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Waste-to-Energy in a Circular Economy: Assessing the Energy Recovery Potential and Economic and Environmental Optimal Pathways

Autor:

Ioan-Robert Istrate

Directores:

Dr. José Luis Gálvez Martos

Prof. Dr. Javier Dufour Andía

Programa de Doctorado en Tecnologías Industriales: Química, Ambiental, Energética, Electrónica, Mecánica, y de los Materiales

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Waste-to-Energy in a Circular Economy: Assessing the Energy Recovery Potential and Economic and Environmental Optimal Pathways

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Ioan-Robert Istrate

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Department of Chemical, Energy and Mechanical Technology

Supervisors:

Dr. José Luis Gálvez Martos

Prof. Dr. Javier Dufour Andía

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CERTIFICADO

Dr. José Luis Gálvez Martos, Investigador Titular del Instituto IMDEA Energía y **Prof. Dr. Javier Dufour Andía**, Jefe de la Unidad de Análisis de Sistemas del Instituto IMDEA Energía y Catedrático de Universidad del Departamento de Tecnología Química, Energética y Mecánica de la Universidad Rey Juan Carlos

certifican

que Ioan-Robert Istrate ha realizado bajo su dirección y en las instalaciones del Instituto IMDEA Energía la presente tesis doctoral titulada:

"Waste-to-Energy in a Circular Economy: Assessing the Energy Recovery Potential and Economic and Environmental Optimal Pathways"

La presente tesis ha cumplido con los objetivos planteados, proporcionando resultados innovadores y originales. Por lo tanto, se expresa su conformidad para proceder a la defensa pública de la tesis para optar al grado de Doctor.

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Prof. Dr. Javier Dufour Andía

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Summary

The circular economy continues to gain traction in the political agenda for a competitive and climate-neutral economy. In this context, the prevention, re-use, and recycling of municipal solid waste (MSW) are at the top of the waste policy. Meanwhile, energy recovery from MSW, or waste-to-energy (WTE), seems to have received rather scarce attention despite the dual benefit of avoiding landfilling and providing a domestic energy source. However, an increased circularity of materials will have broad implications on the WTE sector that remain poorly understood.

The goal of this thesis is to assess the energy recovery potential of MSW and the economic and environmental optimal WTE pathways within the context of an increasingly circular economy. To achieve this goal, new tailored tools capable of tackling the complexity of MSW and anticipating system-wide consequences have been developed. First, a material flow analysis (MFA) model is applied to quantify the energy recovery potential of MSW under future scenarios of increased separate collection and recycling. Secondly, the MFA approach is combined with life cycle assessment (LCA) to quantitatively evaluate the potential environmental consequences of phasing-out MSW incineration. Finally, building on the previous tools and mathematical programming, a multi-period optimization framework is developed to determine time-dependent waste treatment capacities and flows according to economic and climate objectives and subject to waste availability and composition, system constraints and restrictions, and policy targets.

Focusing on the case study of Madrid, it has been found that a higher separate collection and recycling of MSW do not compromise the energy recovery potential. This relatively low impact can be attributed to inefficiencies in separate collection and the limited efficiency of materials recovery at automatized sorting facilities. Overall, the future availability and characteristics of feedstocks are found adequate to sustain the operation of large-scale incineration and anaerobic digestion (AD) WTE facilities. Within this context, phasing-out the existing incineration facility in Madrid could reduce some of the environmental impacts, including climate change impacts, human toxicity, and ecotoxicity, at the expense of jeopardizing the realization of the 10% landfill target by 2035. In fact, the optimization results reveal that a more intensive use of incineration will be necessary to achieve this landfill target. Based on the work presented in this thesis, envisioning a future with a relevant role for WTE appears highly likely even within the context of an increasingly circular economy.

Resumen

La economía circular continúa ganando terreno en la agenda política para una economía competitiva y climáticamente neutra. De este modo, la prevención, reutilización y reciclado de residuos sólidos urbanos (RSU) ocupan un lugar destacado en la política de residuos. Mientras tanto, la valorización energética de RSU parece haber recibido una menor atención a pesar de su potencial para reducir la tasa de vertedero y el consumo de combustibles fósiles. Sin embargo, una economía cada vez más circular traerá grandes desafíos para la industria de la valorización energética que aún no se han evaluado en profundidad.

El objetivo de esta tesis es evaluar el potencial de valorización energética de los RSU y las rutas de valorización energética óptimas desde un punto de vista económico y ambiental en el contexto de una economía circular. Para lograr este objetivo, se han desarrollado nuevas herramientas capaces de abordar la creciente complejidad de los RSU y anticipar las consecuencias de grandes cambios en el sistema de gestión de RSU. En primer lugar, se desarrolla un modelo de análisis del flujo de materiales para cuantificar el potencial de valorización energética de los RSU en escenarios futuros de mayor recogida selectiva y reciclaje. En segundo lugar, el enfoque de análisis del flujo de materiales se combina con el análisis de ciclo de vida (ACV) para cuantificar las posibles consecuencias ambientales de la eliminación de la incineración del sistema de gestión de RSU. Finalmente, sobre la base de las herramientas anteriores, se desarrolla un modelo de optimización capaz de determinar las capacidades de tratamiento y los flujos de residuos de acuerdo con objetivos económicos y climáticos y sujeto a la disponibilidad y composición de RSU, las limitaciones y restricciones del sistema y los objetivos establecidos en las políticas de residuos.

Mediante el caso de estudio de Madrid, se concluye que una mayor recogida selectiva y reciclado de RSU no compromete necesariamente el potencial de valorización energética. La futura disponibilidad y las características de los flujos de RSU serían adecuadas para sostener el funcionamiento de grandes instalaciones de incineración y digestión anaeróbica. En este contexto, la eliminación de la incineración del sistema de gestión de RSU de Madrid podría reducir algunos de los impactos ambientales, incluidos los relativos a cambio climático, toxicidad humana y ecotoxicidad, a expensas de poner en peligro la consecución del objetivo del 10% de residuos llevados a vertedero para 2035. Los resultados de la optimización revelaron que será necesario un uso más intensivo de la incineración para lograr este objetivo. Según el trabajo presentado en esta tesis, la visualización de un futuro con un papel relevante para la valorización energética parece muy probable incluso en el contexto de una economía cada vez más circular.

