gases emissions could range from 145.00 to 1016.00 kg CO_2/t of wet waste, depending on the type of landfill. However, based on a report by IEA Bioenergy (IEA Bioenergy, 2003), that value could reach up to 1610.00 kg CO_2/t for the landfill disposal of MSW. A life cycle assessment conducted for a landfill in Northern Germany recorded an intermediate emission value of 398.51 kg CO_2/t (Wittmaier et al., 2009). The overall range is between 145.00 and 1,610.00 kg CO_2/t.

Regarding solar drying, a 384 m^2 pilot plant for drying food waste generated 132.01 kg CO_2/t of wet waste (Abeliotis et al., 2022). Almost identical values resulted from the solar drying of tomatoes, with emissions of 132.15 kg CO_2/t produced from substrate with a water content of 94.6% down to 10% (Eltawil et al., 2018). A study of photovoltaic panels in solar-drying greenhouses reported the lowest values from the CIGS PV system (40.96 kg CO_2/t), whereas c-Si modules generated the maximum value (141.73 kg CO_2/t) (Saini et al., 2017). The lowest values for CO_2 emission were found by extrapolating the results obtained for the solar drying of pumpkins. The amount was 12.16 kg CO_2/t for a natural convection greenhouse and 16.44 kg CO_2/t for forced convection (Chauhan et al., 2018). Based on these findings, the CO_2 emissions for solar drying range from 12.16 to 141.173 kg CO_2/t.

The thermal drying of wood sawdust resulted in 72.75 kg CO_2/t in a plant where 20 t of pellets were produced per hour (Mobini et al., 2013). In a study conducted in Sweden, similar values were found, with 62.58 kg CO_2/t for the production of 80 tons per year of pellets (Zakrisson, 2002). The emissions generated in a simulating a small-scale plant in Italy were 137.56 kg CO_2/t for 37% water content drying (Thek and Obernberger, 2004). Overall, the emissions from thermal drying fall within a range of 62.58 to 137.56 kg CO_2 per ton.

In the literature, the CO_2 emission levels of shredding and pelletizing differ greatly. The minimum value for shredding is 0.75 kg CO_2/t (Mobini et al., 2013), similar to that found by Zakrisson M (2002), which is 0.82 kg CO_2/t. However, some authors report values of up to 39.3 kg CO_2/t (Thek and Obernberger, 2004). The data for pelletization follow the same dynamics, with a minimum value of 1.22 kg CO_2/t in (Mobini et al., 2013) and a maximum of 56.9 kg CO_2/t (Thek and Obernberger, 2004). The remaining values for pelletizing, reported by Thek and Obernberger (2004), Urbanowski (2005), and Zakrisson (2002), are within this range.

According to the processes in each scenario and based on the emission price defined above (80.87 €/t of CO_2), the costs for each scenario are shown in Table 28. Any of the proposed alternatives (Scenarios S2, S3, S4 and S5) has a substantially lower cost derived from the generation of emissions than landfill disposal (Scenario 0). The results are hardly comparable, with S1 having a maximum value of 130.20 €/t, whereas the maximum value of the remaining scenarios is 14.39 €/t. Although the alternative scenarios showed similar maximum values, considering the mean value as a reference, there were slightly higher average results, 8.70 and 9.59 €/t, for the scenarios that include thermal drying (S3 and S5) compared to those that include solar drying, with 6.83 and 7.71 €/t (S2 and S4). The minimum and maximum values show a high variability, which impedes decision making.

Table 28. Value ranges of CO_2 emissions and monetised CO_2 emissions of the proposed scenarios.

Items	Value	Scenario S1	Scenario S2	Scenario S3	Scenario S4	Scenario S5
CO_2 emissions (kg CO_2/t)	Min	145.00	12.44	62.86	12.90	63.32
	Max	1,610.00	156.55	152.38	178.00	173.83
CO_2 cost (€/t)	Min	11.73	1.01	5.08	1.04	5.12
	Max	130.20	12.66	12.32	14.39	14.06

3.2.2 Social NPV

Considering the NPVs as a relevant factor in a decision-making process, both financially and socially, it was calculated using Eq. 5, based on the monetised cost of CO_2 emissions described in Section 3.2.1. and the NPV_f. The NPV_s values for each scenario, whose minimum, maximum and average values are shown in Table 29, were obtained by MC analysis. The values obtained are negative since Eq. 5 has the NPV_f as a variable, which is neutral for the alternative scenarios and whose value is -649.78 €/t for Scenario 0.

Table 29. Value range and average values for the Social Net Present Value (NPVs) of the proposed scenarios.

Items	Value	Scenario S1	Scenario S2	Scenario S3	Scenario S4	Scenario S5
	Min	-1,384.93	- 71.53	-69.62	-81.33	-79.43
NPV_s (€/t)	Max	-716.29	-5.69	-28.73	-5.90	-28.94
	AV	-1,052.60	-38.39	-49.25	-43.90	-53.91

The results further highlight the different order of magnitude for costs between S1 and the proposed alternatives, indicating the non-comparability of the NPV_s. Concerning the scenarios leading to SRF production, thermal drying (S2 and S5) is more costly than the options using solar drying (S2 and S4). According to the average values, S3 is the most viable one, with an NPV_s of -38.39 €/t for SRF production without densification. The inclusion of densification together with thermal drying resulted in the maximum NPVs of -53.91 €/t for S5, making this one the least viable one.

Figure 28 shows the results of the MC simulation for the NPV_s. The values ranged from -85 to 0 €/t. The distribution of values was more comprehensive for the scenarios with solar drying (Figure 28a and Figure 28c), covering the entire proposed range with a frequency of approximately 7% for most range classes. The scenarios with thermal drying reached values above 13% for the class between -65 and -60 €/t, S3 (Figure 28b.), and 10% for five different classes, S5 (Figure 28d).

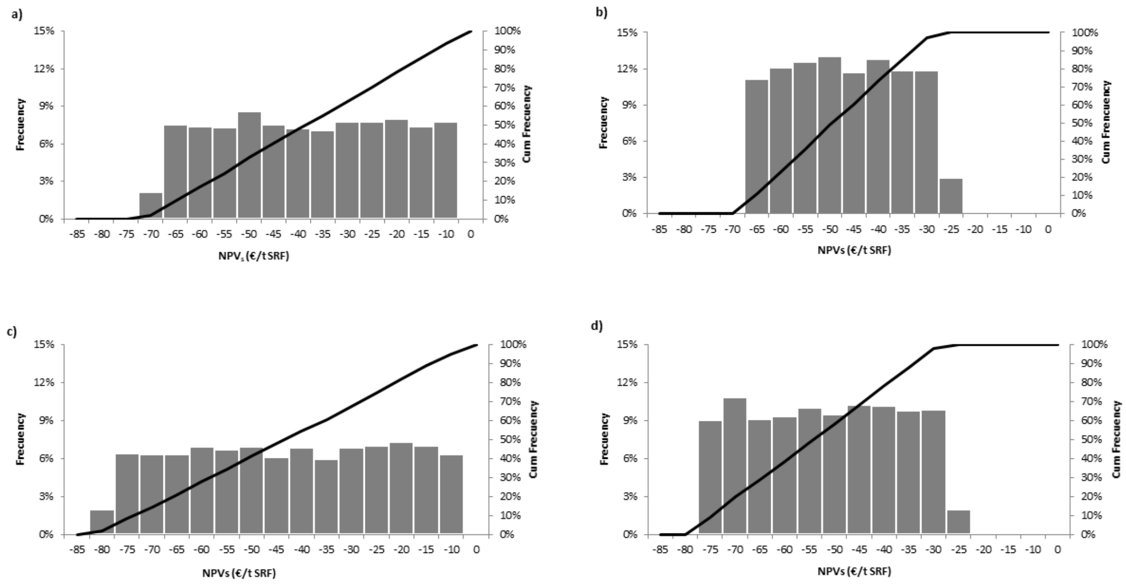


Figure 28. Net Present Value Social (NPVs) for ton of solid recovered fuel (SRF) distribution and density function. a) Scenario S2. Non-densified SRF production with solar drying. b) Scenario S3. Non-densified SRF production with thermal drying. c) Scenario S4. Densified SRF production with solar drying. d) Scenario S5. Densified SRF production with thermal drying.

For the study of the probability (P) of the different ranges, and to compare mainly the drying type, the classes in which S3 (Figure 28b.) had its maximum and minimum limits were considered the reference values. These values for NPVs would be -65 and -25 €/t with a P (NPV_S< -65 €/t) of 0% and a P (NPV_S > -25 €/t) equal to 100%, respectively, for S3. S5 (Figure 28d.) coincides with the P (NPV_S > -25 €/t), equal to 100%. However, when including the densification process, the P (NPV_S < -65 €/t) was higher, reaching 18.82%. The scenarios with solar drying outperformed S5 in terms of the most expensive values, with a maximum P (NPV_S < -65 €/t) of 15.12%. However, the P (NPV_S > -25 €/t) was 27.94% for S2 (Figure 28a.) and 25.86% for S4 (Figure 28c.).

Taking into consideration the values obtained for the NPV_S, S2 (production of SRF without densification using solar drying) is most acceptable in social terms, with the lowest NPV_S. In contrast, S5, with the highest NPV_S, is the least acceptable alternative.

4 CONCLUSIONS

The production of SRF from screening waste, densified and non-densified, was proposed through four scenarios as an alternative to the landfill disposal. The scenarios were evaluated via economic and environmental obtaining the following conclusions:

- In the decision making, both the initial costs and the operation and maintenance costs (OMC) should be considered, as well as the cost derived from CO_2 emissions, which can be combined with the Net Present Value.
- Current landfill disposal does not require any investment costs. However, the costs derived from its management and the high CO_2 emissions produce NPV_S of -1052.60 €/t. This value, compared to that determined for the other scenarios (-53.91 to -39.39 €/t), means that landfill disposal is not considered a viable option.
- Drying costs are the most relevant in SRF production, regardless of whether it is densified or not. Although the OMC values for thermal drying are slightly higher than those for solar drying, the initial investment is substantially lower, making thermal drying the most economically viable option.
- The densification of the SRF implies an increase in the simulated selling price of 7.88% (solar drying) and 8.48% (thermal drying). However, this economic difference must be evaluated concerning the logistical benefits attributed to the densified SRF to evaluate the effectiveness of densification.

CHAPTER 5

ENVIRONMENTAL ASSESSMENT OF SOLID RECOVERED FUEL PRODUCTION FROM SCREENING WASTE

1 INTRODUCTION

Treatment at WWTPs generates waste streams of various compositions and characteristics, such as sands, oils and mostly sludge (Raheem et al., 2018). Bibliometric analysis of the first Chapter of this report come to the conclusion that all these wastes are currently recycled or valorised for energy recovery (Hanum et al., 2019a). However, screening waste from pretreatment generally offers no sustainable alternatives to current landfill disposal, which generates large environmental negative impacts (Boni et al., 2021), such as soil and water pollution, generating harmful gases (Nabavi-Pelesaraei et al., 2017a). As a result, there is a pressing need to identify and implement innovative approaches to waste screening, thus promoting zero-waste strategies in WWTPs (Ballesteros et al., 2022).

In the case of screening waste at the Biofactoría Sur of Granada, as shown in Chapter 3, non-densified and densified SRF was produced at a laboratory scale. The production process includes the drying, cleaning and shredding stages for non-densified SRF. The pelletizing stage is incorporated for densified SRF, taking as variables the size of the matrix and the moisture content of the input stream (Chapter 3). This study has demonstrated the technical feasibility of the production of SRF from screening waste. In addition, Chapter 4 of this thesis presents the results of the economic analysis of the SRF production scenarios as an alternative to landfill. The next necessary step towards potentially implementing the process would be to analyse the environmental impact derived from the exposed processes using a greater number of variables (Tang et al., 2013).

LCA, standardised by ISO 14044 (ISO 14044: Environmental Management—Life Cycle Assessment—Requirements and Guidelines, International Organisation for Standardisation, Geneva, Switzerland (2006), is one of the most useful and established methodologies in the analysis of potential environmental impacts (Ferrari et al., 2021). LCA is a powerful computerised tool that, in the case of waste, analyses impacts from generation to disposal (Finnveden et al., 2009). This methodology identifies and quantifies all inputs (including both energy and resources) and outputs (main emissions to water, air and land) (Mukherjee et al., 2020).

Several studies have applied LCA to assess the environmental impacts of SRF production from different types of waste, such as municipal solid waste, construction and demolition waste and industrial waste (Sora Yi and Jang, 2018). The first publication dates back to 2001 and compares the production of refuse-derived fuel (RDF) in different waste treatment plants, considering wet, dry and pellet RDF as production options (Corti and Lombardi, 2001). The environmental impact of the SRF manufacturing process MSW and its utilisation has been compared in several publications on MSW disposal by landfilling (Fyffe et al., 2016; Grosso et al., 2016; Hupponen et al., 2015; Patel et al., 2012).

Therefore, this thesis Chapter aims to analyse the environmental impact of managing screening waste. For this purpose, the LCA methodology was used to compare the impacts caused by the current landfill disposal versus the potential scenarios of SRF production from the screening waste.

2 MATERIALS AND METHODOLOGY

In this research, environmental impact was evaluated using the LCA methodology, which has been widely applied to evaluate many waste treatments (Corominas et al., 2020). LCA comprises the following phases: (i) definition of the goal and scope, statement of the objective of the analysis, setting of the functional unit and identification of the system boundaries; (ii) inventory and scenario analysis; (iii) impact assessment with assignment of the impact potential of the unit flows to the category indicators and impact factors; and (iv) interpretation of the results (Kovacs et al., 2022; Laurent et al., 2014; Nabavi-Pelesaraei et al., 2017b).

This study used SimaPro 9.2 software with Ecoinvent 3.8 and Agri-footprint as its databases, to allow the modelling and analysis of various life cycles systematically and transparently as well as to measure the environmental impact of processes across selected life cycle stages and to identify the hotspots in all aspects of the chain (Malijonyte et al., 2016).

2.1 GOAL AND SCOPE DEFINITION

This work was carried out based on the results obtained in previous studies developed in Granada, specifically on the wastewater management processes of the Biofactoría Sur. This facility treats more than 18 M m^3 annually and generated 442.18 tonnes of screening waste in 2021.

The objective of such a study includes the rationale and audience for the assessment, while the scope establishes a functional unit (FU) and boundary of the system under analysis (Kovacs et al., 2022). The main objective of this kind of comparative LCA study is to assess the environmental impact of SRF production from screening waste as a substitute for landfill disposal.

The primary function of the process is to transform the waste. An FU is an objective criterion for comparing defined scenarios, relating inputs and outputs (UNE-EN ISO 14040:2006/A1:2021). In such studies, an FU should be defined in terms of input to the system (Cherubini et al., 2009); for this research, this was 1 kg of raw screening waste. This type of waste was defined according to the characterization developed for the screening waste of the Biofactoría Sur: 77.3% moisture, while the remaining total solids were composed of the fractions presented in Table 30, and obtained from Table 11 (Chapter 2).