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List of acronyms

AD	Anaerobic Digestion
APC	Air Pollution Control
BAT	Best Available Technique
BAU	Business-as-Usual
BMP	Biochemical Methane Potential
CEPCI	Chemical Engineering Plant Cost Index
CHP	Combined Heat and Power
CRF	Capital Recovery Factor
EU	European Union
GHG	Greenhouse Gas
GDP	Gross Domestic Product
HDPE	High-Density Polyethylene
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDPE	Low-Density Polyethylene
LFG	Landfill Gas
LHV	Lower Heating Value
LP	Linear Programming
MBT	Mechanical-Biological Treatment
MFA	Material Flow Analysis
MINLP	Mixed-Integer Nonlinear Program
MILP	Mixed-Integer Linear Program
MRF	Material Recovery Facility
MSW	Municipal Solid Waste
NLP	Nonlinear Programming
NPV	Net Present Value
PAYT	Pay-As-You-Throw
PET	Polyethylene Terephthalate
PP	Polypropylene
PPI	Industrial Producer Price Index
RDF	Refuse-Derived Fuel
SCR	Selective Catalytic Reduction
TRL	Technology Readiness Level
VS	Volatile Solids
WTE	Waste-to-Energy

Chapter 0

Resumen en castellano

0.1 Introducción

La dependencia de unos recursos naturales cada vez más escasos junto con el deterioro del medio ambiente ha puesto de relieve la necesidad de un cambio de paradigma en el modelo económico (Ellen MacArthur Foundation, 2015). En este contexto, la economía circular sigue ganando terreno en la agenda política para alcanzar una economía competitiva y climáticamente neutra (European Commission, 2020). La economía circular tiene por objetivo alinear el crecimiento económico con las limitaciones medioambientales a través del uso eficiente de los recursos disponibles (McDowall et al., 2017). En una economía circular, el valor de los recursos se retiene dentro del sistema económico durante el mayor tiempo posible a través de la prevención, reutilización, reciclaje y recuperación de residuos.

Los países de la Unión Europea (UE) generan más de 226 millones de toneladas de residuos sólidos urbanos (RSU) al año, lo que equivale a 505 kg per cápita al año (Eurostat, 2020). Los RSU consisten en una mezcla heterogénea de residuos alimentarios, papel, cartón, plástico, metal, vidrio, textiles, madera y otros materiales. Aprovechar estos residuos para producir nuevos materiales y energía representa una oportunidad para proporcionar recursos locales a la industria, reducir las importaciones de materias primas y, en definitiva, acelerar la transición hacia una economía más circular y menos dependiente de los combustibles fósiles (European Commission, 2011). En este contexto, la prevención, reutilización y reciclaje de RSU ocupan un lugar destacado en la política de residuos de la UE. Por ejemplo, la Directiva 2018/851 obliga a los Estados Miembros a preparar para reutilización y reciclaje al menos el 55 % de los RSU para 2025, 60% para 2030 y 65% para 2035.

Mientras tanto, la valorización energética de RSU parece haber recibido una menor atención en la agenda política (Malinauskaite et al., 2017). La valorización energética abarca una amplia gama de procesos termoquímicos (incineración, gasificación y pirólisis) y bioquímicos (p. ej., digestión anaerobia) capaces de explotar el contenido energético de los residuos. La incineración y la digestión anaerobia son las rutas de mayor desarrollo e implantación en el mercado, mientras que procesos como gasificación y pirólisis se encuentran todavía en una etapa temprana de desarrollo (Foster et al., 2021; Mukherjee et al., 2020). El interés en la valorización energética es cada vez más prominente debido a una mayor generación de RSU y los posibles riesgos ambientales que surgen de su disposición en vertederos (UNEP, 2015). En este sentido, la valorización energética ofrece el doble beneficio de (1) prevenir el uso de vertederos y los posibles riesgos ambientales y para la salud humana y (2) proporcionar una fuente de energía doméstica con el potencial de reducir el consumo de combustibles fósiles y las emisiones de gases de efecto invernadero (World Energy Council, 2016).

La transición hacia una economía cada vez más circular requiere estrategias efectivas para prevenir la generación de RSU, aumentar su reutilización y reciclaje y minimizar el uso de vertederos. La realización de estos objetivos trae grandes desafíos para el sector de la valorización energética que aún no se han evaluado en profundidad (European Commission, 2017). En primero lugar, es fácil imaginar que la recogida selectiva obligatoria y los objetivos de reciclaje tendrán un impacto en el potencial de valorización energética de los RSU. Por ejemplo, el aumento del reciclaje de materiales combustibles como papel, cartón, plástico y textiles puede reducir el contenido energético de los residuos que se destinan a incineración (Saveyn et al., 2016). En segundo lugar, la idoneidad de la incineración en el contexto de una economía circular sigue siendo muy controvertida. Un número creciente de organizaciones y grupos consideran la eliminación de la incineración como una medida urgente para proteger el medio ambiente y la salud humana, y promover el desarrollo de negocios circulares (GAIA, 2013; Zero Waste Europe, 2022). Sin embargo, los beneficios ambientales de eliminar esta tecnología del sistema de gestión de RSU siguen siendo muy inciertos, considerando que tal decisión podría intensificar el uso de vertederos. Por último, los posibles cambios en los flujos de residuos debido a una mayor circularidad afectarán a la valorización energética en todos los niveles, desde el despliegue y uso de tecnologías hasta su viabilidad económica y ambiental. En este sentido, las rutas de valorización energética deben planificarse cuidadosamente considerando la disponibilidad y las características de los RSU para evitar malas decisiones y garantizar el mejor rendimiento ambiental (European Commission, 2017).

0.2 Objetivos

Esta tesis está motivada por la necesidad de llevar a cabo un análisis en profundidad de los desafíos que afronta la valorización energética de RSU en el contexto de una economía circular. El objetivo es evaluar el potencial de valorización energética de los RSU y las rutas de valorización energética óptimas desde un punto de vista económico y ambiental en el contexto de una economía circular. Este objetivo se logrará respondiendo a las siguientes preguntas de investigación (PI):

PI1 ¿Cuál será el potencial de valorización energética de los RSU en escenarios futuros de mayor recogida selectiva y reciclaje?

PI2 ¿Cuáles son las posibles consecuencias ambientales de la eliminación de la incineración del sistema de gestión de RSU?

PI3 ¿Cuáles son las rutas de valorización energetica óptimas desde el punto de vista económico y/o ambiental que podrían priorizarse en el futuro dada la disponibilidad y características de los RSU?

Dar respuestas científicamente sólidas a estas preguntas requiere grandes esfuerzos de modelización. En este sentido, en esta tesis se han desarrollado nuevas herramientas capaces de abordar la creciente complejidad de los RSU y anticipar las consecuencias de grandes cambios en el sistema de gestión. En primer lugar, se desarrolló un modelo de análisis del flujo de materiales para cuantificar el potencial energético (**PI1**). En segundo lugar, el modelo de análisis del flujo de materiales se integró con un modelo de análisis de ciclo de vida (ACV) para cuantificar los impactos y beneficios ambientales de la gestión de RSU (**PI2**). Por ultimo, sobre la base de las herramientas anteriores, se desarrolló un modelo de optimización multiperiodo que determina las capacidades de tratamiento óptimas y el flujo de residuos de acuerdo con objetivos económicos y climáticos, y sujeto a la disponibilidad y composición de los residuos, las limitaciones y restricciones del sistema y los objetivos de las políticas de residuos. (**PI3**).

0.3 Metodología

0.3.1 Análisis de flujo de materiales

El análisis del flujo de materiales es una metodología para cuantificar de manera sistemática los flujos y acumulaciones de materiales y/o sustancias en un sistema (Brunner and Rechberger, 2017). El análisis del flujo de materiales sigue la ley de conservación de la masa: las cantidades de entrada son iguales a las cantidades de salida más las acumulaciones. Esta metodología se ha aplicado ampliamente a la gestión de RSU con distintos objetivos: cuantificar indicadores de desempeño como la tasa de reciclaje (Turner et al., 2016), trazar el flujo de sustancias clave como el carbono, los nutrientes y los metales pesados (Allesch and Brunner, 2017; Jensen et al., 2017) y realizar balances energéticos (Tonini et al., 2014). La mayoría de estos trabajos abordan situaciones pasadas y/o actuales (Elgie et al., 2021; Vujovic et al., 2020). Mientras tanto, la combinación del análisis del flujo de materiales con el análisis de escenarios se ha propuesto como un enfoque adecuado para predecir cambios futuros en los flujos de residuos (Allesch and Brunner, 2017; Eriksen et al., 2020; Klotz et al., 2022). La aplicación prospectiva del análisis del flujo de materiales ha sido limitada hasta la fecha, aunque este enfoque resulta particularmente adecuado para cuantificar el potencial futuro de valorización energética de RSU (**PI1**).

0.3.2 Análisis de ciclo de vida

Los potenciales beneficios ambientales surgen como el principal motor del creciente interés en la valorización energética de RSU (UNEP, 2015). La valorización energética puede proporcionar beneficios ambientales mediante (1) la reducción de la cantidad de residuos que se depositan en vertederos y (2) la recuperación de energía que puede sustituir a la producción con combustibles fósiles y así evitar las emisiones asociadas. No obstante, la valorización energética puede causar impactos ambientales debido a las emisiones generadas por el propio proceso, la gestión de los rechazos y otros residuos y el consumo de recursos (p. ej., electricidad y productos químicos). Es, por lo tanto, de suma importancia seguir una perspectiva holística del ciclo de vida al evaluar y comparar escenarios alternativos de valorización energética (Manfredi and Pant, 2011) (**PI2**). En este sentido, el análisis de ciclo de vida (ACV) surge como la metodología más adecuada para cuantificar los impactos ambientales asociados a la gestión de RSU (Finnveden et al., 2007).

El ACV es una metodología estandarizada (ISO, 2006a, 2006b) para evaluar los impactos ambientales potenciales de productos, sistemas y servicios durante todo su ciclo de vida, desde la extracción de materias primas hasta el final de su vida útil (Hellweg and Canals, 2014). La metodología del ACV consta de cuatro fases (European Commission, 2010):

(1) Definición de objetivo y alcance. En esta fase, se deben definir aspectos clave como el objetivo del estudio, el contexto, la aplicación prevista y la audiencia objetivo. La definición del alcance incluye la descripción del sistema analizado, los límites del sistema, la unidad funcional, la asignación de cargas ambientales, las categorías de impacto evaluadas, los métodos de evaluación de impacto, requisitos de datos, suposiciones y limitaciones.

(2) Análisis de inventario. En la segunda fase, se desarolla el inventario del ciclo de vida (ICV), recopilando las entradas y salidas relevantes para cada proceso dentro de los límites del sistema.

(3) Evaluación de impacto. Durante esta fase, los flujos de entrada y salida desde y hacia el medio natural incluidos en el ICV se agregan en categorías de impacto mediante el uso de factores de caracterización.

(4) Interpretación. En la última fase, los resultados se interpretan y discuten en relación con el objetivo y el alcance del estudio.

El ACV es, con mucho, la metodología más popular a la hora de apoyar la toma de decisiones en el sector de la gestión de residuos (Allesch and Brunner, 2014; Cobo et al., 2018a; Pires et al., 2011). En las últimas dos décadas se han desarrollado varias herramientas específicas para la realización de ACV de sistemas de gestión de RSU (Anshassi and Townsend, 2021; Gentil et al., 2010). Estudios previos han demostrado que el desempeño ambiental de los procesos de tratamiento y disposición de RSU (p. ej., incineración, digestión anaerobia o vertedero) puede variar significativamente dependiendo de la formulación de los escenarios, la composición de los residuos, la tecnología, el sistema energético, las condiciones climáticas y las suposiciones metodológicas (Bernstad and Jansen, 2012; Cleary, 2009; Laurent et al., 2014a). La composición y las propiedades fisicoquímicas (p. ej., contenido en energía, carbono, nutrientes y metales pesados) de los residuos se reconocen a menudo como los factores más críticos (Bisinella et al., 2017; Lausselet et al., 2016). En este sentido, la capacidad de conectar el inventario de los procesos de tratamiento de residuos (es decir, emisiones, consumo de recursos y materiales y recuperación de energía) con la composición y propiedades fisicoquímicas de los residuos ha sido reconocida como la piedra angular de cualquier herramienta de ACV de residuos (Gentil et al., 2010).

Otro factor determinante es la fuente de energía sustituida a través de la valorización energética de residuos (Fruergaard et al., 2009). En la práctica, la suposición más común es que la electricidad producida a partir de residuos sustituye una cantidad equivalente de electricidad producida por el mix eléctrico del país analizado (Anshassi et al., 2021). Por lo tanto, los impactos evitados pueden variar sustancialmente y tienden a disminuir en países con altas proporciones de energías renovables. En la UE, por ejemplo, la huella de carbono del mix eléctrico oscila entre 0,765 kg CO_2/kWh en Grecia y 0,033 kg CO_2/kWh en Suecia (Scarlat et al., 2022).

0.3.3 Optimización matemática

El análisis del flujo de materiales y el ACV son herramientas descriptivas que permiten evaluar y comparar escenarios de gestión de residuos. Los escenarios analizados se deben definir a priori y su número está generalmente limitado entre 3 y 24 (Tascione and Raggi, 2012). Sin embargo, los sistemas integrados de gestión de RSU incluyen numerosos flujos y procesos dando lugar a varios cientos de escenarios posibles. Además, las restricciones del sistema (p. ej., limitaciones de capacidad), los objetivos políticos (p. ej., objetivos de vertedero) y la dinámica temporal de la generación y composición de los residuos hacen que la evaluación de todos los escenarios posibles sea inviable. Dentro de este contexto, la optimización matemática surge como una herramienta que permite evaluar simultáneamente todas las vías posibles de tratamiento de residuos para determinar la mejor solución de acuerdo con uno o varios criterios y, al mismo tiempo, satisfacer las restricciones y limitaciones del sistema (**PI3**).

La optimización matemática consiste en (1) desarrollar un modelo matemático que describa el sistema analizado y (2) utilizar un algoritmo para determinar la solución óptima del problema (Vanderbei, 2017). El modelo consiste en una función objetivo (o, en el caso de la optimización multi-objetivo, varias funciones objetivo) y una serie de restricciones que imponen límites a las variables de decisión. La solución óptima corresponde a los valores de las variables de decisión (p. ej., capacidades de las plantas de tratamiento) que minimizan o maximizan la función objetivo (p. ej., el coste del sistema o el impacto ambiental) a la vez que satisfacen las restricciones del sistema. Un modelo de optimización puede ser mono-objetivo o multi-objectivo dependiendo del del número de funciones objetivo. Dependiendo de la naturaleza de las expresiones matemáticas, los modelos de optimización se pueden clasificar como programación lineal o programación no lineal. La programación lineal implica que todas las variables son continuas y todas las ecuaciones son lineales. Por otro lado, estaríamos hablando de programación no lineal si todas las variables son continuas pero la función objetivo y/o al menos una restricción es una función no lineal. Si el modelo involucra tanto variables continuas como discretas, se puede clasificar como programación lineal (o no lineal) entera mixta.