Table 30. Fractions present in the screening waste

Waste fractions	Description	Process unit in SimaPro	Volume from total solids (%)	Amount in relation to the FU for SimaPro (kg)
Sanitary textiles	Tampons, sanitary towels, wipes etc.	Sanitary textiles	52.10	0.1183
Paper and cardboard	Newspapers, brown corrugated cardboard, package paper rolls, office paper	Waste paperboard, sorted (GLO) market for cut-off, S	11.80	0.0268
Vegetal	Leaves, flowers, plant parts, food scraps, etc.	Wood chips, wet, measured as dry mass (Europe without Switzerland) market for cut-off, S	5.50	0.0125
Plastics	Plastic film, bottles, rigid plastic, packaging, condoms, wrapping and bags	Polystyrene, general purpose (GLO) market for cut-off, S	5.00	0.0113
Other	Fractions that are very costly to separate, including inert	Compost (GLO) market for cut-off, S + sand	25.90	0.0588

Waste fractions	Description	Process unit in SimaPro	Volume from total solids (%)	Amount in relation to the FU for SimaPro (kg)
	debris, hair, organic matter and fine particulates (<20 mm)	(RoW), market for sand cut-off, S		

2.2 PROPOSED SCENARIOS

The inventory modelling stage plays a pivotal role in LCA analyses by establishing a connection between all unit processes within the study up to the final product (Bottausci et al., 2021). This phase strives to procure all the required quantities to develop the product/waste flows and elementary flows, which are subsequently categorised into inputs and outputs within the chosen system boundaries (Magrini et al., 2022). The inputs comprise the materials, energy and resources that enter the unit process, while the outputs comprise the products, waste and emissions generated due to the process (Zanni et al., 2019). Specifically, a gate-to-grave analysis was conducted here, starting from the raw screening up to the various final waste scenarios. Biogas recovery or the thermochemical process after SRF production were not considered part of the present research.

As in the previous Chapter, the following five scenarios were defined (Figure 29):

- Scenario 1 (S1). Disposal in landfill. The current elimination of waste in the landfill of Alhendín (Granada) will be considered, so this scenario will be composed of transport from the WWTP and its subsequent elimination.

- Scenario 2 (S2). Production of non-densified SRF with solar drying. Greenhouse drying will be considered and a shredded fuel with homogeneous particle size will be obtained.

- Scenario 3 (S3). Production of non-densified SRF with thermal drying. For this scenario, drying will be conventional by means of thermal heating, obtaining the same fuel after shredding as in the previous scenario.

- Scenario 4 (S4). Production of densified SRF with solar drying. As a continuation of scenario 2 and as a post-treatment to improve SRF characteristics, in this case the fuel obtained will be in the form of pellets.

- Scenario 5 (S5). Production of densified SRF with thermal drying. Scenario 3 will be complemented with the densification stage to obtain pellets as SRF.

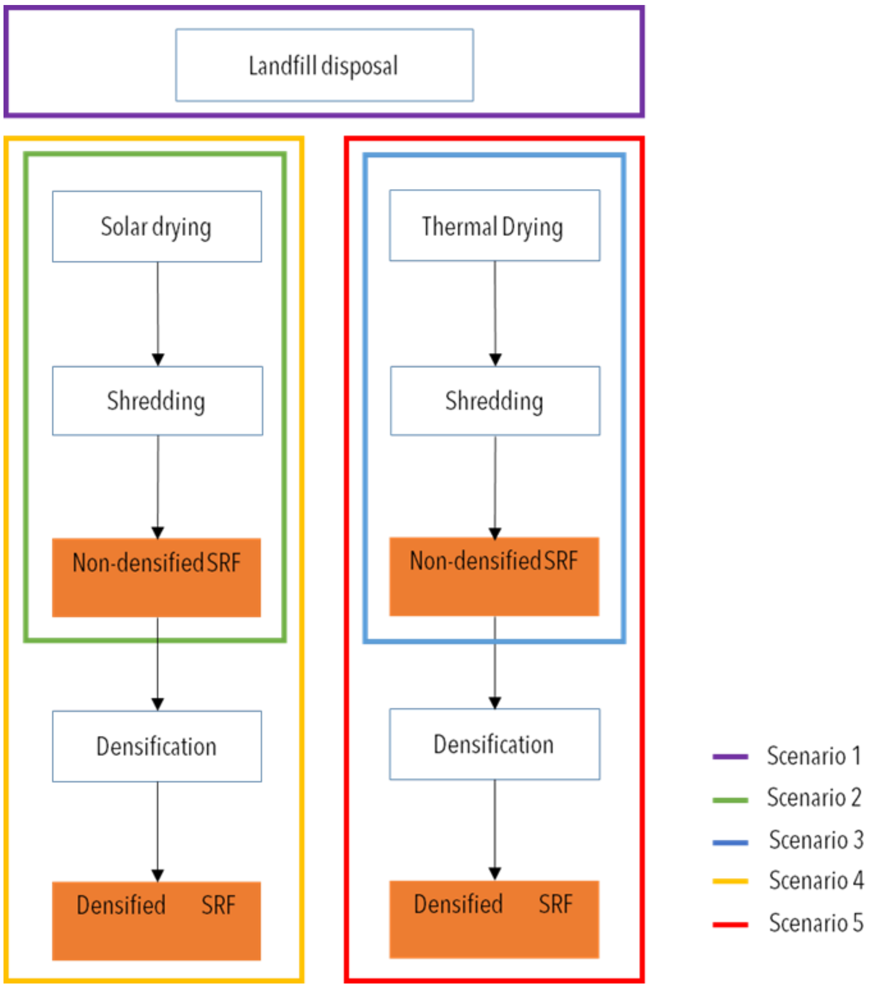


Figure 29. Scheme of scenarios proposed

To set out the boundaries of the system and to define what was and was not included in the environmental assessment, the figures below are presented. The diagrams define the materials and processes as well as the inputs and outputs corresponding to landfill disposal (Figure 30) and alternative SRF production scenarios (Figure 31).

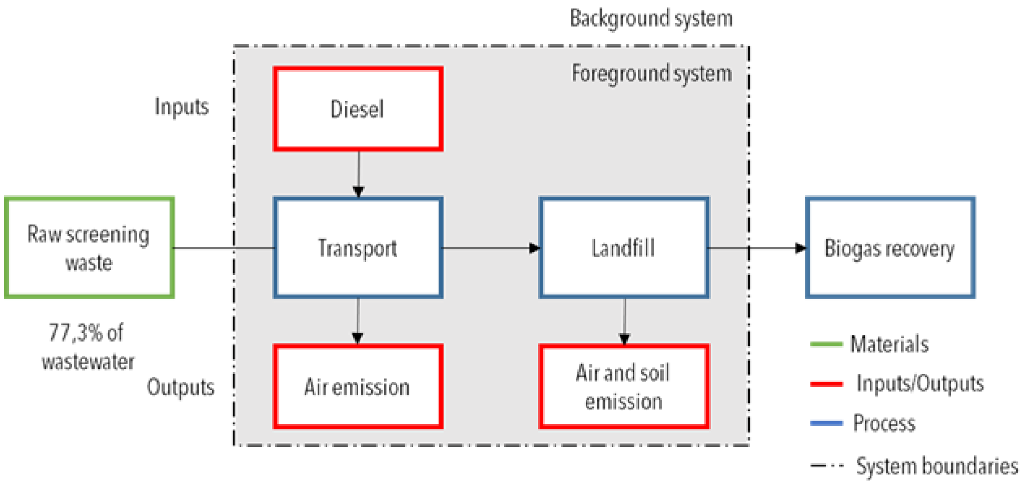


Figure 30. Schematic of landfill disposal of screening waste. Scenario S1.

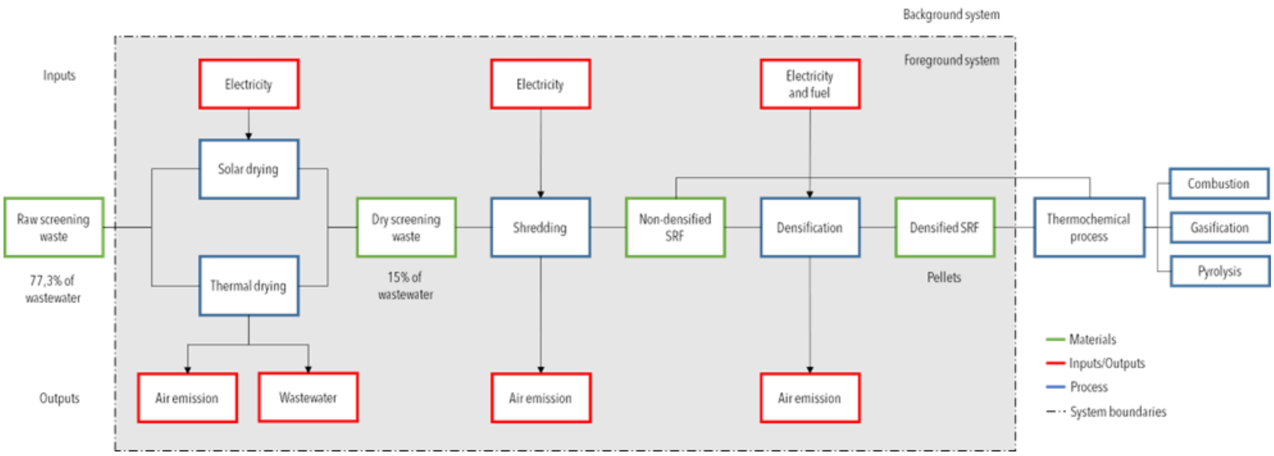


Figure 31. Schematic of solid recovered fuel (SRF) production from screening waste. Scenarios S2, S3, S4 and S5.

2.3 INVENTORY ANALYSIS

All processes were defined for the FU obtained after the compaction process, which was outside the system boundaries. The data collected for the life cycle inventory were mainly primary data obtained from Emasagra, the company that operates the WWTP being considered. This was supplemented with secondary data from the Ecoinvent v.3.8 and Agri-footprint databases, then completed with literature data (Table 31). The inventory of processes involved in the scenarios and defined in SimaPro is described below.

- Transport: Only road transport to landfill was considered, whose environmental impact is fundamental. The distance from the waste collection point (Biofactoría Sur of Granada) to the destination landfill is 19.7 km. For the scenarios with SRF production, all the screening waste treatment would be carried out at the Biofactoría per se, so transport did not need to be considered in such cases.

- Landfilling: It was not possible to obtain data on the emissions produced from the current disposal of screening waste in landfills.

- Drying: The objective of this process is to achieve approximately 15% moisture content in the screening waste, a percentage that some authors regard as 'dry residue', which would also meet the moisture requirements for use as fuel in some thermochemical processes (De la Torre-Bayo et al., 2023b). This stage was defined for two drying processes to compare their potential environmental impact. In the first, solar drying or bio-drying was carried out in a greenhouse containing a scarification roller and a system for moving air in and out (Suez, 2003). The second, which is more established in drying processes, involved trommel drying, which is more efficient for waste with a large amount of water (Malijonyte et al., 2016).

- Shredding: Triturating the already dry screening waste is a complicated task due to the high percentage of sanitary textiles and their resistance to grinding. The process aims to reduce and homogenise the particle size of the residue. Once this stage is completed, the residue is becoming SRF without densification.

- Densification: The output stream of the previous process, non-densified SRF, is the input stream of this process. This stage involves the conditioning of the SRF as pellets through compaction.

Table 31. Life cycle inventory Data for 1 kg of raw screening waste

Inputs (materials/processes)	Database process unit	Unit	Value	Notes
	Transport			
Materials	Raw screening waste	kg	1	Moisture of 77.3%
Processes	SRF_Transport, truck <10 t, EURO3, 20% LF, empty return/GLO energy	tkm	0,0394	
	Landfill			
Materials	Raw screening waste	kg	1	Moisture of 77.3%
Processes	Screening from WWTP (waste scenario). Treatment of municipal solid waste, landfill, cut-off, S	p	1	
	Solar drying			
Materials	Raw screening waste	kg	1	Moisture of 77.3%
Processes	Greenhouse for solar drying	p	1	To evaporate 0.623 kg of wastewater
Electricity	Electricity, high voltage {ES}| market for | cut-off, S	kWh	0,04	
	Thermal drying			
Materials	Raw screening waste	kg	1	Moisture of 77.3%
Processes	Trommel drying	p	1	To evaporate 0.623 kg of wastewater
Electricity	Electricity, high voltage {ES}| market for | cut-off, S	kWh	0,56	
	Trituration			
Materials	Dry screening waste	kg	0,377	Moisture of 15%

Inputs (materials/processes)	Database process unit	Unit	Value	Notes
Processes	Chipper, stationary, electric {GLO}| chipper production, stationary, electric | cut-off, S	p	1	A water loss of 0.3% is considered
Electricity	Electricity, high voltage {ES}| market for | cut-off, S	kWh	0,014	
		Densification		
Materials	Non-densified SRF	kg	0,376	
Processes	Pelletiser found in bibliography (Yay, 2015)	p	1	A water loss of 3% is considered
Electricity	Electricity, high voltage {ES}| market for | cut-off, S	kWh	0.0011	
Diesel	Diesel {Europe without Switzerland}| market for | cut-off, S	kg	0.00377	

2.4 LIFE CYCLE IMPACT ASSESSMENT

In this phase of the LCA, the significance of the potential environmental impact is evaluated by using the life cycle inventory results and associating the inventory data with specific impacts (ISO 14040:2006). These results are translated into the environmental impacts derived from each proposed scenario.

In this study, the methodology used for calculation was CML-IA baseline v3.08 (mid-point system) (Guinée et al., 2001), which focuses on the following 11 impact categories: abiotic depletion (ADP), abiotic depletion fossil (ADP fossil), global warming potential (GWP100a), ozone layer depletion potential (ODP), human toxicity potential (HTP), freshwater aquatic ecotoxicity potential (FAETP), marine aquatic ecotoxicity potential (MAETP), terrestrial ecotoxicity potential (TETP), photochemical oxidation (POI), acidification potential (AP) and eutrophication potential (EP).

3 RESULTS AND DISCUSSION

The LCA results for each impact category for the proposed scenarios in the screening waste treatment are presented in Table 32. For clarity, for each impact category, the results are normalised to that of the scenario with the highest impact in the category (Figure 32).

It can be seen that landfill disposal generates the most significant impact in six of the 11 categories proposed by the CML-IA baseline v3.08 method, with noteworthy differences in FAETP and EP. The most negative impact in the rest of the categories corresponds to scenario S5, with very similar data to S4. The categories on the depletion of natural resources (ADP and ADP fossil) stand out, in which the impact of the increased electricity production required for the processes is exposed. On the other hand, scenario S2, the production of SRF without densification through solar drying, is the most environmentally viable alternative, with the lowest impact index in six of the 11 categories.