Los modelos no lineales pueden ser convexos o no convexos. Si las restricciones son funciones convexas y la función objetivo es convexa si se minimiza o cóncava si se maximiza, forman un conjunto convexo. Un conjunto convexo implica que la combinación lineal de dos puntos a y b cualesquiera pertenecen al conjunto. Los modelos convexos tienes una única solución que, además, es el óptimo global. Por el contrario, si la función objetivo o cualquiera de las restricciones son funciones no convexas, forman un conjunto no convexo. Los modelos de optimización no convexos pueden tener múltiples soluciones locales, lo que hace que la determinación de la solución óptima global sea una tarea difícil (Floudas, 2000).

A lo largo del tiempo, se han desarrollado numerosos modelos de optimización de sistemas de gestion de RSU abarcando una amplia gama de objetivos y alcances (Ghiani et al., 2014; Juul et al., 2013). La atención se ha centrado principalmente en determinar el sistema de menor coste (Garibay-Rodriguez et al., 2018; Rizwan et al., 2018; Shahid and Hittinger, 2021), aunque se está prestando cada vez mayor atención a los objetivos ambientales a través de la integración del ACV (Cobo et al., 2019; Mavrotas et al., 2013). Algunos modelos también incorporaron un análisis multiperiodo para abordar la dinámica de parámetros clave, como la generación de residuos (Batur et al., 2020; Roberts et al., 2018), la capacidad del vertedero (Garibay-Rodriguez et al., 2018; Santibañez-Aguilar et al., 2017) o el impacto de la descarbonización del sector eléctrico (Levis et al., 2013).

Los modelos de optimización presentados en la literatura han facilitado la identificación de rutas hacia sistemas de gestión de RSU más sostenibles (Cobo et al., 2018a). Sin embargo, el futuro presenta desafíos que requieren nuevos desarrollos. En primer lugar, la mayoría de los modelos se han centrado únicamente en nueva infraestructura sin tener en cuenta la posibilidad de extender la vida útil o mejorar las instalaciones existentes (Ng et al., 2014; Shahid and Hittinger, 2021; Tan et al., 2014). En segundo lugar, se ha prestado poca atención al efecto de la economía de escala. Por último, al igual que en el caso del ACV, una limitación recurrente de los modelos de optimización es que el desempeño tecno-económico y ambiental de los procesos de tratamiento no está conectado con la composición de los residuos. Los modelos existentes suelen aplicar rendimientos fijos (p. ej., kWh de electricidad por tonelada de residuo) y factores de emisión (p. ej., kg de CO_2 por tonelada de residuo), ignorando así la dependencia de la composición de los residuos (Chang et al., 2012; Ooi et al., 2021; Yousefloo and Babazadeh, 2020). Esto se debe en gran medida a que la introducción de la composición de los residuos como una variable del modelo daría lugar a un modelo no lineal no convexo, lo cuál aumenta significativamente la complejidad del problema.

0.3.4 Caso de estudio de Madrid

Madrid es un caso de estudio interesante para abordar los objetivos planteados en esta tesis. Madrid es un ejemplo representativo de una ciudad grande, muy densamente poblada y con un sistema de gestión de RSU moderno y diverso, pero que todavía presenta bajas tasas de recogida selectiva y reciclaje y una alta tasa de vertedero. En 2019, el municipio de Madrid trató aproximadamente 1,4 milliones de toneladas de RSU (Madrid City Council, 2019). Con una población de 3,2 millones, esta cifra corresponde a una tasa de generación de 434 kg per cápita al año. Solo el 30% de los RSU generados por los hogares madrileños en 2019 se recogieron selectivamente, mientras que el 70% restante fue recogido en la fracción “restos”. El porcentaje de RSU enviado a reciclaje, digestión anaerobia y compostaje se ha mantenido casi constante en un 35% durante el período 2015-2019. Mientras, la tasa de incineración aumentó ligeramente del 21% al 24% y la tasa de vertedero disminuyó del 44% al 42%.

Madrid es un caso de estudio interesante por diferentes razones. En primer lugar, Madrid necesita un esfuerzo considerable para mejorar la recogida selectiva y el reciclaje, y reducir la tasa de vertedero para cumplir los objetivos de la política de residuos de la UE. Esto brinda una excelente oportunidad para evaluar los posibles impactos de la consecución de estos objetivos en la valorización energética. En segundo lugar, la incineración es una parte fundamental del sistema de gestión de RSU de Madrid. Durante el periodo 2015-2019, se incineraron una media de 300.000 t/año de rechazos producidos por las plantas de separación de las fracciones restos y envases. Sin embargo, la incineración ha encontrado una fuerte oposición pública debido a la percepción de riesgo para la salud humana. En este contexto, una nueva estrategia de residuos publicada en 2018 planteaba la eliminación de la incineración para

2025. Aunque esta nueva estrategia fue declarada inválida en 2019, el cierre de la incineradora sigue en el debate público. De este modo, Madrid ofrece un caso de estudio real para evaluar las consecuencias ambientales de la eliminación de la incineración del sistema de gestión de RSU. Por ultimo, la incineradora de Madrid se inauguró en 1993 y está próxima al final de su vida útil. Además, la recogida selectiva de materia orgánica se implantó en 2017 y se espera que la digestion anaerobia adquiera todavía una mayor relevancia. Este contexto proporciona la base para abordar las rutas de valorización energetica óptimas que podrían priorizarse en el futuro, incluyendo cuestiones tales como la sustitución o desmantelamiento de la incineradora actual o la ampliación de la capacidad de digestión anaerobia. Además, el mix eléctrico español está ampliamente dominado por energías renovables, que se espera siga aumentando en el futuro, lo que afectará significativamente los beneficios ambientales de la valorización energética.

0.4 Resultados

0.4.1 Evaluación del potencial de valorización energética de RSU en escenarios futuros

<u>Introducción</u>

El desempeño económico y ambiental de las tecnologías de valorización energética de RSU es altamente sensible a la disponibilidad y características de los residuos. La disponibilidad de residuos determina, además, la ubicación de las instalaciones, la capacidad de tratamiento y el diseño de la tecnología, entre otros parámetros. Resulta, por tanto, esencial entender en qué medida una mayor circularidad de los materiales (a través de la recogida selectiva y el reciclaje) podría afectar el potencial de valorización energética. En este apartado, se resume el trabajo realizado en esta tesis para cuantificar el potencial de valorización energética de RSU en escenarios futuros (**PI1**).