Table 32. Life cycle characterization results

Impact category	Unit	S1	S2	S3	S4	S5
Abiotic depletion	kg Sb eq	4.40E-06	1.33E-05	1.35E-05	1.93E-05	1.95E-05
Abiotic depletion (fossil fuels)	MJ	2.90E+01	2.88E+01	3.06E+01	4.17E+01	4.36E+01
Global warming (GWP100a)	kg CO_2 eq	2.41E+00	1.27E+00	1.43E+00	1.83E+00	1.99E+00
Ozone layer depletion (ODP)	kg CFC-11 eq	5.05E-07	1.56E-06	1.56E-06	2.25E-06	2.26E-06
Human toxicity	kg 1,4-DB eq	5.19E-01	6.43E-01	7.06E-01	9.28E-01	9.90E-01
Fresh water aquatic ecotox.	kg 1,4-DB eq	2.11E+00	4.87E-01	5.42E-01	7.02E-01	7.56E-01
Marine aquatic ecotoxicity	kg 1,4-DB eq	1.40E+03	1.35E+03	1.60E+03	1.94E+03	2.19E+03
Terrestrial ecotoxicity	kg 1,4-DB eq	3.18E-03	1.61E-03	1.81E-03	2.32E-03	2.52E-03
Photochemical oxidation	kg C_2H_4 eq	6.30E-04	3.02E-04	3.45E-04	4.36E-04	4.79E-04

Impact category	Unit	S1	S2	S3	S4	S5
Acidification	kg SO_2 eq	9.94E-03	4.65E-03	5.82E-03	6.69E-03	7.86E-03
Eutrophication	kg PO_4--- eq	4.93E-03	1.26E-03	1.52E-03	1.80E-03	2.07E-03

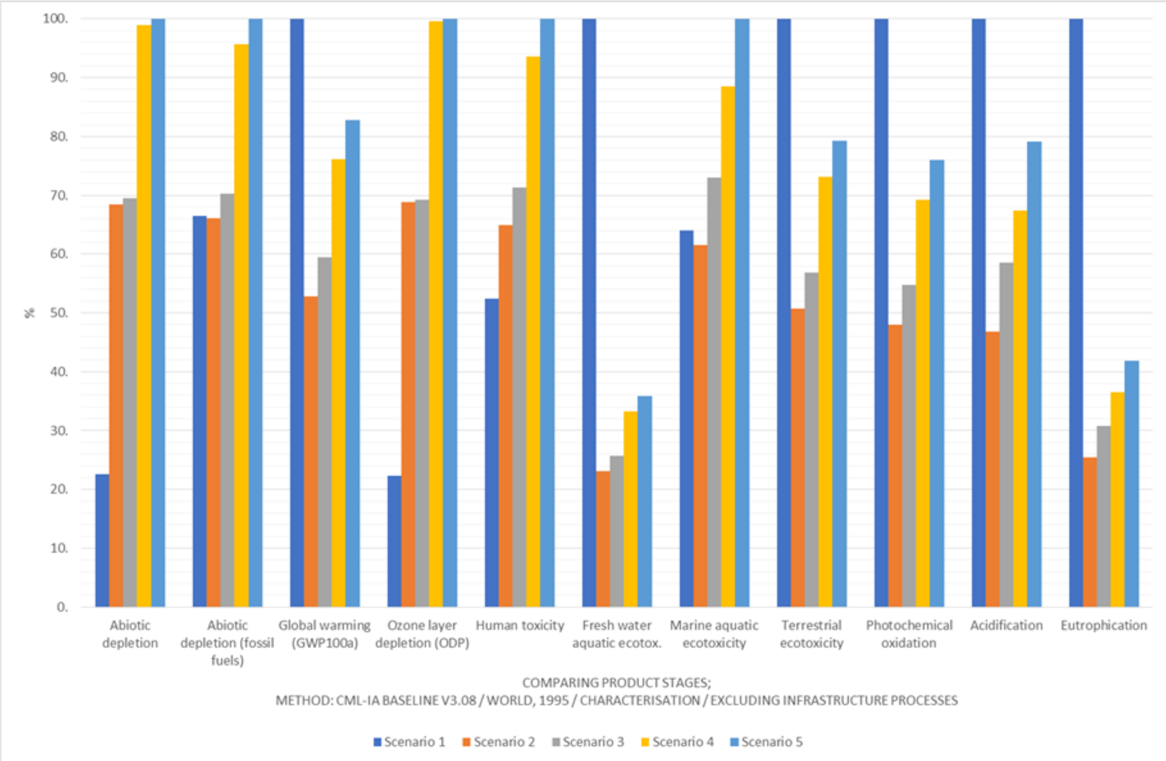


Figure 32. Comparison of impact assessment of the various scenarios according to the CML-IA baseline v3.08 methodology.

The different categories of impacts shown in Table 32 and Figure 32 are analyzed below:

- The abiotic depletion and abiotic depletion fossil (ADP and ADP fossil) categories are determined by extracting minerals and fossil materials (Rajcoomar and Ramjeawon, 2017). The highest impacts recorded in ADP and ADP fossil are 1.95E-05 and 4.36E+01 kg Sb eq (Table 32), respectively, corresponding to S5 (densified SRF with thermal drying). It can be observed that the scenarios with densification (S4 and S5) have the most significant impact on these categories due to the increased consumption of fossil fuels (such as coal and natural gas) for electricity production, as necessary for the densification process (Siddiqui et al., 2021). In these cases, densification represents an increase of approximately 30% with respect to the scenarios that produce SRF without densification (S2 and S3). Landfill disposal (S1) presents the lowest values for ADP, due to the absence of utilisation of any mineral/elements during landfilling (Aryan et al., 2019). The results for fossil ADP for S1 are very similar to those of S2 and S3 and represent an increase with respect to ADP due to diesel transportation, as has been demonstrated in a study concerning the treatment of MSW in Sakarya (Yay, 2015).

- Global warming potential (GWP100a) represents, in general terms, the increase in temperature due to greenhouse gas (GHG) emissions such as CO_2, CH_4, N_2O and CFCs (Khandelwal et al., 2019) over a time horizon of 100 years. As expected, Table 32 shows that landfill disposal (S1) contributes the most to this category, with 2.41E+00 kg CO_2 eq per kg of raw waste, values higher than the 0.90E+00 kg CO_2 eq identified for the landfill disposal of MSW in Brazil (Mendes et al., 2003) and 0.63E+00 kg CO_2 eq for the same treatment in Thailand (Liamsanguan and Gheewala, 2008). Such high impact is due to atmospheric emissions, mainly of CO_2, CH_4 and N_2O, which are generated in the degradation of organic matter in landfill (Edwards et al., 2018), accounting for 60.5% which was obtained in the characterization of this waste (Chapter 2). The amount of kg CO_2 eq per kg of raw waste for the rest of the scenarios is higher for S4 and lower for S2. GWP100a in the alternatives scenarios is derived from emissions from fossil combustion for energy production in the SRF process (Ripa et al., 2017). This assertion is corroborated in Table 31, which shows that the energy consumption of each process in the scenarios is related to the impact on this category.

- Ozone layer depletion potential's (ODP) impact is mainly caused by the emission of methane bromotrifluoro-Halon 1301 resulting from the production of oil and natural gas (Yadav and Samadder, 2018). In this case, scenarios with higher energy consumption (S4 and S5) have the most significant environmental impact in this category, with 2.25E-06 and 2.26E-06 kg CFC-11 eq, respectively (Table 32), which is about 30% more than S2 and S3 and consistent with the results of a study on wood pellet production (Fantozzi and Buratti, 2010). This densification-driven increase is similar to that shown in the abiotic depletion categories, demonstrating that all three categories (ADP, fossil ADP and ODP) are proportionally linked to fossil resource expenditure, as Hospido (Hospido et al., 2008) has pointed out in an LCA on wastewater treatment, in which he concluded that ODP is entirely dependent on electricity production.

- Human toxicity potential (HTP) measures human exposure to toxicity from elements such as lead, zinc and other contaminants within dichlorobenzenes (Goedkoop, 2008). Table 32 exposes that the highest impact values are for densification scenarios, with 9.28E-01 and 9.90E-01 kg 1.4-DB eq, for S4 (solar drying) and S5 (thermal drying). Comparing the results with the production of non-densified SRF, values of 5.19E-01 and 6.43E-01 kg 1.4-DB eq are observed for scenarios S2 (solar drying) and S3 (thermal drying), respectively, which represent values 43% and 35% lower, owing to the extra energy consumption in densification (S Yi and Jang, 2018). In this category, the impact of S1, which is 50% lower than the highest figure, is determined by transportation; however, compared with the other scenarios, it has a smaller footprint due to the difference in energy expenditure. This assertion aligns with an LCA on alternatives to landfilling for MSW, in which this type of disposal also did not have the most significant impact on HTP (Rajcoomar and Ramjeawon, 2017).

- Regarding the ecotoxicity categories, it can be seen that the scenarios' impact is caused by the emission of contaminants into water, whether to oceans and seas, or to rivers and to the soil (Corona and Miguel, 2015). S1 causes the most significant impact on freshwater aquatic ecotoxicity potential (FAETP) and terrestrial ecotoxicity potential (TETP) with values of 2.11E+00 and 1.40E+03 kg 1.4-DB eq (Table 32), possibly due to contaminants such as nickel, arsenic, lead, zinc, mercury and barium, which are discharged during landfill disposal (Yadav and Samadder, 2018). The maximum for the marine aquatic ecotoxicity potential (MAETP) category corresponds to S5, with 3.18E-03 kg 1.4-DB eq (Table 32). The normalised values of the alternative scenarios to landfill for these categories show that their differences follow a similar progression. The percentages for FAETP are 23, 26, 33 and 36% for scenarios S2, S3, S4 and S5, respectively. For MAETP's impact, the values are, respectively, 62, 73, 89 and 100%, while for TETP, the results reach 51, 55, 73 and 77% (Figure 4). Analysing these values, it can be observed that the differences between solar drying and thermal drying for both non-densified and non-densified SRF are practically the same across the three categories. Thus, in these categories, thermal drying negatively affects 3% of FAETP, 12% of MAETP and 5% of TETP.

- Photochemical oxidation (POI) defines the reaction of nitrogen oxides with volatile organic compounds to produce tropospheric ozone (Khandelwal et al., 2019). In this case, landfill disposal is considered the worst scenario, with a value of 6.30E-04 kg C_2H_4 eq (Table 32), mainly due to methane emissions (Yay, 2015), as also asserted by Abeliotis (2012) in a paper on MSW management in Athens. The other scenarios have little impact on this category, in line with Edwards et al.'s (2018) research on the environmental impact of a mechanical-biological treatment plant for food waste.

- The production of SO_x, NO_x, H_2S, HF, HCl and HNO_3 causes environmental acidification potential (AP) and consequent damage to continental ecosystems (Atta et al., 2020). Landfill disposal (scenario S1) has the most significant impact, with 9.94E-03 kg SO_2 eq (Table 32), due to sulphur emission (Yay, 2015). Regarding SRF production, the most significant impact is caused by the scenario with thermal drying and densified fuel production (S5) (Eriksson et al., 2016). Densification represents an increase of 26% for scenarios with thermal drying (S3 and S5) and 31% for solar drying (S2 and S4).

- The impact on eutrophication potential (EP) is where nitrogen and phosphorus affect terrestrial and aquatic systems (Fantozzi and Buratti, 2010). In this category, there is the most noteworthy difference between the results of the proposed scenarios – 58.13% – between landfill disposal with 4.93E-03 kg PO_4- eq and the following most impactful scenario (S5) with 2.07E-03 kg PO_4- eq (Table 32). The percentages of increase implied by densification in this category for the rest of the scenarios are very similar to those for acidification potential (AP): 27% and 30% between the scenarios with thermal and solar drying, respectively.

4 CONCLUSIONS

Regarding the life cycle assessment about the environmental impacts derived from current landfill disposal of screening waste and the four scenarios proposed for potential production of SRF, the following conclusions are drawn:

- The use of LCA can provide valuable insights to guide decision-making processes towards more environmentally and economically viable options. The results of this LCA study show that landfill disposal has the most damaging environmental impact among the waste management options evaluated. This is due to the release of various contaminants, such as heavy metals, organic pollutants and greenhouse gases, during the decomposition of screening waste in landfill. These pollutants can have significant impacts on air, water and soil quality as well as on human health and ecosystems.
- The production SRF from screening waste, especially without densification and with solar drying, is the most environmentally viable process among the scenarios evaluated here. This process has lower environmental impacts compared to landfill disposal, as it avoids the emissions of pollutants during landfill decomposition and reduces reliance on fossil fuels for energy production.

CHAPTER 6

STUDY OF THE APPLICABILITY OF THERMOCHEMICAL PROCESSES FOR SOLID RECOVERED FUEL

1 INTRODUCTION

Treatment at WWTPs generates waste streams of various compositions and characteristics, such as sands, oils and mostly sludge (Raheem et al., 2018). As pointed out from the bibliometric analysis presented in Chapter 1 of this thesis, all these wastes are currently recycled or valorised for energy recovery (Hanum et al., 2019a).

The feasibility of SRF production from screening waste of WWTP has been studied in technical, economic and environmental terms and is presented in this report in Chapters 3, 4 and 5 respectively. On a technical level, the feasibility of the process was demonstrated for both the production of non-densified and densified SRF with input moisture up to 40%. In economic and environmental terms, landfill disposal was found to have the most negative impacts compared to any of the SRF production scenarios, so it is necessary to bet on these alternatives.

The properties of the produced biofuel met the requirements of ISO 21640:2021 (AENOR, 2021) for LHV, chlorine content and mercury content. At this point, the scope of the study for the management of this SRF produced is in the fourth step of the waste hierarchy proposed by Directive 2018/850 (European Parliament and Council, 2018), which refers to "other recovery, e.g. energy recovery". In this context, thermal routes such as combustion and gasification, which aim at the energy recovery of SRF (Al-Moftah et al., 2021), and pyrolysis, whose focus is mainly on the generation of value-added products in the form of gas, liquid and solid (Chen et al., 2015), should be considered. Thus, a comparison between the three thermochemical processes is not possible from an energetic point of view (Kumagai et al., 2015), as in addition to having different objectives, combustion and gasification are exothermic processes, while pyrolysis is endothermic (Khiari et al., 2004).

Combustion, also known as controlled incineration, is a process that involves the burning of some feedstock in the presence of oxygen at temperatures between 850 and 1100 ºC and can be applied to waste (Edo-Alcon et al., 2016). The primary purpose of this process is to reduce the volume and weight of the waste while recovering some of the energy contained in the waste to generate electricity, steam or heat (Karlsson et al., 2015). After incineration, effluents, atmospheric emissions and the resulting ash must be treated appropriately (Wu et al., 2013). In combustion, controlling different parameters, such as excess oxygen, minimum combustion temperature and minimum residence time at minimum combustion temperature after the last air injection (Vainio et al., 2013), are essential parameters for the proper development of the process (Lombardi et al., 2015). SRF is produced from non-hazardous solid waste components and its composition aligns with the combustion requirements set by national and EU specifications (Gerassimidou et al., 2020). There are numerous studies on the use of SRF obtained from the non-recyclable part of municipal solid waste in combustion processes in cement kilns, lime kilns and WtE plants (Conesa et al., 2011) in which mainly the economic and environmental impact was analyzed, confirming that alternative could be viable solution (Reza et al., 2013). Currently, the thermal substitution rate of traditional fossil fuels with SRF in an individual plant can vary from 40% to 70% (Saveyn et al., 2016).