<u>Metodología</u>

Para llevar a cabo este análisis, se desarrolló un modelo matemático que simula el flujo de materiales y sustancias a través del sistema de gestión de RSU. El modelo calcula las cantidades y las propiedades fisicoquímicas (p. ej, contenido energético) de los flujos de residuos a partir de su composición material. Para ello, cada uno de los flujos del sistema se ha desagregado en los siguientes 18 materiales: materia orgánica, restos de jardín y poda, celulosa (p. ej., papel de cocina y pañales), papel, cartón, tereftalato de polietileno, polietileno de alta densidad, polietileno de baja densidad, polipropileno, otros envases de plástico (p. ej., poliestireno y cloruro de polivinilo), plásticos que no sean de envases (p.

ej., juguetes), cartones para bebidas y similares, vidrio, metales ferrosos, metals no ferrosos (aluminio), textiles, madera y otros. Cada material está definido por un conjunto de propiedades biológicas y fisicoquímicas (p. ej., contenido de humedad, poder calorífico, contenido de sólidos volátiles o potencial de metano). El modelo traza el flujo de cada uno de estos materiales a través del sistema, desde su generación hasta la recolección y el tratamiento. La cantidad y las propiedades de cualquier flujo en cualquier punto del sistema se pueden evaluar a partir de la cantidad de materiales contenidos en dicho flujo y las propiedades específicas de cada material. De este modo, el modelo es capaz de predecir las consecuencias de cambios estructurales en el sistema como, por ejemplo, un aumento de la recogida selectiva de materia orgánica.

El modelo desarrollado se ha aplicado al caso de estudio de Madrid para comparar el potencial de valorización energética de los RSU en el año 2019 (escenario de referencia) con el de un conjunto de escenarios plausibles para los años 2030 y 2040. Los flujos de residuos susceptibles de ser valorizados energéticamente dependen de la configuración del sistema de gestión. La configuración actual del sistema se simula en el escenario de referencia. En este escenario, se identifican los siguientes flujos de residuos susceptibles de ser valorizados energéticamente: fracción orgánica separada en origen, fracción orgánica residual (materia orgánica separada de la fracción restos en las plantas de separación), rechazos de las plantas de separación restos, rechazos de las plantas de separación de envases, rechazos del pretratamiento de las plantas de digestion anaerobia, fracción restos de comercios y residuos de limpiezas viarias y zonas verdes (Figura 0-1).

Para los años 2030 y 2040, se han evaluado una serie de escenarios que describen configuraciones alternativas del sistema:

- *Business-as-Usual* (BAU). Este escenario simula la configuración actual del sistema para los años 2030 y 2040. Dado que el sistema se mantiene inalterado, cualquier variación en el potencial de valorización energética estará asociada exclusivamente a cambios en la generación y la recogida selectiva de los RSU.
- Mejora de las plantas de separación (I-MRF): Este escenario simula la sustitución de las plantas de separación existentes por nuevas plantas más eficientes. Por lo tanto, este escenario permite evaluar el efecto conjunto de mejorar la eficiencia de la recogida selectiva y la recuperación de materiales en las plantas de separación.
- Combustible Derivado de Residuos (CDR): Este escenario simula la implementación de la producción de CDR a partir de la fracción restos de hogares y comercios. El CDR se produce mediante la separación mecánica de aquellos materiales con un alto contenido de energía (p. ej., plásticos y papel).

- Sin plantas de separación (No-MRF): Este escenario simula un sistema sin plantas de separación para la fracción restos. Por lo tanto, la fracción restos representa de por sí un flujo susceptible de ser valorizada energéticamente.

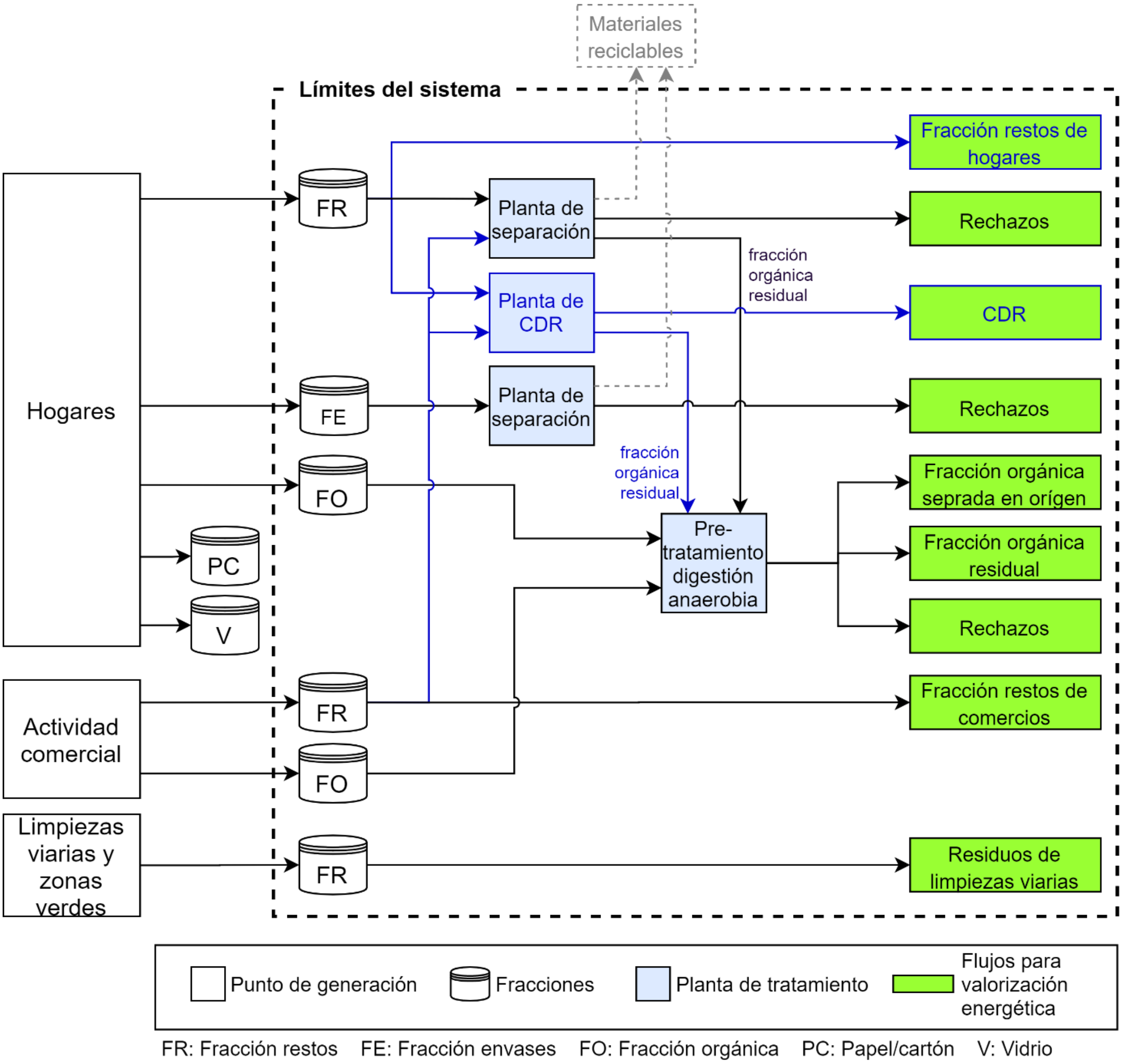

Figura 0-1. Flujos de residuos susceptibles de ser valorizados energéticamente en el sistema de gestión de residuos sólidos urbanos de Madrid. Las líneas y los recuadros de color negro representan los flujos existentes en Madrid en el escenario de referencia 2019. Las líneas y los recuadros de color azul representan flujos alternativos investigados para los años 2030 y 2040. Las líneas discontinuas representan materiales reciclables que quedan fuera de los límites del sistema. CDR: combustible derivado de residuos.