Gasification converts a carbonaceous material into a gaseous fuel by heating it in a gasification medium, such as air and oxygen. The gas obtained, whose quality will depend on its composition and the presence of tars generated during gasification (Nguyen et al., 2020), can be used in gas engines and turbines or as a chemical feedstock to produce liquid fuels (McKendry, 2002). Operating conditions, such as temperature, equivalence ratio (ER) or gasifying agent, play a relevant role in gas quality (Santamaria et al., 2021). The influence of temperature and ER has been analyzed in several works (Recari et al., 2016), resulting in general terms that increasing the temperature with a fixed ER implies an increase in gas yield (Arena and Di Gregorio, 2016). SRF has been the subject of gasification in several studies, both from MSW (Al-Moftah et al., 2021) and automotive and plastics recycling industries (Vonk et al., 2019). It has also served as feed for co-gasification processes with other input streams such as wood (Pinto et al., 2014), sewage sludge or paper waste (Akkache et al., 2016). The LHV of the resulting gas has also been the subject of study about the previous variables (Dunnu et al., 2012). However, unanimity on its effect has not been achieved, which, as concluded (Hervy et al., 2019), seems more related to the composition of SRF.

The pyrolysis process, through the decomposition of organic matter, generates three products that can be valuable for energy or added-value products (Ruiz Gómez et al., 2017). Pyrolysis oil, also known as bio-oil, has a high calorific value and the potential to substitute other commercial fuels (Quesada et al., 2019), in addition, its chemical composition may be of great value in the production of carbon blacks (Okoye et al., 2021). The properties of char, the solid product obtained from pyrolysis, may propose it as a low-cost adsorbent, besides being able to be used as a fertilizer and improvement for soils (Nobre et al., 2019). In addition, although it is not the main objective of the process, a review of the gas obtained from pyrolysis determined its possibility of being an alternative to other gases for producing energy from their combustion in gas turbines or internal combustion engines (Asimakopoulos et al., 2018).A variety of reactor configurations have been employed for pyrolysis, with the primary ones being: i) fixed bed, ii) fluidized bed, iii) moving bed, iv) rotary bed (Afailal et al., 2023) . However, most pyrolysis studies in the literature have been carried out using fixed-bed reactor configurations in laboratory-scale plants (Santamaria et al., 2021). The consequences derived from variables such as temperature and pyrolysis heating rate have been studied for SRF, generally from MSW (Tokmurzin et al., 2022). Primarily the literature provides studies on the distribution in the three output streams concerning the input variables (Zhou and Yang, 2015). Gas composition (Sabogal et al., 2021), GC-MS analysis of oil (Park et al., 2019) and char adsorption capacity (Al-Rahbi et al., 2016) have also been investigated in the application of pyrolysis in SRF.

To the best of our knowledge, there is a lack of research on the application of thermochemical processes to screening waste in the existing literature. This research paper is based on successfully analyzing the technical feasibility of SRF production from screening waste. Energy balance and experimental studies were developed to evaluate the possibility of applying thermochemical processes as combustion, gasification, and pyrolysis to SRF from screening waste. The findings of this study provide potential uses for the SRF produced and offer sustainable alternatives to the current landfill disposal of the screening waste.

2 MATERIALS AND METHODOLOGY

2.1 RAW MATERIALS

The main input stream considered for the experiments and energy balances was the densified and non-densified SRF, which was characterized in Chapter 3. In addition, some experiments have been conducted for sanitary textiles, which, as discussed in Chapter 2, are the predominant fraction in the screening with 52.10%.

2.1.1 SRF samples preparation and characterization

The SRF used for the thermochemical processes was the one described in Chapter 3 of this report. SRF was produced after drying, cleaning, crushing and a subsequent pelletization in the case of densified SRF. Table 33 provides a brief characterization, including the properties of the ultimate and proximate analyses, with the results on a dry basis, as fed to the gasification and pyrolysis plants. The determination of properties was carried out at facilities of the University of Granada and Zaragoza according to the following method standards: elemental analysis according to UNE-EN-ISO 21663:2021; ash according UNE-EN ISO 21656 2021; volatiles based on UNE-EN ISO 22167:2022; LHV according to UNE-EN ISO 21654-2021; Cl content was determined from a sample of ash derived from the procedure of calorific value determination which was diluted in distilled water, and calculated applying ion-exchange chromatography based on UNE-EN ISO 10304-1:2009; Hg content according to UNE-EN 15411:2012; and bulk density according to UNE-EN 15103:2010.

Table 33. Characterization of solid recovered fuel (SRF) from screening waste.

Parameter (dry basis)	Value
C (%)	52.80±0.40
H (%)	7.43±0.09
N (%)	2.685±0.007
S (%)	0.010±0.002
O (%)	27.40±0.50
Ash (%)	9.40±3.40
Volatile solids (%)	91.00±2.80
Cl (wt%)	0.31±0.18
Hg (mg/MJ)	$3.8*10^{-5}±2.9*10^{-5}$
Higher Heating Value (MJ/kg)	26.00±2.70
Lower Heating Value (MJ/kg)	24.30±2.60

Parameter (dry basis)	Value
Bulk density (kg/m3)	58.16±2.86

Non-densified dried SRF was used in the pyrolysis experiments, whereas the SRF was densified in pellet form to enable continuous feeding in the gasification runs. Table 34 shows the range of mechanical characteristics of the densified SRF as pellet, which were calculated according to the following standards: diameter and length according to UNE-EN 16127:2012, pellet density according to UNE-EN 15150:2012, bulk density according to UNE-EN 15103:2010, the durability according to UNE-EN 15210-1:2010, and hardness according to previous studies (Garcia-Maraver et al., 2015) using a manual hardness tester (Amandus Khal mod. 21465).

Table 34. Characterization of densified solid recovered fuel (SRF pellets) from screening waste.

Parameter	Value (min – max)
Moisture (%)	7.75 – 34.80
Pellet diameter (mm) (wt basis)	5.84 – 9.04
Pellet Lenght (mm) (wt)	17.79 – 36.80
Pellet density (kg/m3) (wt)	522.61 – 1198.03
Bulk density (kg/m3) (wt)	301.07 – 517.53
Durability (%) (wt)	62.63 – 99.76
Hardness (kgf) (wt)	3.33 – 20.00

More details on the screening waste and the SRF produced from it can be found in Chapter 3.

2.1.2 Sanitary textiles samples and characteristics

As was determined in Chapter 2, which presents the analysis of the composition of the screening waste, sanitary textiles fraction is the majority one with 52.10% of the total. This fraction was undergone to pyrolysis process. Its characterization (based on the same standards that SRF characterization) is shown in Table 35. It is composed of wipes and other hygiene products, including cellulose and several synthetic fibres such as high-density polyethylene, polyethylene-vinyl acetate, polypropylene and polystyrene (Marques et al., 2020).

Table 35. Characterization of sanitary textiles

Parameter (dry basis)	Value
C (%)	47.4±0.9
H (%)	7.5±0.4
N (%)	2.6±0.2
Ash (%)	8.0±3.0
Volatiles (%)	92.0±2.0
Cl (wt%)	0.41±0.04
Hg (mg/MJ)	5*10-5±1*10− 5
Lower Heating Value (MJ/kg)	22.7±0.3
Bulk density (kg/m3)	31.2±1.3

2.2 COMBUSTION

An energy balance was carried out to predict the amount of energy required for the optimal operation of the process (Figure 35), which has the outgoing gas temperature (T) as the main incognita. The analysis was based on experimental data, which are described in Chapter 2, and literature data (Abedin et al., 2013). Moisture content is a key factor influencing the performance of alternative fuels in incineration, including combustion completeness, energy recovery and pollutant emission (Xiao et al., 2015). So, in consideration of improving the operation and mitigating corrosion, moisture should not exceed 10% (Xu et al., 2018).

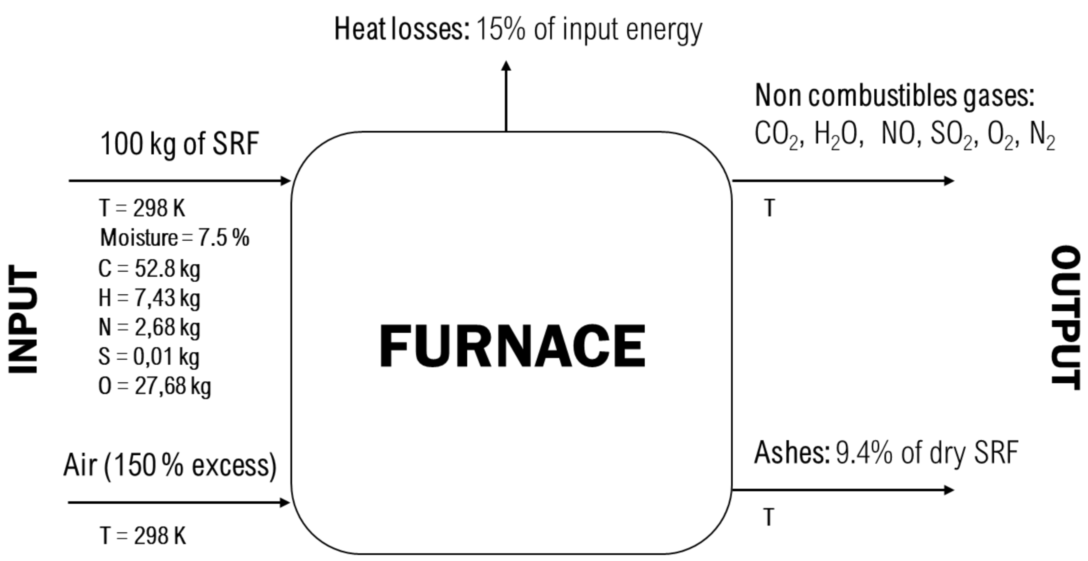


Figure 33. Combustion energy balance diagram

The following assumptions, simplifications and stream properties were adopted for the energy balance:

- The standard reference state was T0 = 25 ºC (298 K) and P0 = 1.01 x 10^5 Pa.
- The characterization of the input solid stream is shown in Table 33.
- Input mass considered was of 100 kg of raw SRF.
- Input air excess considered was of 150%.
- Moisture contents considered of SRF stream was of 0, 2.5, 5, 7.5 and 10%.
- Liquid phases were considered to be ideal solutions.
- Thermodynamic properties of liquid and gaseous compounds were obtained from the literature (Perry et al., 1998).
- Heat losses were taken into account as 15% of input energy (Manganaro et al., 2011).
- It is considered an ideal case, with perfect and total combustion of the SRF, with no unburned fuels.
- The estimation of input energy with energy recovery provided a maximum reference value for the energy efficiency of the combustion processes considered.

Energy assessment

The procedure followed for the energy balance determined the heat required for the process in the case that feedstock and products were considered to be at the standard reference state. The enthalpy balances were expressed by Eq 6.

$$h_{input} = Q_{losses} + h_{output} \qquad \textit{Eq. 6}$$

where:

Q_{losses}: heat losses, 15% of input energy, MJ/kg

h_{input}: specific input enthalpy, MJ/kg

h_{output}: specific output enthalpy, MJ/kg

The composition of the output gas is a consequence of several intricate reactions, among which the most significant ones are presented below:

$$C + O_2 \rightarrow CO_2$$

$$2H_2 + {}^{1}/_{2}O_2 \rightarrow H_2O$$

$$2N + O_2 \rightarrow 2NO$$

$$S + O_2 \rightarrow SO_2$$

The enthalpy of the input stream (h_{input}) was calculated using Eq. 7.

$$h_{input} = \Delta h_f^0,_{\,feedstock}$$

$$h_{input} = m_{SRF} \cdot h_{SRF} + m_{H_2O} \cdot h_{H_2O\,(liq)} + m_{O_2e} \cdot h_{O_2e} + + m_{N_2e} \cdot h_{N_2e} \qquad \textit{Eq. 7}$$

where:

$\Delta h_f^0,_{feedstock}$: standard enthalpy of formation estimated for the feedstock, MJ/kg $_{feedstock}$

m_{SRF}: mass of input SRF, kg

h_{SRF}: enthalpy of input SRF, MJ/kg $_{SRF}$

m_{H_2O}: mass of input water, kg

$h_{H_2O\ (liq)}$: enthalpy of input water in liquid form, MJ/kg $_{H_2O}$

m_{O_2e}: mass of input O_2 (21% of input air steam), kg

h_{O_2e}: enthalpy of input O_2, MJ/kg $_{O_2}$

m_{N_2e}: mass of input N_2 (79% of input air stream), kg

h_{N_2e}: enthalpy of input N_2, MJ/kg $_{N_2}$

The enthalpy of the output stream (h_{output}) was calculated using Eq. 8.

$$h_{output} = \sum m_i \cdot \Delta h^0_{f,i} + \sum m_i \cdot Cp_i\ (T - 298) \qquad \text{Eq. 8}$$

where:

m_i: mass of product i, kg $_{\text{product i}}$

$\Delta h^0_{f,i}$: standard enthalpy of formation estimated for product i, MJ/kg $_{\text{product i}}$

Cp_i: specific heat capacity at constant pressure for product i, MJ/kg K $_{\text{product i}}$

T: reference temperature, K

Finally, from the combinations of equations 1, 2 and 3, the T of the output gas could be obtained from Eq. 9.

$$T = \frac{h_{input} - |Heat\ losses| - \sum m_i \cdot \Delta h^0_{f,i}}{\sum m_i \cdot Cp_i} \qquad \text{Eq. 9}$$

The gases obtained must pass through a heat exchanger to cool them before they are subsequently expelled. An energy balance is also generated in the heat exchanger according to Figure 34, in which a mass of water vapour at 400 °C is obtained.

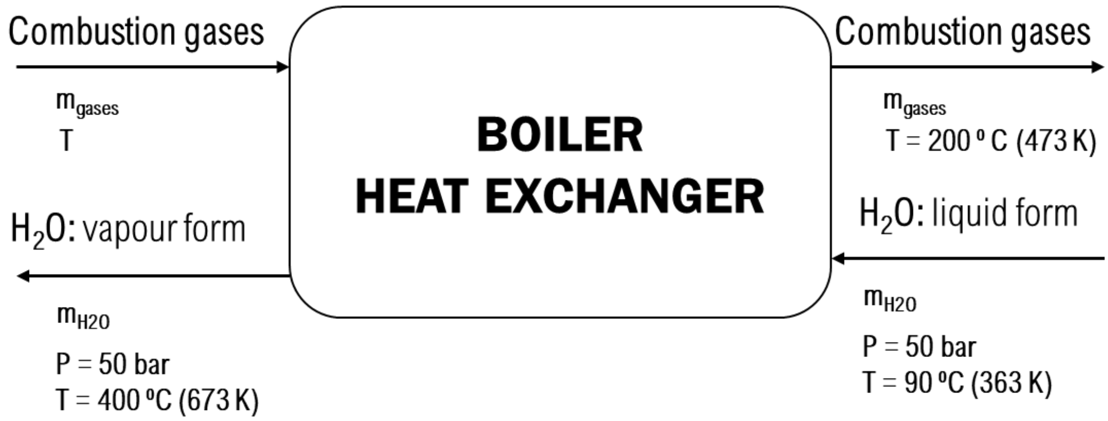


Figure 34. Heat exchanger energy balance diagram

The following assumptions, simplifications and stream properties were adopted for the energy balance:

- Temperature of input combustion gases is the T obtained in Eq 9.
- Assumes a thermal efficiency (η_t) of the process between 80 and 90%.
- Liquid phases were considered to be ideal solutions.
- Thermodynamic properties of liquid and gaseous compounds were obtained from the literature (Perry et al., 1998).