Resultados y discusión

La Tabla 0-1 muestra las cantidades anuales de RSU recogidas en Madrid en los escenarios analizados para los años 2019, 2030 y 2040. En el escenario de referencia 2019, el 25% de los RSU provienen de la separación en origen (354.328 t), mientras que cerca del 75% se recogen en la fracción restos (1,04 millones de toneladas). Los escenarios futuros analizados suponen que la tasa de recogida selectiva alcanza un 39% para 2030 y un 46% para 2040. A pesar de esta mejora, la fracción restos alcanza 849.047 t/año en 2030 (61% de los RSU) y 748.095 t/año en 2040 (54% de los RSU). Estas cantidades relativamente altas se deben principalmente a que no todos los materiales son objeto de la recogida selectiva. En el caso de Madrid, se estableció que la celulosa, los plásticos que no sean de envases, los residuos textiles, la madera y la categoría otros no sean separados en origen. Estos materiales suman 421.343 t/año en 2030 y 392.765 t/año en 2040, lo cuál implica que entre el 35% y el 37% de los RSU son materiales que no están incluidos en el sistema de recogida selectiva. Estos resultados revelan las limitaciones de la recogida selectiva y, al mismo tiempo, muestran un gran potencial para la valorización energética.

Tabla 0-1. Cantidades anuales de residuos sólidos urbanos recogidas en Madrid en 2019, 2030 y 2040.

Fracción	2019	2030	2040
Hogares (t/año)			
Restos	802.346	605.825	497.477
Envases	96.231	106.313	109.546
Papel/cartón	73.475	150.802	188.685
Vidrio	58.809	80.755	93.612
Orgánica	110.327	188.773	236.195
Comercios (t/año)			
Restos	100.653	101.863	104.961
Orgánica	15.487	18.167	18.720
Limpiezas viarias y de zonas verdes (t/año)			
Restos	136.777	141.359	145.658
Tasa de recogida selectiva (% RSU)			
Sólo hogares	30%	47%	56%
Todos	25%	39%	46%

La Figura 0-2a muestra las cantidades anuales de flujos de residuos susceptibles de ser valorizados energéticamente en cada escenario. Estos flujos representan aproximadamente el 86% de la cantidad total de RSU en el escenario de referencia 2019 (alrededor de 1,20 millones de toneladas). Los flujos de residuos considerados adecuados para conversion termoquímica (p. ej., rechazos de las plantas de separación) alcanzan 909.072 t/año (65% de los RSU), mientras que aquellos considerados adecuados para digestion anaerobia (es decir, fracción orgánica) alcanzan 291.899 t/año (21% de los RSU). Los escenarios para los años 2030 y 2040 muestran una ligera disminución en las cantidades anuales de residuos disponibles para valorización energética. Los escenarios BAU sugieren que la mejora de la recogida selectiva reduciría la disponibilidad en solo el 8% para 2030 y un 12% para 2040 con respecto a 2019. La mejora de la recogida selectiva junto con la implementación de plantas de separación más eficientes (escenarios I-MRF) reduciría la disponibilidad en un 14% para 2030 y un 17% para 2040. Por otro lado, la implementación de la producción de CDR (escenarios CDR) y la eliminación de las plantas de separación (escenarios No-MRF) tienen un impacto todavía menor, con reducciones que van desde el 10% al 12% para 2040.

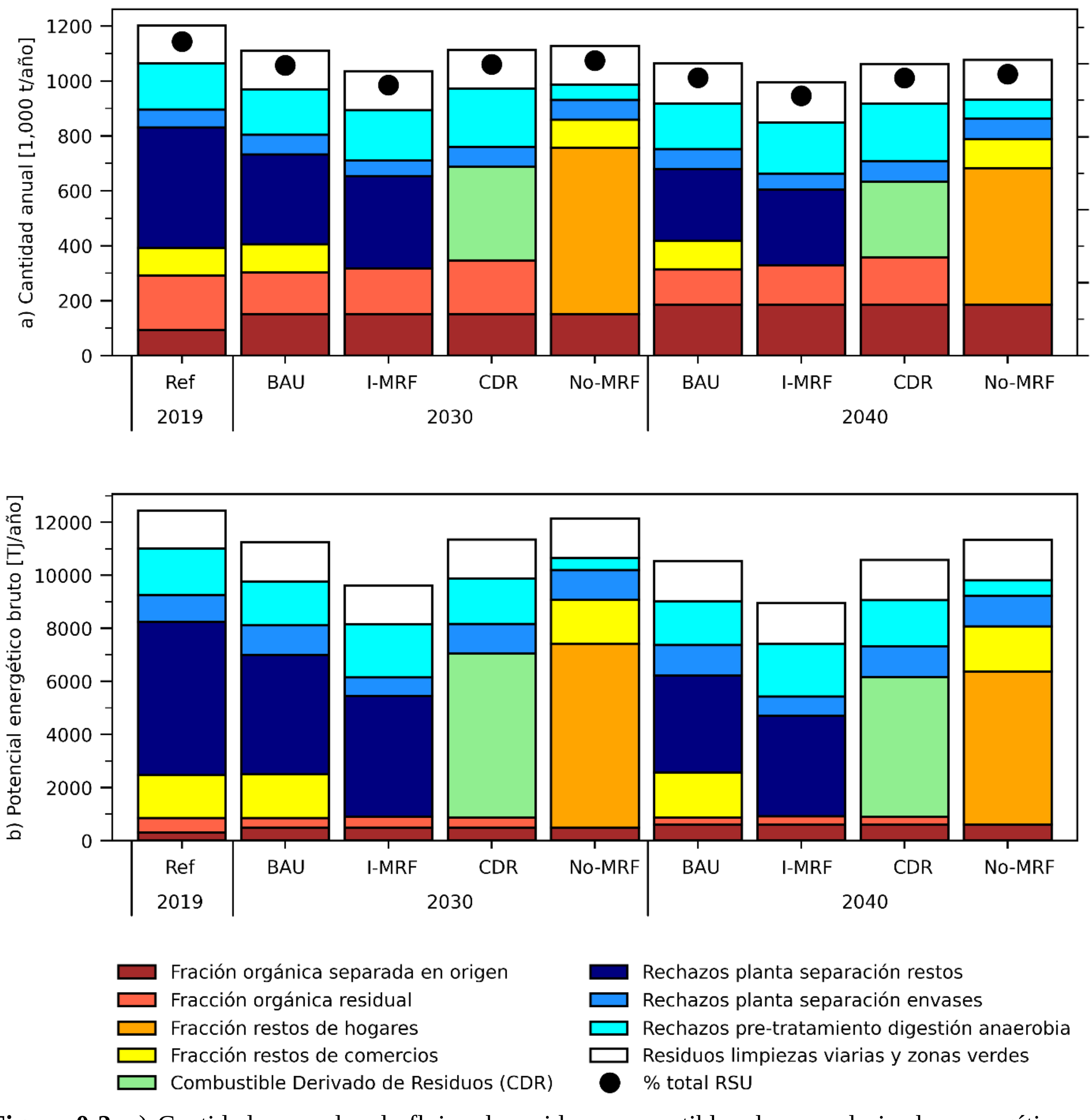

Figura 0-2. a) Cantidades anuales de flujos de residuos susceptibles de ser valorizados energéticamente en Madrid en cada escenario. b) Potencial energético bruto de los flujos de residuos en cada escenario.