The mass of output water was obtained from Eq. 10, relating the mass fractions and heat capacities of the combustion gases. From this mass, the energy contained and in it is obtained, which after the energy balance will be the energy obtained for the 100 kg of raw SRF input, unit that has been taken as a reference

$$\eta_t \left[m_{gases} \cdot Cp_{gases} \, (T - 473)\right] = m_{H_2O} \cdot \left(h_{H_2O\,(vap)} - h_{H_2O\,(liq)}\right) \qquad \textit{Eq. 10}$$

However, to evaluate the energy recovery of the SRF, it is necessary to consider the drying energy up to the combustion input humidities. For this purpose, an energy balance was carried out in which, through the enthalpy of vaporization of the water, the energy consumption results corresponding to the drying process were obtained. These values were related to the energy production described above, thus arriving at the final results derived from the energy benefit of SRF combustion.

2.3 GASIFICATION

2.3.1 Experimental setup

The gasification experiments were carried out in the laboratories of the Thermochemical Processes Group of the Engineering Research Institute of Aragon (I3A) of the University of Zaragoza. A laboratory-scale fluidized bed reactor was used to gasify SRF from screening waste, at atmospheric pressure. The gasifier, made of AISI 310 refractory steel, was divided in two parts: a bed zone with an inner diameter of 40 mm and a freeboard zone with an inner diameter of 63 mm. Some problems were encountered during feeding because bridging was detected in the silos due to the low density of the material, so it was decided to feed the process with SRF in pellet form. Pellets were fed manually through a double valve feeding system placed at the upper part of the reactor at a constant rate of 1.5 g/min. An electric furnace heated the reactor with three independent heating zones for the bed, freeboard and cyclone. During the tests, the bed temperature was maintained between 750 and 950 °C, while the cyclone temperature was kept constant at 650 °C. During the testing process, calcinated dolomite was utilized as the bed material, sifted at a mesh size of 500 µm.

The diagram of the gasification plant is shown in Figure 35. The gasifying agent was atmospheric air (coming from a compressor), the feed rate of which was adjusted by a mass flow controller to ensure de equivalent ratio required. When necessary, water was also fed through a HPLC pump, which was vaporized (200 ºC) before mixing with the inlet air stream.

During the gasification process, gases were retained inside the reactor for 7 to 8 seconds. After this, small solid particles accompanying the gas as it leaves the reactor were collected by a filtration process through the cyclone and the hot filter, which were maintained at a constant temperature of 650 °C and 450°C, respectively, thus ensuring the only capture of solid particles. The gas and vapours produced during the process were conducted through two condensers (cooled using an ice-bath or a chiller) and a precipitator. A condensed fraction composed of water and organic compounds (tar) was recovered in the condensers. Subsequently, the resulting gas is subjected to an additional filtration process using a cotton filter to remove any particles or aerosols in the gas stream.

Once the flue gas had been purified of particulates, it was measured volumetrically, and then its composition was analyzed online using a gas chromatograph (Agilent 3000-A). This analysis made it possible to determine the volume percentages of the components present in the gas, including H_2, O_2, CO, CO_2, CH_4, C_2H_4, C_2H_6, C_2H_2, H_2S and N_2. The composition of the gas is a consequence of several intricate reactions, among which the most significant ones are presented below:

Oxidation $C + O_2 \leftrightarrow CO_2$ $\Delta H < 0$

Partial oxidation $C + 1/2O_2 \leftrightarrow CO$ $\Delta H < 0$

Boudouard $C + CO_2 \leftrightarrow 2CO$ $\Delta H > 0$

Water - gas primary $C + H_2O \leftrightarrow CO + H_2$ $\Delta H > 0$

Water - gas secondary $C + 2H_2O \leftrightarrow CO_2 + 2H_2$ $\Delta H > 0$

Water - gas shift (WGS) $CO + H_2O \leftrightarrow CO_2 + H_2$ $\Delta H < 0$

Methanation $C + 2H_2 \leftrightarrow CH_4$ $\Delta H < 0$

Steam reforming $C_nH_x + nH_2O \leftrightarrow (x/2 + n)H_2$ $\Delta H > 0$

Dry reforming $C_nH_x + nCO_2 \leftrightarrow 2nCO + (x/2)H_2$ $\Delta H > 0$

Cracking $C_nH_x \leftrightarrow C + (x/2)H_2$ $\Delta H > 0$

Finally, a Karl Fischer titration was performed to determine the amount of water present in the condensed fraction. This, in turn, made it possible to determine by difference the amount of tar present in the gas, whose composition was studied by GC-MS analysis.

It is important to note that the experiments were carried out over 60 minutes to ensure the system to reach a steady state.

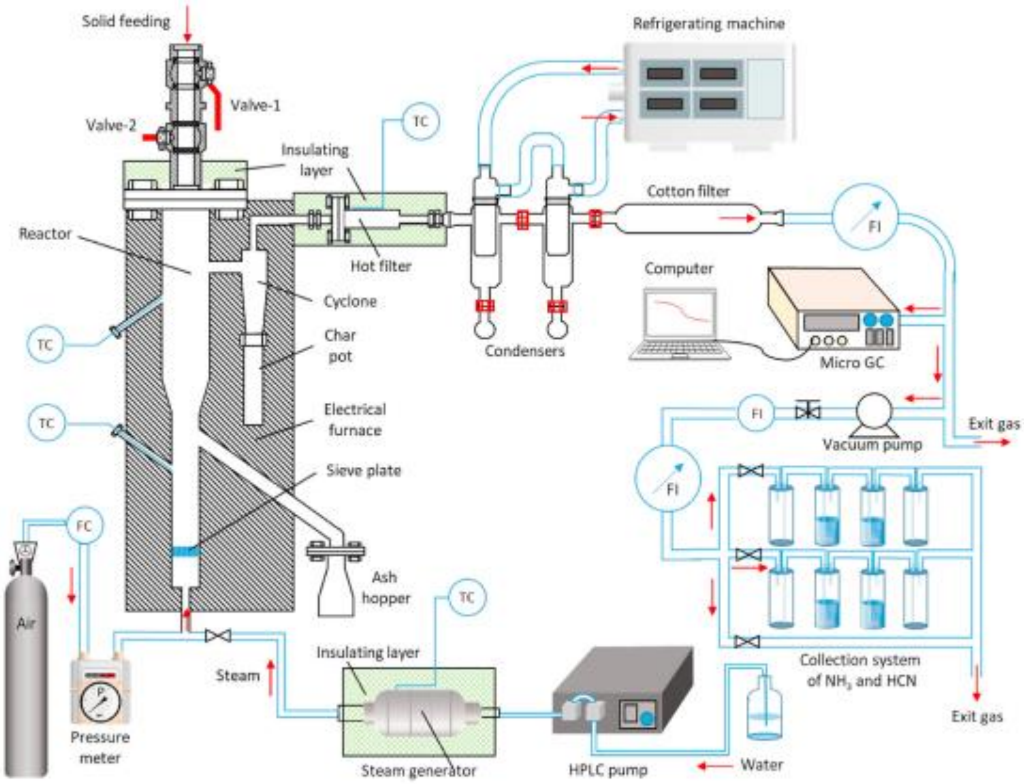


Figure 35. Scheme of the experimental gasification setup

2.3.2 Experimental design and data analysis

Two gasification trials were conducted in order to have a first and rough idea about the influence of the gasifying medium on SRF gasification yields and gas quality. The temperature was set at 800 ºC in both cases. Air was used as gasifying agent in the first case, while a mixture of air and steam was used in the second one. The ratio between the inlet mass of steam and the inlet mass of carbon contained in the SRF (which is denoted as S/C) was virtually 0 in the first case (the moisture of SRF was very low as it was previously dried), while this ratio was increased up to 1 kg steam/kg C in the second experiment by feeding steam to the reactor together with the air flow. Considering the elemental analysis of SRF and its carbon content, this S/C ratio of 1 kg steam/kg C can be converted to a steam/SRF ratio, obtaining a value of 0.51 kg steam/kg SRF. Therefore, these two experiments could be seen as testing the gasification of the dried SRF (test 1) and the wet SRF with a moisture content of 33.7% (test 2).

The amount of air fed to the reactor is represented by the equivalence ratio (ER), which represents the percentual fraction of stoichiometric air really fed to the system. This ER was set according to theoretical calculations based on the assumption that chemical equilibrium could be reached. Hence, the amount of O_2 required for the auto-thermal gasification of SRF was calculated aiming at maintaining the temperature at 800 ºC for a S/C ratio of 0 (test 1) or 1 (test 2). Adding more steam as gasifying agent involves the occurrence of more endothermic reactions (heat demandant), so the requirement of O_2 increases to oxidize a higher fraction of the SRF and release, in turn, more heat able to keep the endothermic reactions.

Table 36 shows the operational conditions of the 2 tests performed. The response variables analyzed were: (i) product distribution (yields of the different gasification products: solid, gas and tar); (ii) gas composition, determined on-line using a gas chromatograph; (iii) lower heating value of the gas product (LHV_{gas}); (iv) cold gasification efficiency; (v) gas phase carbon yield; and (vi) tar composition.

Table 36. Operational conditions in the gasification tests

Test number	1	2
Experiment name	T800_SC0_ER29.58	T800_SC1_ER37.71
Date	23/03/2022	29/04/2022
Temperature (ºC)	800	800
Steam to carbon, S/C (kg/kg)	0	1
ER (%)	29.6	37.7

2.3.3 Energy balance

Based on the data obtained in the experimental gasification design, an energy balance was carried out for the combustion of the gases obtained. This analysis aimed to obtain a result of the potential energy produced from the SRF introduced in the gasification tests, which could be comparable to the energy balance of the SRF combustion. The balance exposed in Figure 36, is similar to that of SRF combustion, with the main unknown being the outgoing gas temperature (T).

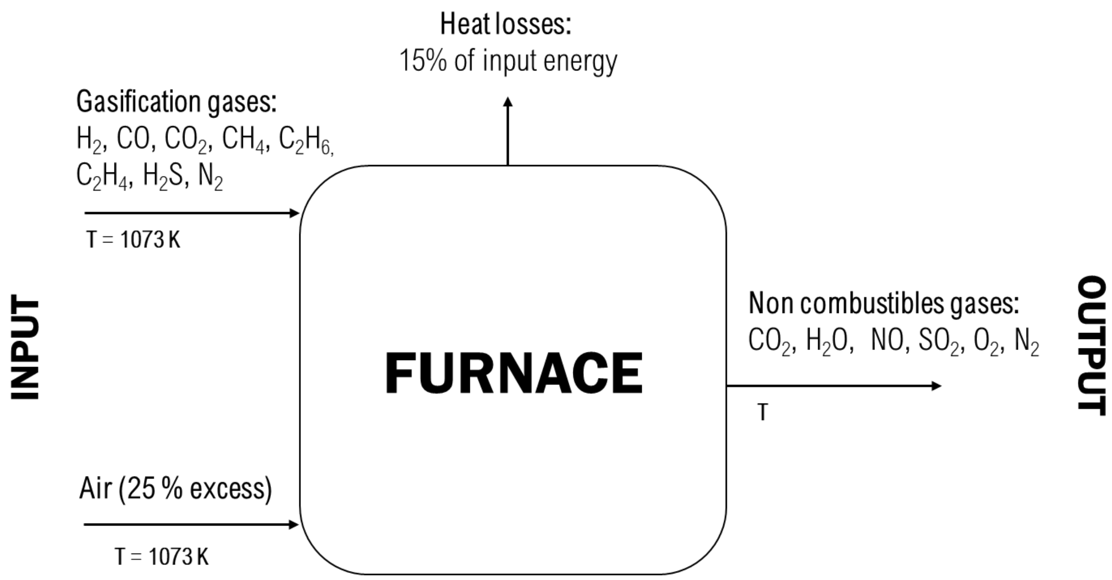


Figure 36. Combustion of gasification gases. Energy balance diagram

The following assumptions, simplifications and stream properties were adopted for the energy balance:

- The input gas temperature was the gasification temperature T_0 = 800 ºC (1073 K).
- The characterization and quantities of the input stream were the outputs of the gasification tests.
- Input air excess considered was of 25%.

- It is considered an ideal case, with perfect and total combustion of the gas, with no unburned fuels.
- The estimation of input energy with energy recovery provided a maximum reference value for the energy efficiency of the combustion processes considered.

Energy assessment

The balance is very similar to the one presented for combustion, with Eq. 11 as the main starting point and having its variables defined in section 2.2.

$$h_{input} = Q_{losses} + h_{output} \qquad \textit{Eq. 11}$$

In this section, the composition of the flue gas is a consequence of several other intricate reactions, the most significant of which are presented below:

$C + O_2 \rightarrow CO_2$

$2H_2 + O_2 \rightarrow 2H_2O$

$CO + {}^{1}/_{2}\, O_2 \rightarrow CO_2$

$CO_2 \rightarrow CO_2$

$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$

$C_2H_6 + {}^{7}/_{2}\, O_2 \rightarrow 2CO_2 + 3H_2O$

$C_2H_4 + 3\, O_2 \rightarrow 2CO_2 + 2H_2O$

$H_2S + {}^{3}/_{2}O_2 \rightarrow SO_2 + H_2O$

Finally, as for the combustion energy balance from the combinations of equations 6, 7 and 8, the T of the output gas could be obtained from Equation 12.

$$T = \frac{h_{input} - |Heat\ losses| - \sum m_i \cdot \Delta h^0_{f,i}}{\sum m_i \cdot Cp_i} \qquad \textit{Eq. 12}$$

The gases obtained follow the same path as in Figure 34, passing through the heat exchanger and generating water vapour at a temperature of 673 K. The result will be the energy production in MJ/kg of raw SRF.

2.4 PYROLYSIS

2.4.1 Experimental setup

The pyrolysis tests were developed in the same laboratory of the Thermochemical Processes Group. The schematic of the experimental pyrolysis plant is shown in Figure 37. The experiments were carried out in a cylindrical fixed-bed reactor, discontinuous for the solid and continuous for the gas. The reactor capacity varies between 2-6 g depending on the density of the material; for the tests conducted in this work the average feeding was 3.7 g. The reactor, made of AISI 310 refractory steel, contains an effective volume of 31,42 cm^3. The reactor was located inside a furnace, which provided the necessary heat to reach the different pyrolysis temperatures. This temperature was controlled by introducing a thermocouple into the bed of the reactor.

The pyrolysis process occured in an inert atmosphere, with a flow rate of approximately 45 mL (STP)/min of N_2, regulated with a mass flow controller, which was fed at the top of the reactor. The vapours generated in the pyrolysis (condensable and non-condensable gases) left the reactor at the bottom towards a condenser. The reactor outlet was heated to 300 °C by an electrical resistance controlled by a thermocouple in order to prevent condensation of the vapours before reaching the condenser area, as well as possible clogging. The condenser was connected to the reactor by means of a metal part for fastening and is placed in a cooling bath of ethylene glycol with a temperature of approximately - 4 °C to facilitate condensation of the vapours. Subsequently, the non-condensed gases pass through a cotton filter to be analyzed in a chromatograph. The chromatograph used is capable of identifying and quantifying H_2, CO, CO_2, CH_4, C_2H_2, C_2H_4, C_2H_6 and H_2S. The composition of the liquid was studied by GC-MS analysis.

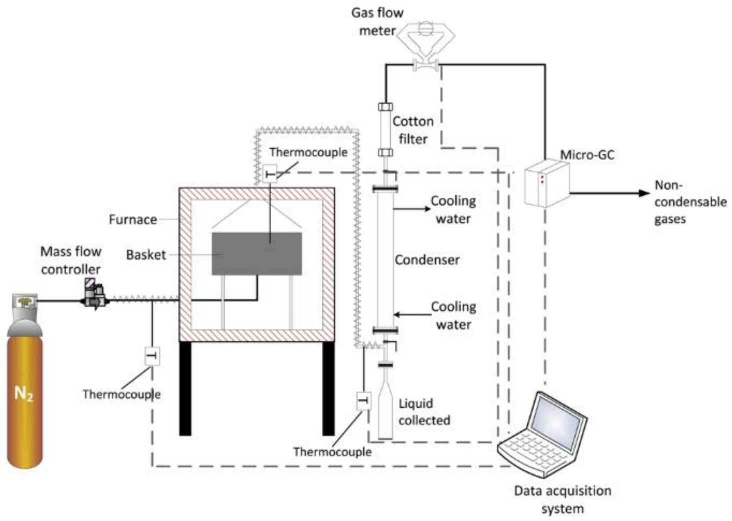


Figure 37. Scheme of the experimental pyrolysis setup

The H_2S adsorption capacity of each char at room temperature and atmospheric pressure has been measured in a fixed-bed device equipped with a mass spectrometer to monitor the effluent gas composition. A synthetic gas containing 1 vol% H_2S in a mixture of N_2 and Ar has been used for testing the performance of char. Around 0.5 g of the char has been placed in a fixed bed adsorption column and a total gas flow of 65 mL (STP)/min passed through it. The adsorption step has been maintained for 3 h up to reach the adsorbent saturation. Later, a desorption step has been performed heating the saturated adsorbent at 150 ºC for 30 min in Ar atmosphere. The content of sulfur remaining in the chars after the H_2S adsorption – desorption cycle was analyzed (elemental analyzer LECO CHN628 with sulfur analyzer module) to quantify the sulfur chemically retained in the char. The adsorption capacity has been calculated for each char by adding the integration of the desorption curve (mL(STP)/min of H_2S leaving the device vs. time) and the sulfur retained in the char (expressed as H_2S) after adsorption-desorption cycle.

The CO_2 adsorption capacities of the char samples at 25 ºC K and atmospheric pressure were determined by thermogravimetric analysis (TGA) with a Netzsch STA 449 Jupiter® thermobalance. About 80 mg of ground char sample was first degassed at 150 °C for one hour in a N_2 stream (100 mLSTP/min). After cooling to 25 ºC, the char sample was exposed to various CO_2/N_2 mixtures, with a CO_2 volume fraction ranging from 5 % to 83 % (relative pressure range of–0.013). The char was exposed to each concentration of CO_2 for 1 h to allow equilibrium between the gas phase and surface adsorption. The CO_2 adsorption capacity of chars (mg CO_2/g char) for each CO_2 partial pressure was calculated from the sample weight gain relative to the weight in a pure N_2 atmosphere.

Once adsorption tests for a preselected set of partial CO_2 pressures were done, the CO_2 flow was replaced by N_2 and the temperature was increased to 150 °C. This leads to CO_2 desorption if the adsorption process is reversible. The mass loss occurring when increasing the temperature indicates how reversible the adsorption process is. Once this test was completed, another set of adsorption measurements was carried out.

2.4.2 Experimental design and data analysis

Four pyrolysis tests were carried out (Table 37), two of them were replicates done at 750 ºC with the non-densified SRF coming from the screening waste (Table 33), and the remaining two were replicates carried out with the sanitary textiles Table 35 at 550 ºC.

The response variables analyzed were: (i) product distribution (yields of the different gasification products: char, gas and liquid); (ii) gas composition, determined on-line using a gas chromatograph; (iii) liquid composition using GS-GM chromatography; (iv) adsorption capacity of the char.

Table 37. Operational conditions in the pyrolysis tests

Test number	1	2	3	4
Input stream	Sanitary textiles	Sanitary textiles	Non densified SRF	Non densified SRF
Data	25/04/2022	28/04/2022	11/05/2022	11/05/2022
Temperature (ºC)	550	550	750	750

3 RESULTS

The results are structured in three parts, each corresponding to a thermochemical process analyzed, combustion, gasification and pyrolysis.

3.1 COMBUSTION

The results from the mass balance for combustion are shown in Table 38, while energy balance is presented in Table 39. Given the high moisture content of the raw SRF and the importance of this factor in the combustion processes (Lin and Ma, 2012), the results are presented according to the percentage of water in the input SRF for the energy balance, having been dried from the initial value of 77.3% to moisture contents of 0, 2.5, 5, 7.5 and 10%. The parameters are calculated based on the mass of raw SRF, which before drying, is 100 kg.

Regarding the energy balance and considering an air excess of 150%, the amount of oxygen and nitrogen required for the complete combustion of the SRF does not vary according to the input moisture, 99.67 kg O_2 and 328.09 kg N_2. The amount of non-combustible flue gas is stated in kg and as can be seen, the only variation concerning the moisture is for the mass of H_2O flue gas, from 15.18 kg for dry SRF to 17.70 kg for 10% moisture. The increased output mass is the same input mass as the input humidities' increase.

The energy balance relates the input and output enthalpy to obtain the final temperature at which the non-combustible gases leave. The highest h_{input}, 590.251 MJ/kg, coincides with the dry SRF, its value being due to the negative enthalpy of the inlet liquid water. Likewise, based on this enthalpy, the heat loss, which is 15%, was calculated. The h_{output} relates the masses of the flue gases, their enthalpy of formation and their heat capacities. The sum of the product between masses and enthalpies shows negative results, being the lowest for 10% moisture due to the increase of H_2O vapour leaving and its lower enthalpy compared to CO_2 and NO leaves. To the sum of the product between masses and heat capacities, the situation is opposite, with the highest result for the test with 10% inlet water, as the Cp of H_2O vapour is the highest among the outgoing compounds. As stated in the methodology, the key parameter for the energy balance of the combustion is the temperature of the flue gas which marks the tendency of the energy efficiency of the process. Thus, it is observed that after relating the variables described above, the highest T with a value of 1269.9 K corresponds to the test with 100% dry SRF.

The combustion exhaust gases pass to the boiler to heat the inlet liquid water and obtain water vapor at 673 K, which will be transformed into energy. The thermal efficiency of the heat exchanger is considered to be 85%. The higher mass of outgoing H_2O vapor, related to the flue gas inlet temperature, also corresponds to the 0% moisture test.

The last part of the table summarizes the results obtained for the combustion energy balance. Based on the above variables, the highest energy production corresponds to the dry SRF. However, the range of values obtained is minimal, between 347.43 MJ for 10% moisture and 348.08 MJ for no water. These values have been related to the drying balance, which gave a range of results between 174.50 and 168.80 MJ per 100 kg SRF with 77.3% for drying the SRF to respective moisture contents between 0 and 10%, respectively. Finally, the energy benefit per 100 kg of raw SRF has been obtained by the difference between the energy production and the energy expenditure derived from the drying of the SRF. Thus, as a final result, it is shown that the highest benefit belongs to the test with 10% moisture in the fuel, obtaining a value of 178.63 MJ. Analyzing the effects of the variables, it can be concluded that the energy consumed in drying is a determining factor in the energy benefit, with more significant differences in its relationship with moisture than those observed in the values for energy production. According to the balance, it can be deduced that the higher the moisture in the SRF, the greater the energy benefit of its combustion. However, problems related to corrosion, the formation of tars and the difficulty in achieving complete combustion (Viklund et al., 2013) lead to a maximum moisture of 10% (Colomer, 2008).

Table 38. Mass balance results of SRF combustion.

Raw SRF moisture (%)			77.3		
Input SRF stream moisture (%)	0	2.5	5	7.5	10
Calculation of necessary input air					
Excess (%)			150		
O_2 (kg)			99.67		
N_2 (kg)			328.09		
Calculation of generated output gases (kg)					
CO_2	43.95	43.95	43.95	43.95	43.95
H_2O	15.18	15.76	16.37	17.02	17.70
NO	1.30	1.30	1.30	1.30	1.30
O_2	59.80	59.80	59.80	59.80	59.80
N_2	328.09	328.09	328.09	328.09	328.09
Total mass	448.32	448.90	449.51	450.16	450.84

Table 39. Energy balance results of SRF combustion

Raw SRF moisture (%)			77.3		
Input SRF stream moisture (%)	0	2.5	5	7.5	10
Energy balance, regarding 100 kg of raw SRF					
hinput (MJ/100 raw kg)	590.251	590.241	590.232	590.221	590.211
Heat losses (MJ/100 raw kg)	88.538	88.536	88.535	88.533	88.532
$\sum \boldsymbol{m_i} \cdot \Delta\mathbf{h}^{\mathbf{0}}_{\boldsymbol{f,i}}$ (MJ)	-0.593	-0.601	-0.609	-0.618	-0.627
$\sum \boldsymbol{m_i} \cdot \boldsymbol{Cp_i}$ (MJ)	0.517	0.518	0.519	0.521	0.522
Output T (K)	1269.9	1267.6	1265.2	1262.6	1259.9

Raw SRF moisture (%)			77.3		
Input SRF stream moisture (%)	0	2.5	5	7.5	10
Heat exchanger, energy balance regarding 100 kg of raw SRF					
Thermal efficiency (%)	85	85	85	85	85
Mass of output H_2O (kg)	123.87	123.82	123.76	123.70	123.64
Summary, regarding 100 kg of raw SRF					
Produced energy (MJ/100 kg raw SRF)	348.08	347.93	347.78	347.61	347.43
Drying energy (MJ/100 kg raw SRF)	174.50	173.20	171.80	170.30	168.80
Energetic Benefit (MJ/100 kg raw SRF)	173.58	174.73	175.98	177.31	178.63

3.2 GASIFICATION

3.2.1 Process performance parameters

The findings resulting from the densified SRF gasification tests are tabulated in Table 40.

The metric referred to as "solid yield" is defined as the percentage of solid mass retrieved after the completion of the gasification process. The tests have demonstrated values of 5.4 and 13.8 %, with the highest value corresponding to the experiment devoid of H_2O addition (S/C ratio of 0) and a lower ER. In alignment with prior literature (Chiang et al., 2013), the tests with similar operating factors have shown that the solid yield is lowered with an increase in ER. The "gas yield," referring to the volume of tar-free gas produced, is also presented in Table 5. The results demonstrate an inverse correlation with the solid yield, with Test 1 having the highest solid yield and the lowest gas yield. The "liquid yield", which refers to the whole fraction of liquid recovered in the condensers, exhibits a similar pattern as the yield to gas, with Test 1 displaying the lowest value and Test 2 the highest one. The addition of steam as gasifying agent in Test 2, as well as the promotion of combustion reactions because of the increased ER, could explain the fact of producing more liquid (water specifically) under these operational conditions. Finally, the "tar yield," which refers to the organic compounds present in the liquid fraction, was reduced when increasing both S/C and ER. Reducing tar formation is a beneficial aspect for subsequent application of the gasification gas.

Table 40 also illustrates the gas production from the gasification process. The results show that the gas production rate increased from 2.3 m^3N gas/kg SRF (Test 1) to 2.9 m^3N gas/kg SRF (Test 2), which means an increase of around 25%. The main reason for this increase could be the higher ER used in Test 2, which also leads to a higher flow of N_2 at the inlet and exit gas (Afailal et al., 2023). Introducing more steam as gasifying agent also promotes gasification reactions (see reactions in section 2.3.1 of this Chapter), thus reducing the remaining solid in favor of the production of gas. The LHV of the gas varied from 2.9 to 4.4 MJ/m^3N. In contrast to the gas product flow rate, the results indicate that the best outcome for the LHV occurred at the lowest ER, corresponding to Test 1, with an ER of 29.6%. As such, the LHV follows a decreasing trend concerning the ER, whose increase promotes complete oxidation reactions, thus generating more CO_2 to the detriment of the other gases.

The cold gasification efficiency, which represents the ratio of the energy content of the produced gas (at room temperature) to the energy content of the solid fuel, was calculated. The cold gasification efficiency values obtained from the gasification tests ranged between 37.8% for Test 2, which had the highest gas production but the lowest LHV, and 44.6% for Test 1 (lower gas production but higher LHV of the gas).

3.2.2 Gas composition

Table 40 displays the process gas composition for each of the conducted tests. In previous studies, for gasification processes applied to biomass, the primary gases generated are H_2, CO, CO_2 and CH_4 (Sánchez et al., 2014). However, for this SRF, the H_2 concentration detected accounted for only 7.9% (Test 1) and 6.8% (Test 2). Although a higher H_2 content in the gas could be expected with increasing presence of steam in the gasification medium (primary water-gas reaction, secondary water-gas reaction, WGS reaction), the parallel increase in ER could be the cause of this lower production by promoting combustion reactions (including H_2 combustion) as well as dilution of the gas components due to the higher presence of N_2. These H_2 content values were very similar to those obtained by previous studies of gasification of SRF obtained from municipal solid waste. Specifically, for a temperature of 800°C and a steam/carbon ratio of 0.85 a range of H_2 content between 5.8 and 11.2% was achieved by Pinto (Pinto et al., 2014).

In relation to CO, it presented significant differences in both tests, with values ranging from 4.1% (Test 2) to 9.0% (Test 1). Other studies previously conducted under similar conditions (Pinto et al., 2014; Recari et al., 2016; Siedlecki and Jong, 2011) are consistent with the value of 9.0 %; however, the value of 4.1 % is substantially lower than the lowest value found in literature of 6.8% (Arena and Di Gregorio, 2014).

Finally, the CO_2 content was the highest among all gases in the two tests (with the exception of N_2), reaching 17.8% for Test 2 and 4.5% (Test 1), which shows a clear improvement of the combustion reactions. Unlike CO, in the case of CO_2, during Test 2, values similar to those compiled in literature were obtained in which the CO_2 content varied between 11.66 and 15.69% (Arena and Di Gregorio, 2014). The CH_4 content decreased from 3.1% in trial 1 to 2.0 % in Test 2, obtaining in both cases values similar to those reported in literature (Arena and Di Gregorio, 2014; Recari et al., 2016).