El potencial energético bruto alcanza 12.436 TJ/año en el escenario de referencia 2019 (Figura 0-2b). Alrededor del 55% de este potencial está contenido en los rechazos de las plantas de separación (6.794 TJ/año), el 14% en los rechazos del pretratamiento de la digestión anaerobia (1.755 TJ/año), el 13% en la fracción restos de comercios (1.610 TJ/año) y el 11% en los residuos de limpiezas viarias y zonas verdes (1.426 TJ/año). El potencial de valorización energética mediante conversión termoquímica (p. ej., incineración) alcanza 11.585 TJ/año (93% del potencial total), mientras que el potencial de valorización mediante digestión anaerobia apenas alcanza los 852 TJ/año.

Centrándonos en los escenarios futuros, el potencial energético bruto se sitúa entre 9.621 y 11.346 TJ/año en 2030 y entre 8.942 y 11.327 TJ/año en 2040. Los escenarios BAU sugieren que la mejora de la recogida selectiva reduce el potencial energético un 10% en 2030 y un 15% en 2040, principalmente debido a un mayor reciclaje de materiales combustibles (es decir, papel, cartón, plástico y textil). La mayor reducción se observa en los escenarios I-MRF (23% en 2030 y 28% en 2040), donde la implementación de plantas de separación más eficientes aumenta aún más el reciclaje de materiales combustibles. Por el contrario, la menor reducción se observa en los escenarios No-MRF (2% en 2030 y 9% en 2040). En este escenario, la fracción restos de hogares y comercios no se envía a plantas de separación y, por lo tanto, todos los materiales combustibles permanecen en los flujos de residuos disponibles para valorización energética. Finalmente, los escenarios CDR muestran un potencial similar a los escenarios BAU e inferior a los escenarios No-MRF.

En base a los resultados obtenidos, se concluye que un aumento de la recogida selectiva y el reciclaje de RSU no necesariamente compromete el potencial de valorización energética (**PI1**). Esto se explica en gran medida por las ineficiencias en la recogida selectiva y las plantas de separación. En primer lugar, existe una tasa máxima factible de recogida selectiva. En este trabajo, estas tasas máximas se fijaron en 85% para envases, 85% para papel/cartón y 60% para textil. Además, algunos flujos de residuos recogidos selectivamente, como la fracción envases, pueden contener altos niveles de impurezas, lo que genera altas tasas de rechazos en las plantas de separación. En segundo lugar, no todos los materiales están sujetos a recogida selectiva. En Madrid, por ejemplo, el plástico que no sea envase no debe depositarse junto con la fracción envases. Por último, la eficiencia de la recuperación de materiales en las plantas de separación también es limitada.

0.4.2 Evaluación ambiental de la eliminación de la incineración del sistema de gestión de RSU

Introducción

La incineración es una parte fundamental de los sistemas integrados de gestión de RSU. Actualmente se incinera alrededor del 27% de los RSU generados en la UE (Eurostat, 2020). Sin embargo, la percepción de riesgo para la salud humana ha contribuido a consolidar una fuerte oposición pública a la incineración en muchos lugares (Huang et al., 2015; Liu et al., 2021; Subiza-Pérez et al., 2020), incluido Madrid. En este contexto, la eliminación de la incineración continúa ganando terreno en el debate público (GAIA, 2013; Zero Waste Europe, 2022). En este apartado, se presenta el trabajo realizado para evaluar las consecuencias ambientales potenciales de la eliminación de la incineración del sistema de gestión de RSU de Madrid (**PI2**).

Metodología

Para realizar este análisis, el modelo de análisis del flujo de materiales presentado en el apartado anterior se integró con un modelo de ACV para cuantificar los impactos y beneficios ambientales de la gestión de RSU. Siguiendo la perspectiva de ciclo de vida, este modelo aborda el sistema de gestión en conjunto con los sectores económicos con los que interacciona. De este modo, los límites del sistema están formados por las siguientes etapas: (1) tratamiento/disposición de RSU (p. ej., plantas de separación, compostaje, digestion anaerobia, incineración y vertedero), (2) procesamiento de los productos recuperados por el sistema (p.ej., reciclaje, aplicación en suelo de compost o combustión de biometano), (3) producción de las materias primas, energía y servicios consumidos por el sistema (p.e., electricidad, productos químicos o transporte) y (4) sustitución de producción primaria (Figura 0-3).

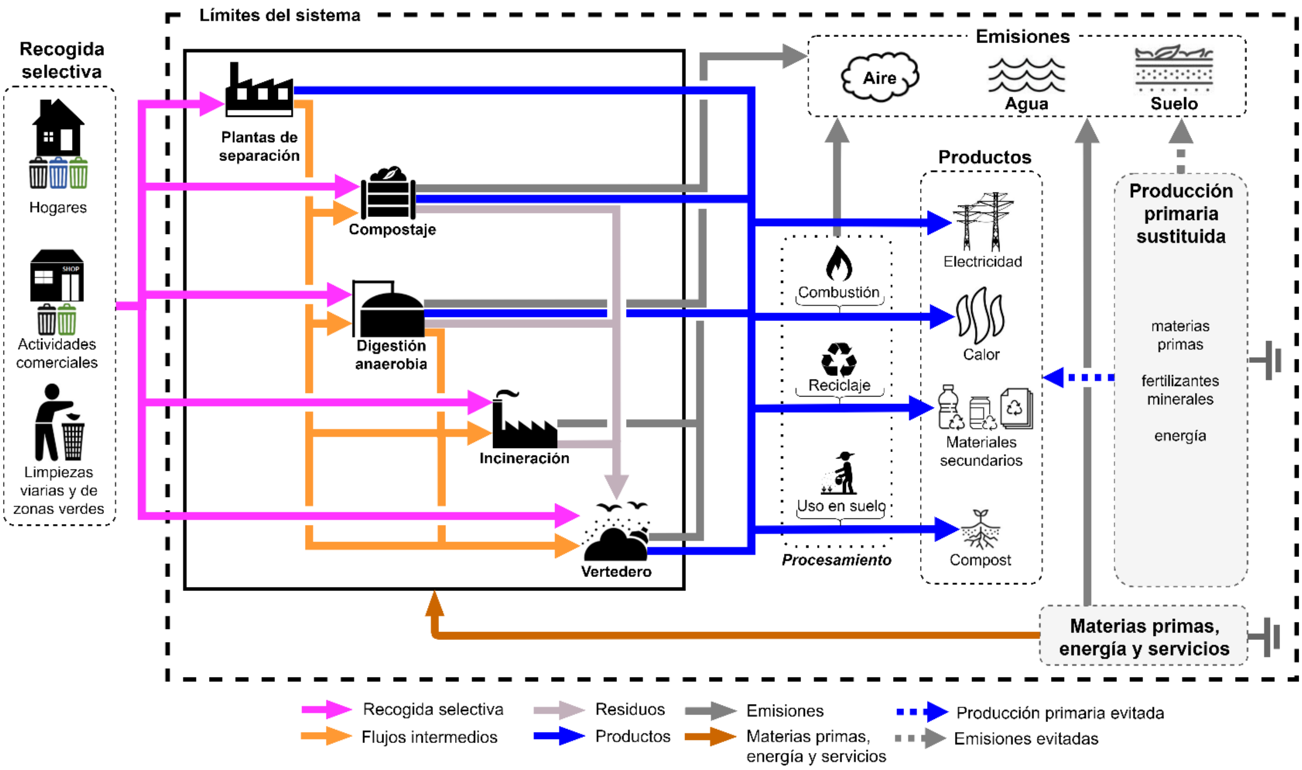

Figura 0-3. Límites del sistema de gestión de residuos sólidos urbanos.