Table 40. Gasification experimental results

Test number	1	2
Experiment code	T800_S/C0_ER29.6	T800_S/C1_ER37.7
Mass balance (wt. %)	90.7	96.3
Moisture of the input SRF stream to gasification (%)	0	33.7

Test number	1	2
Experiment code	T800_S/C0_ER29.6	T800_S/C1_ER37.7
Product distribution (wt.%) *		
Solid yield	13.7	5.4
Liquid yield	18.4	76.6
Tar yield	3.6	1.4
Gas yield (N_2-free basis)	63.3	57.7
Gas quality parameters		
Gas production (m^3N/kg SRF)	2.3	2.9
Gas LHV (MJ/m3N)	4.4	2.9
Cold gasification efficiency (%)	44.6	37.8
Gas phase carbon yield (%)	80.6	91.2
g tar/m3N gas	15.7	4.8
Gas composition (vol.%, dry basis)		
H_2	7.9	6.8
CO	9.0	4.1
CO_2	13.3	17.8
CH_4	3.1	2.0
C_2H_6	0.2	0.1
C_2H_4	2.0	1.5
H_2S	0.02	0.11
N_2	63.3	67.6
H_2/CO molar ratio	0.874	1.660

3.2.3 Tar composition

The tar produced during the gasification process is one of the critical points of the process (Anis and Zainal, 2011) since it can cause the formation of cracks in the pores of the filters, the production of coke that clogs the filters and the condensation and clogging of cold spots (Corella et al., 1998). For this reason, the composition of the tar generated must be analyzed to know the extent of its impact. For the present SRF gasification process, the tar was analyzed by gas chromatography (GC/MS), identifying the primary compounds shown in Table 41. It is important to emphasize that the results refer to chromatographic area percentages, i.e., these percentage data do not represent the exact composition of the samples since the area/concentration response factor is different for each compound; however, these data would be very useful for the comparison of the two tests performed.

Based on the molecular weight of tar compounds, some researchers divided the composition into five groups (Li and Suzuki, 2009; Ponzio et al., 2006). In this study, a similar classification of tar compounds was employed, including: (i) light aromatics with a single ring, such as styrene; (ii) polycyclic aromatic hydrocarbon (PAH) compounds with two or three rings, including indene, naphthalene, n-methyl-naphthalene, biphenyl, biphenylene, fluorene, anthracene, and phenanthrene; (iii) heterocyclic aromatics containing nitrogen, such as n-methyl-pyridine, benzonitrile, n-methyl-benzonitrile, quinoline, n-methyl-quinoline, indole, n-phenyl-pyridine, n-naphthalenecarbonitrile, benzoquinoline, and 5H-indeno[1,2-b]pyridine; (iv) heterocyclic aromatics containing oxygen, including phenol and benzofuran; and (v) organic compounds containing sulfur, specifically 2-benzothiophene and propanenitrile, 3,3'-thiobis-).

Table 41. Gasification tar composition

Test code	1	2
Experiment code	T800_S/C0_ER29.6	T800_S/C1_ER37.7
Moisture of the input SRF stream to gasification (%)	0	33.7
Compounds group (%)		
Light aromatics with a single ring	1.5	1.7
Polycyclic aromatic	87.2	95.4
Heterocyclic aromatics containing nitrogen	7.6	0.8
Heterocyclic aromatics containing oxygen	1.4	1.5
Organic compounds containing sulfur	1.5	1.7

As seen in the Table 41, the majority percentage of compounds belong to the polycyclic aromatic group for the two experiments, with 87.2% and 95.4%. This difference is directly reflected in the group of nitrogen-containing heterocyclic aromatics, which is lower for trial 2, with 1.5% versus 7.6% for trial 1, a consequence of the difference between the S/C ratio of each trial. The rest of the compound classification groups remain very similar for the two trials, with maximum differences of 0.2%, which are not considered significant. This composition resembles the typical one identified for tars from biomass gasification (Coll et al., 2001).

3.2.4 Energy balance

The energy balance developed is similar to the one carried out for the SRF combustion (Section 3.1), being, in this case, the input stream of the combustible gases coming from the gasification process. Table 42 and Table 43 show the mass and energy balance results, following the same structure as Table 38 and Table 39. The calculations have been performed for the two tests carried out, in which and according to the S/C and ER variables, it is considered that the inlet SRF moisture was 0 and 33.7% for tests 1 and 2, respectively. The parameters were calculated based on the mass of inlet gasification gas, which was calculated from the volume values shown in Table 40. Thus the gas from gasification is mainly composed of N_2 due to the gasifying agent of the gasification process, followed by CO_2 and CO.

The excess of air has been set at 25%, and in this case, and unlike the SRF combustion, the amount of air needed to carry out the combustion of gases varies since the input gas mass is different in experiments 1 and 2. Thus, with less mass of input gas (Test 1), more air would be needed to carry out the reactions, obtaining values of 79.38 g of O_2 and 261.29 g of N_2, compared to 68.41 and 225.19 g in test 2. The mass balance gives the results for the amount of non-combustible flue gases. The total mass output is higher for the test with 33.7% moisture, a repeated pattern for the compounds CO_2, NO and N_2.

About the energy balance, the highest hinput corresponds to the test performed without moisture, with a value of -0.00142 MJ/g of gas compared to -0.00172 MJ/kg for the test with the wet SRF. This difference is mainly due to the amount of CO_2 input, which is considerably higher for test 2, and due to having the lowest enthalpy of formation, it represents this decrease in hinput. As for the gases combustion energy balance, the heat loss in absolute value depends directly on the hinput, being 15%. The sum of the products between the mass of the flue gases and their enthalpy shows lower values for test 2 due to the higher mass of gases and their negative enthalpy values. For the case of the summation between the masses and the calorific capacities, the result is the opposite since the calorific capacities of the exit gases are positive. Finally, and as conclusive data of the combustion energy balance from the rest of the variables, the outlet temperature is higher for test 1, with 1892.9 K, compared to 1665.7 K for test 2. The efficiency of the heat exchanger was also considered to be 85%. The amount of water vapour generated, directly related to the flue gas and its temperature, is higher for test 1, with 1.18 g of vapour H_2O for each g of gasification gas.

The summary of the results obtained for the balance is presented based on 100 kg of raw SRF. Thus, the energy production for the test corresponding to the dry SRF amounts to 214.74 MJ, reaching 190.88 MJ for the test with 33.7% moisture content. In contrast, the energy required to dry the SRF from the initial moisture content of 77.3% to the water content established in the tests would be 174.50 MJ per 100 kg of raw SRF for test 1 and 148.80 MJ for test 2. When analyzing these values, it is observed that the difference between the energy production between both conditions is 23.86 MJ. In contrast, the distance between the drying values would be more significant, 25.70 MJ. Likewise, and based on these differences, it is obtained that the energy spent in drying is more relevant than that produced from the combustion of the gasification gases, obtaining more excellent final energy benefits for the test with 33.7% humidity, with 42.88 MJ compared to 40.24 MJ for the test with dry SRF.

Table 42. Mass balance results of gasification gas combustion.

Test number	1	2
Experiment code	T800_S/C0_ER29.6	T800_S/C1_ER37.7
Moisture of the input SRF stream to gasification (%)	0	33.7
Input Gasification gases (g)		
H_2	1.493	1.626
CO	23.739	13.606
CO_2	55.041	92.581
CH_4	4.587	3.734
C_2H_6	0.479	0.264
C_2H_4	5.181	5.076
H_2S	0.064	0.441
N_2	166.273	223.677
Calculation of necessary input air		
Excess (%)	25	
O_2 (g)	79.38	68.41
N_2 (g)	261.29	225.19
Calculation of generated output gases (g)		
CO_2	122.651	140.956
H_2O	31.317	30.269
NO	0.120	0.830
O_2	15.876	13.682
N_2	427.562	448.862
Total mass	597.527	634.600

Table 43. Energy balance results of of gasification gas combustion

Test number	1	2
Experiment code	T800_S/C0_ER29.6	T800_S/C1_ER37.7
Moisture of the input SRF stream to gasification (%)	0	33.7
Energy balance, regarding mass of input gases		
h_{input} (MJ/g)	-0.00142	-0.00172
Heat losses (MJ/g)	0.000212	0.000258
$\sum \boldsymbol{m_i} \cdot \Delta\mathbf{h}^{\mathbf{0}}_{\boldsymbol{f,i}}$ (MJ)	-1.519	-1.672
$\sum \boldsymbol{m_i} \cdot \boldsymbol{Cp_i}$ (MJ)	0.00069	0.00072
Output T (K)	1892.86	1665.66

Test number	1	2
Heat exchanger, regarding 1 g of input gases		
Thermal efficiency (%)	85	85
Mass of output H_2O (g)	1.18	0.79
Summary, regarding 100 kg of raw SRF		
Produced energy (MJ/100 kg raw SRF)	214.74	190.88
Drying energy (MJ/100 kg raw SRF)	174.50	148.40
Energetic Benefit (MJ/100 kg raw SRF)	40.24	42.48

From these results from the combustion of the gasification gases and those obtained in the energy balance for the direct combustion of the SRF, which were presented in section 3.1, a comparison can be made between the two thermochemical processes. The energy used for SRF drying has been calculated in the same way for both processes, being these calculations applicable up to any level of drying. The energy balance for the direct combustion of the SRF was performed for humidities of 0, 2.5, 5, 7.5 and 10%. In comparison, the combustion of the gasification gases has been analyzed for the SRF at 0 and 33.7% humidity. Starting from these conditions, the most obvious comparison would be the applicability of the processes to dry SRF. Thus the energy benefit per 100 kg of raw SRF would be 173.58 MJ for the combustion of SRF and 40.24 MJ for the combustion of SRF gasification gases, highlighting the suitability of the combustion process from a pure energy point of view. Concerning the incoming moisture, the two processes generate higher energy benefits at higher moisture levels, so it would be necessary to study the maximum permitted water content to achieve the best ratio between the energy produced and the energy used in the drying process.

3.3 Pyrolysis

3.3.1 Product distribution

Table 44 shows the results of the products obtained in the pyrolysis process. The results of the 4 experiments have been grouped and are presented according to the input current and the pyrolysis temperature. The values for ST_T550 are those resulting from the two experiments with sanitary textiles and at 550 ºC, and SRF_T750 for those carried out with SRF and at 750 ºC.

The mass balance for the four trials is above 89%, being higher for SRF trials with an average of 95.1%. The analysis of the yields shows a high liquid production in all four trials, exceeding 50% yield in all of them regardless the temperature and the feedstock. The char yield has a minimal variation for the different tests, obtaining a range between 23.1 and 25.3%. Finally, the gas yield is also hardly affected by the material and/or the temperature, showing a narrow range of percentages.

Table 44. Pyrolysis experimental results

Experiment code		ST_T550	SRF_T750
Yields (%)	Char	23.9±1.1	24.5±1.1
	Liquid	50.7±0.9	54.6±1.4
	Gas	15.9±0.5	16.0±0.7
Mass balance (%)		90.4±1.6	95.1±1.9
Gas composition (vol.%, N_2-free basis)			
	H_2	0.5±0.1	2.5±0.2
	CO	22.2±7.0	26.3±1.1
	CO_2	69.1±6.4	59.3±0.3
	CH_4	2.0±0.6	7.2±0.6
	C_2H_6	1.1±0.1	3.0±0.1
	C_2H_4	4.3±0.3	1.2±0.1
	H_2S	0.8±0.4	0.6±0.05

3.3.2 Characterization of products

Gas composition

The composition of the gas obtained from the pyrolysis of SRF and sanitary textiles is displayed in Table 44. The primary gas compound in all the experiments was CO_2, derived from the high carbon content of the solid input streams (Ruiz-gómez et al., 2017). The pyrolysis of the sanitary textiles produced a higher volumetric fraction of CO_2 than that generated from the SRF. However, the amount of CO was lower. These two gases are the first to emerge during pyrolysis, ranging in temperature between 100 - 200°C for CO_2 and slightly above 300°C for CO. H_2 was five times higher for the SRF tests than for the sanitary textiles, which is justified by the temperature needed to produce it, which is close to the 750 ºC at which the SRF tests are carried out (Santamaria et al., 2021). The percentage of H_2S was very similar for both gases, with 0.8% for sanitary textiles and 0.6% for SRF. The CH_4 content was significantly higher for the 750°C test (with SRF), increasing with increasing temperature, as noted in another study on pyrolysis of SRF from MSW (Tokmurzin et al., 2022).

Char adsorption capacity

H_2S adsorption

Removing H_2S from gas streams obtained in waste recycling or energy recovery processes has been a growing research topic in recent decades (Yildirim et al., 2012). One of the emerging fields of study has been the removal of H_2S in biogas produced from the anaerobic digestion processes of WWTP sludge (Gutiérrez Ortiz et al., 2014). From the pyrolysis of the SRF obtained from the sludge waste, the char produced as a low-cost valuable adsorbent for biogas cleaning is proposed, thus generating the possibility of closing the circle within the treatment plant.

The results on H_2S adsorption are presented in Table 45. The values obtained from the integration of the desorption curve indicate a value of 2.3 mg H_2S desorbed per g of char, representing the physisorbed H_2S. The H_2S retained after the adsorption-desorption cycle represents the chemisorbed H_2S and is 12 mg/g. From the sum of the two values above, the H_2S adsorption capacity is 14.3 mg/g. Thus, sulfur is mainly chemically retained on the char. In the char from SRF, more than 85% of H_2S is chemisorbed, making char regeneration difficult.

The results for the SRF char show lower H_2S adsorption capacity than the chars of lignin (32.0 mg/g), slurry digestate (23.8 mg/g), and cellulose (22.9 mg/g), being higher than that obtained for the soy protein char (3.5 mg/g) (Navarro-gil et al., 2022). At the same time, the 14.3 mg/g obtained are in the same order of magnitude as those resulting from sludge ash, which were obtained in a study by combustion and gasification, and which depending on the conditions, are between 11.8 and 20.8 mg/g (Gil-Lalaguna et al., 2015).

Table 45. H_2S adsorption results.

Material	Char from SRF_T750
H_2S desorbed (mg/g)	2.3
S content before adsorption-desorption cycle (% wt	0.2± 0.1
S content after adsorption-desorption cycle (% wt)	1.338±0.005
S retained (expressed as H2S) (mg/g)	12
H2S adsorption capacity (mg/g)	14.3

CO_2 adsorption

The carbonaceous solids acquired via pyrolysis possess the ability to sequester CO_2 (Mulu et al., 2021), which stands as the primary gas contributing to the greenhouse effect (Côrtes et al., 2019). Among the plethora of techniques available for CO_2 capture, adsorption proves to be highly efficient, primarily owing to its minimal energy requirements, broad operating range encompassing pressure and temperature, and facile regeneration of the adsorbent without generating unfavourable by-products (Dissanayake et al., 2020). Presently, activated carbon remains the predominant adsorbent employed for this purpose. However, in recent times, there has been a growing interest in biochar due to its diverse environmental applications, such as the elimination of emerging pollutants in soil and water (Qiao et al., 2020). Moreover, biochar exhibits several advantages over activated carbon, including lower cost (Gil-Lalaguna et al., 2022), easier regeneration, and reduced energy consumption during production (Meng et al., 2019).

Figure 38 shows the CO_2 isotherm obtained for the pyrolysis char at 25 ºC, showing the relationship between the adsorption capacity and the volume fraction of CO_2 to which the char was exposed.

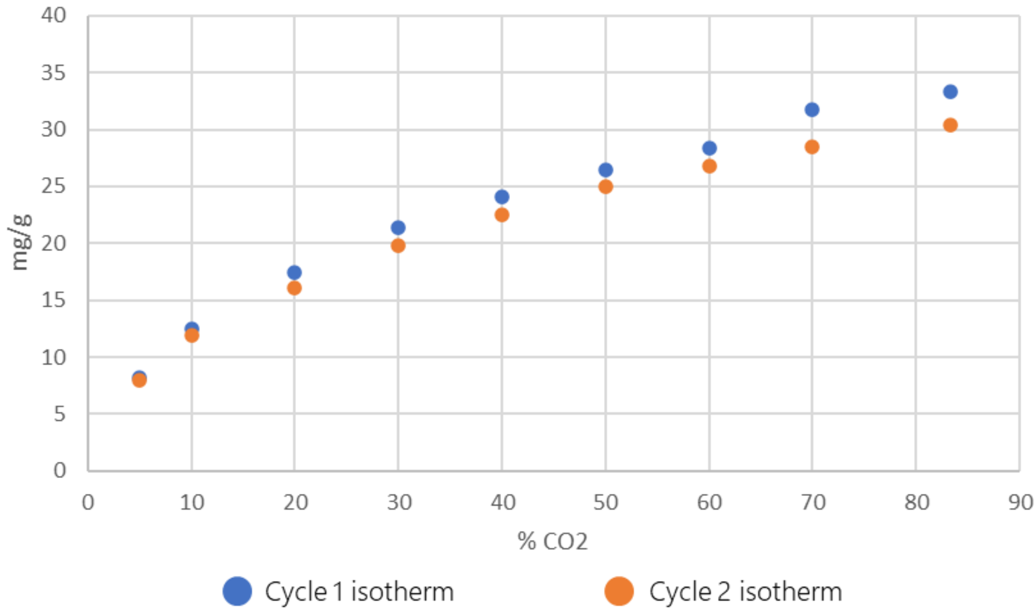


Figure 38. Differences of CO_2 adsorption between cycle 1 and cycle 2

The CO_2 uptake results (obtained at the highest concentration of CO_2 of 83 %) for char after the first and the second adsorption cycle and the amount of CO_2 desorbed (expressed as mg/g) are displayed in Table 46. A slight loss of CO_2 adsorption capacity was observed between cycles #1 and #2. CO_2 uptake expressed as mg CO_2 adsorbed per gram of SRF is calculated considering the char yield.

Table 46. CO_2 adsorption capacity.

Material	Char from SRF_T750
CO_2 uptake cycle 1 (mg/g)	33.4
CO_2 uptake cycle 2 (mg/g)	30.4
CO_2 desorbed after cycle 1 (mg/g)	33.2
CO_2 uptake cycle 1 expressed as mg CO_2/g of SRF	8

These results suggest CO_2 adsorption mechanism occurring in this char is physisorption and in consequence is a reversible process. By comparing with other wastes, such as manure and meat and bone meal, the CO_2 uptake obtained for char from SRF prepared at 750 ºC is around 30 % higher than the ones obtained for manure char and meat and bone meal char prepared at the same pyrolysis temperature (Gil-Lalaguna et al., 2022). It is also higher than that obtained in livestock waste such as slurry or meal, which is around 20 mg/g (Galindo, 2020; López, 2021; Navarro et al., 2021). However, these values are far from the corresponding data for commercially activated carbons, around 80 mg/g of carbon (Gil-Lalaguna et al., 2022) .

Liquid composition

Many of the chemical compounds present in this liquid are of great interest to the industry (Bohutskyi and Bouwer, 2012), and recently numerous studies have focused on the separation and valorization of these compounds (Fonts et al., 2017). In this work and according to the chromatography analyses, the number of compounds in the pyrolysis liquids exceeds 80 different compounds. Given this high number and considering that this liquid's valorization is beyond this thesis chapter's objective, the types and quantity of compounds have yet to be studied. As a note, it has been detected that the pyrolysis liquid of sanitary textiles has as its main component, with 59.21%, benzoic acid that is employed as a preservative additive against moulds and yeasts in the food industry (Straka et al., 2022). In the case of SRF pyrolysis, the primary compound was n-Hexadecanoic acid, with 30.51%. Known as palmitic acid, it has anti-inflammatory properties and could be used as a base for makeup and skin creams (Aparna et al., 2012).

4 CONCLUSIONS

This chapter has analyzed the possible application of thermochemical processes (combustion, gasification and pyrolysis) for the SRF obtained from screening waste from WWTP. After studying the alternatives, it is concluded that:

- SRF valorization would be feasible in the three processes studied. Experimental designs at the laboratory scale of gasification and pyrolysis show the technical feasibility of the processes. Meanwhile, in theoretical terms and through energy balance, SRF combustion and gasification would have positive results for energy production.
- Energetically, combustion, with energy benefits of up to 178.63 MJ per 100 kg of raw SRF, proved to be a more efficient process than gasification, which achieved a maximum benefit of 42.48 MJ. For both cases and within the limits studied, the higher moisture in the input SRF led to increased energy benefit, motivated by the energy expenditure attributed to drying.
- Pyrolysis of the SRF demonstrated the feasibility of obtaining value-added products from the process. The maximum yield for char was 24.5%, while for liquids, it was up to 54.6%. The char obtained, comparable to those obtained in the pyrolysis of sludge, manure, meat, and bone meal, could be used as an alternative H_2S and CO_2 adsorbent for biogas upgrading.

CONCLUSIONES

Las aportaciones más importantes que se han obtenido a partir de este trabajo se han agrupado en varios apartados que se aúnan según los objetivos específicos establecidos en esta memoria y los capítulos que componen su estructura:

- i Revisión bibliográfica sobre la aplicación de tecnologías WtE para los residuos generados en depuradoras.
- ii Producción y caracterización del residuo de desbaste.
- iii Producción y caracterización de CSR a escala de laboratorio.
- iv Viabilidad económica de la producción de CSR.
- v Impacto ambiental de la producción de CSR.
- vi Aplicación de procesos termoquímicos para la valorización del CSR producido.

En relación al análisis de la evolución científica de la aplicación de las tecnologías WtE en los residuos generados de depuradoras:

- El aumento exponencial en el número de publicaciones y la constante evolución de sus palabras clave pone de manifiesto que aún no se ha alcanzado la fase de madurez en el campo de estudio. Las publicaciones en este ámbito de estudio se encuentran principalmente en el área temática de ciencias ambientales. Por tanto, abordarlo desde una perspectiva tecnológica es una oportunidad para afrontar la gestión de las depuradoras en el marco de la sostenibilidad y economía circular.

- La aplicación de tecnologías WtE en las depuradoras se ha centrado en la digestión anaeróbica de lodos, no encontrando referencias que analicen su aplicación a los residuos del desbaste, lo que hace pertinente el desarrollo de esta nueva línea de investigación.

Respecto a la caracterización del residuo del desbaste de depuradora analizados:

- Este residuo, integrado entre otras por diferentes fracciones como textiles sanitarios, plásticos, papel y vegetales, no presenta variabilidad diaria, semanal o estacional. La predominancia de textiles sanitarios con más del 50% sobre el resto de las fracciones permite asimilarlo al rechazo de RSU procedente de plantas TMB.
- Su elevada humedad (77.3%), contenido en materia orgánica (61.6%) y en volátiles (91.0%), así como una adecuada relación C/N (16.67), hacen que procesos de digestión anaeróbica sean una alternativa a considerar frente a la actual eliminación en vertedero.
- De acuerdo a la norma ISO 21640:2021, los valores de PCI en base húmeda (3.59 MJ/kg), contenido en Cl (0.031%) y Hg (3.8 x 10^{-5} mg/MJ), hacen que la producción de CSR sea una alternativa frente a la actual eliminación en vertedero.

En relación a la producción de CSR a escala de laboratorio, caracterización y uso potencial:

- Tomando como referencia una clasificación propia elaborada en base a normas existentes aplicables a otros residuos, la producción de CSR es viable bajo las siguientes condiciones de producción: humedad inferior al 20% para el CSR sin densificar, y humedad inferior al 10% y relaciones de compresión de 6/20, 6/24 y 8/32 para el CSR densificado en forma de pellet.
- Todo el CSR producido puede ser valorizado energéticamente en plantas de producción de energía a partir de residuos. Sin embargo, dado los límites más restrictivos en poder calorífico y contenido en humedad, su valorización

en cementeras y gasificación, está condicionada a las variables de producción, especialmente para la gasificación.

En cuanto a la viabilidad económica de la producción de CSR frente a la actual eliminación en vertedero:

- La actual eliminación en vertedero, con un VAN_s de -1052.60 €/t, no es una opción viable económicamente frente a la producción de CSR cuyo VAN_s ha oscilado entre -56.91 y -39.39 €/t.
- El secado del residuo del desbaste es el proceso más costoso en la producción de CSR, siendo el térmico el más viable económicamente.
- La peletización del CSR supone un incremento de costes que varía entre el 7.88 y 8.48% con respecto al no densificado. Esta diferencia debe ser considerada en su posterior uso a escala real en cuanto los beneficios logísticos atribuidos al CSR densificado en términos de almacenamiento y transporte.

En relación al impacto ambiental de la producción de CSR frente a la actual eliminación en vertedero:

- La eliminación del residuo en vertedero es el escenario con mayor impacto ambiental, presentando valores más altos en 6 de las 11 categorías analizadas.
- Entre las alternativas de producción de CSR analizadas, la producción de CSR sin densificar con secado solar ha presentado menor impacto ambiental, con valores más bajos en 6 de las 11 categorías.

Respecto a la aplicación de procesos termoquímicos para el CSR producido:

- Los procesos WtE de combustión, gasificación y pirólisis son una alternativa técnicamente viable para la valorización del CSR producido.

- La combustión presenta mayor beneficio energético, pudiendo obtener hasta 178.63 MJ frente a los 42.48 MJ de la gasificación por cada 100 kg de CSR en bruto.
- La valorización del CSR mediante pirólisis generó productos de potencial valor añadido, en fase líquida (54.6%) y sólida (24.5%) en forma de char, a valorar en el sector industrial.

CONCLUSIONS

The most significant contributions obtained from this work have been grouped into several sections that are combined according to the specific objectives established in this report and the chapters that make up its structure:

- i Bibliographic review on the application of WtE technologies for wastes generated in WWTPs.
- ii Production and characterization of the screening waste.
- iii Production and characterization of SRF on a laboratory scale.
- iv Economic feasibility of SRF production.
- v Environmental impact of SRF production.
- vi Application of thermochemical processes for the valorization of the SRF produced.

About the analysis of the scientific evolution of the application of WtE technologies in waste generated from WWTPs:

- The exponential increase in the number of publications and the constant evolution of their keywords highlights that the maturity phase in the field of study has yet to be reached. Publications in this field of study are mainly in the thematic area of environmental sciences. Therefore, approaching it from a technological perspective is an opportunity to address wastewater treatment plants' management within the sustainability and circular economy framework.
- The application of WtE technologies in wastewater treatment plants has focused on the anaerobic digestion of sludge, and no references have been

found that analyze their application to screening waste removal, which makes the development of this new line of research pertinent.

Regarding the characterization of the screening waste from the WWTP:

- This waste, integrated by different fractions such as sanitary textiles, plastics, paper and vegetables, does not present daily, weekly or seasonal variability. The predominance of sanitary textiles with more than 50% over the rest of the fractions allows it to be assimilated to MSW rejects from MBT plants.
- Its high moisture content (77.3%), organic matter content (61.6%) and volatile matter content (91.0%), as well as an adequate C/N ratio (16.67), make anaerobic digestion processes an alternative to landfill disposal.
- According to ISO 21640:2021, the values of PCI on a wet basis (3.59 MJ/kg), Cl content (0.031%) and Hg (3.8×10^{-5} mg/MJ) make SRF production an alternative to current landfill disposal.

Concerning SRF production at the laboratory scale, characterization and potential use:

- Taking as a reference an own classification elaborated based on existing standards applicable to other wastes, the production of SRF is feasible under the following production conditions: moisture below 20% for non-densified SRF, and moisture below 10% and compression ratios of 6/20, 6/24 and 8/32 for densified SRF in pellet form.
- All the SRF produced can be energetically recovered in waste-to-energy plants. However, given the more restrictive limits on calorific value and moisture content, its valorization in cement plants and gasification is conditioned by the production variables, especially for gasification.

About the economic viability of SRF production versus current landfill disposal:

- The current landfill disposal, with NPV_S of -1052.60 €/t, is not an economically viable option versus SRF production, whose NPV_S have ranged from -56.91 to -39.39 €/t.
- Drying waste from screening waste is the most costly process in the production of SRF, with thermal drying being the most economically viable.
- Pelletizing the SRF involves a cost increase that varies between 7.88 and 8.48% for the non-densified one. This difference must be considered in its subsequent full-scale use in terms of the logistical benefits of densified SRF in storage and transportation.

To the environmental impact of SRF production versus current landfill disposal:

- Landfill disposal of the waste is the scenario with the highest environmental impact, presenting higher values in 6 of the 11 categories analyzed.
- Among the SRF production alternatives analyzed, the production of SRF without densification with solar drying had the lowest environmental impact, with lower values in 6 of the 11 categories.

Regarding the application of thermochemical processes for the SRF produced:

- The WtE processes of combustion, gasification and pyrolysis are technically viable alternatives for the valorization of the SRF produced.
- Combustion presents a more significant energy benefit, obtaining up to 178.63 MJ compared to 42.48 MJ for gasification per 100 kg of raw SRF.
- The valorization of SRF by pyrolysis generated products of potential added value in liquid (54.6%) and solid (24.5%) phase in the form of char, to be valued in the industrial sector.

LÍNEAS FUTURAS DE INVESTIGACIÓN

Durante el transcurso de este trabajo, han surgido aspectos que requieren de un análisis más exhaustivo, por lo que se proponen como futuras líneas de investigación:

- Diseño de estándares de calidad específicos para los pellets producidos a partir de residuo no agrícola, como los residuos sólidos municipales o el residuo del desbaste, y que tengan en cuenta las particularidades del origen y su posible uso.
- Estudio del proceso de secado para evaluar la eficiencia de los diferentes métodos aplicado al residuo del desbaste para su escalado, como etapa preliminar en la producción de CSR y su posterior valorización.
- Análisis a escala semi industrial de las condiciones óptimas de operación de los procesos termoquímicos, tales como combustión, gasificación y pirólisis, aplicables al CSR.
- Estudio de la implantación de procesos para el aprovechamiento de los productos generados en los procesos de pirólisis y gasificación en las líneas de tratamiento de aguas residuales.
- Análisis de la viabilidad económica y ambiental (Análisis del Ciclo de Vida) de la implantación de procesos termoquímicos para su valorización en depuradoras.

FUTURE LINES OF RESEARCH

During this research, aspects have arisen that require a more exhaustive analysis, so they are proposed as future lines of research:

- Design of specific quality standards for pellets produced from non-agricultural waste, such as municipal solid waste or the screening waste, that consider the origin's particularities and its possible use.
- Study of the drying process to evaluate the efficiency of the different methods applied to the screening waste for its scaling up as a preliminary stage in the production of SRF and its subsequent valorization.
- Semi-industrial scale analysis of the optimal operating conditions of thermochemical processes, such as combustion, gasification and pyrolysis, applicable to SRF.
- Study the implementation of processes for using the products generated in the pyrolysis and gasification processes in the wastewater treatment lines.
- Economic and environmental feasibility analysis(Life Cycle Analysis) of implementing thermochemical processes for valorization in wastewater treatment plants.

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