Es importante mencionar que el sistema de gestión de RSU es un sistema multifuncional ya que, además de proporcionar el servicio de gestión de residuos, el sistema también produce materiales (p. ej., papel y plásticos reciclados), fertilizantes orgánicos (compost) y energía (p. ej., electricidad y biometano). Dado que el objetivo es evaluar las emisiones e impactos específicos de la gestión, debe resolverse el problema de la multifuncionalidad. En este trabajo, se ha adoptado el enfoque de la sustitución (también conocido como "enfoque de impactos evitados" o "créditos ambientales"). De esta forma, los materiales y la energía producidos a partir de RSU sustituyen a otros productos funcionalmente equivalentes, mientras que el sistema de gestión recibe créditos por los impactos evitados (European Commission, 2010; Heijungs et al., 2021). Por ejemplo, se ha supuesto que la electricidad recuperada a partir de RSU sustituye una cantidad equivalente de electricidad generada por el mix nacional. Las emisiones asociadas a la electricidad sustituida se evitan y el sistema de gestión de RSU recibe créditos por ello (Astrup et al., 2015; Laurent et al., 2014b; Viau et al., 2020). Por lo tanto, el impacto neto del sistema de gestión de RSU se calcula como la diferencia entre los impactos generados y los impactos evitados durante el ciclo de vida del sistema.

Los inventarios de los procesos de tratamiento de residuos se han modelado en base al análisis de flujo de materiales. Las emisiones generadas por los procesos de tratamiento se han modelado a partir de las propiedades fisicoquímicas de los 18 materiales contenidos en los flujos de residuos. Por ejemplo, las emisiones de CO_2 de origen fósil y biogénico generadas por la incineración se han modelado a partir del contenido en carbono fósil y biogénico de los materiales de entrada. De este modo, cualquier cambio en la composición de los flujos de residuos se ve reflejado en los inventarios de emisiones y, por lo tanto, en los impactos ambientales. Los inventarios de los sectores económicos que interaccionan con el sistema de gestión de RSU (p. ej. producción de materias primas y energía) se han obtenido de la base de datos Ecoinvent v3.7.1 (Wernet et al., 2016).

El modelo de ACV desarrollado se ha aplicado para cuantificar las consecuencias ambientales potenciales de la eliminación de la incineración del sistema de gestión de RSU de Madrid. Siguiendo la filosofía del análisis de escenarios del apartado anterior, se comparó el desempeño ambiental del escenario de referencia para el año 2019 con el de una serie de escenarios plausibles que describen el sistema de gestión de RSU en 2025, 2030 y 2040, con y sin incineración. Los escenarios futuros con incineración se evaluaron considerando tanto la incineradora actual como una nueva incineradora de mayor eficiencia energética (la nueva incineradora tiene una eficiencia eléctrica neta de 26%, mientras que la actual alcanza un 12%). Para la evaluación de impacto se utilizó el método Environmental Footprint (EF) 2.0 recomendado por la Comisión Europea (Fazio et al., 2018). Este método incluye 16 categorías de impacto, de las cuales 13 se identificaron como relevantes para el tratamiento de RSU: cambio climático, acidificación, eutrofización de agua dulce, eutrofización terrestre, eutrofización marina, formación de ozono fotoquímico, agotamiento del ozono, formación de partículas, toxicidad humana con efectos cancerígenos, toxicidad humana sin efectos cancerígenos, ecotoxicidad, agotamiento de energía no renovable y agotamiento de minerales y metales (Laurent et al., 2014b; Mayer et al., 2019).

Resultados y discusión

La Tabla 0-2 muestra los indicadores de desempeño del sistema de gestión de RSU de Madrid en el escenario de referencia 2019 y los escenarios futuros con y sin incineración. Como se puede observar, la tasa de vertedero en el escenario 2019 alcanza el 51% de los RSU (aproximadamente 705.666 t). Los escenarios futuros sugieren que la eliminación de la incineración aumentaría la tasa de vertedero hasta el 57% para 2040 (aproximadamente 796.295 t), a pesar de la mejora en la recogida selectiva y el aumento del reciclaje y los tratamientos biológicos. La tasa de vertedero se podría reducir hasta el 36% para 2040 si se mantiene la tasa actual de incineración (aproximadamente el 22%).

Tabla 0-2. Indicadores de desempeño del sistema de gestión de residuos sólidos urbanos de Madrid en 2019 (escenario de referencia) y 2025, 2030 y 2040 con y sin incineración.

Indicator		**Sin incineración**			**Con incineración**		
	2019	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**
Tasa de recogida selectiva	25	33	39	46	33	39	46
Tasa de reciclaje	20	26	31	39	26	31	40
Tasa de incineración	22	–	–	–	22	22	22
Tasa de vertedero	51	65	61	57	43	39	36

Para facilitar la lectura, en este resumen se discuten únicamente los resultados para las cuatro categorías de impacto más relevantes para la incineración: cambio climático, toxicidad humana (con y sin efectos cancerígenos) y ecotoxicidad (Figura 0-4).

El impacto neto en la categoría de cambio climático del sistema de gestión de RSU de Madrid en el escenario de referencia 2019 es de 391.151 t CO_2 eq. (Figura 0-4a). Esto corresponde a una huella de carbono de 343 kg CO_2 eq./t RSU. Los principales contribuyentes a los impactos generados son el vertedero (277.570 t CO_2 eq.) y la incineración (188.306 t CO_2 eq.), mientras que el reciclaje evita un total de 86.340 t CO_2 eq. Los otros procesos (plantas de separación, compostaje, digestión anaerobia, y utilización de compost) tienen contribuciones insignificantes. El impacto neto en los escenarios futuros que contemplan la continuación de la incineradora disminuye gradualmente hasta 256.942 t CO_2 eq. para 2040 (34% reducción con respecto a 2019). Esta reducción se debe mayoritariamente al aumento de la tasa de reciclaje. La sustitución de la planta actual por una nueva planta de mayor eficiencia reduciría el impacto en un 17-18%. Por el contrario, la eliminación de la incineración permitiría reducir los impactos en cambio climático hasta 78.456 t CO_2 eq. para 2040 (69% de reducción con respecto a 2019).

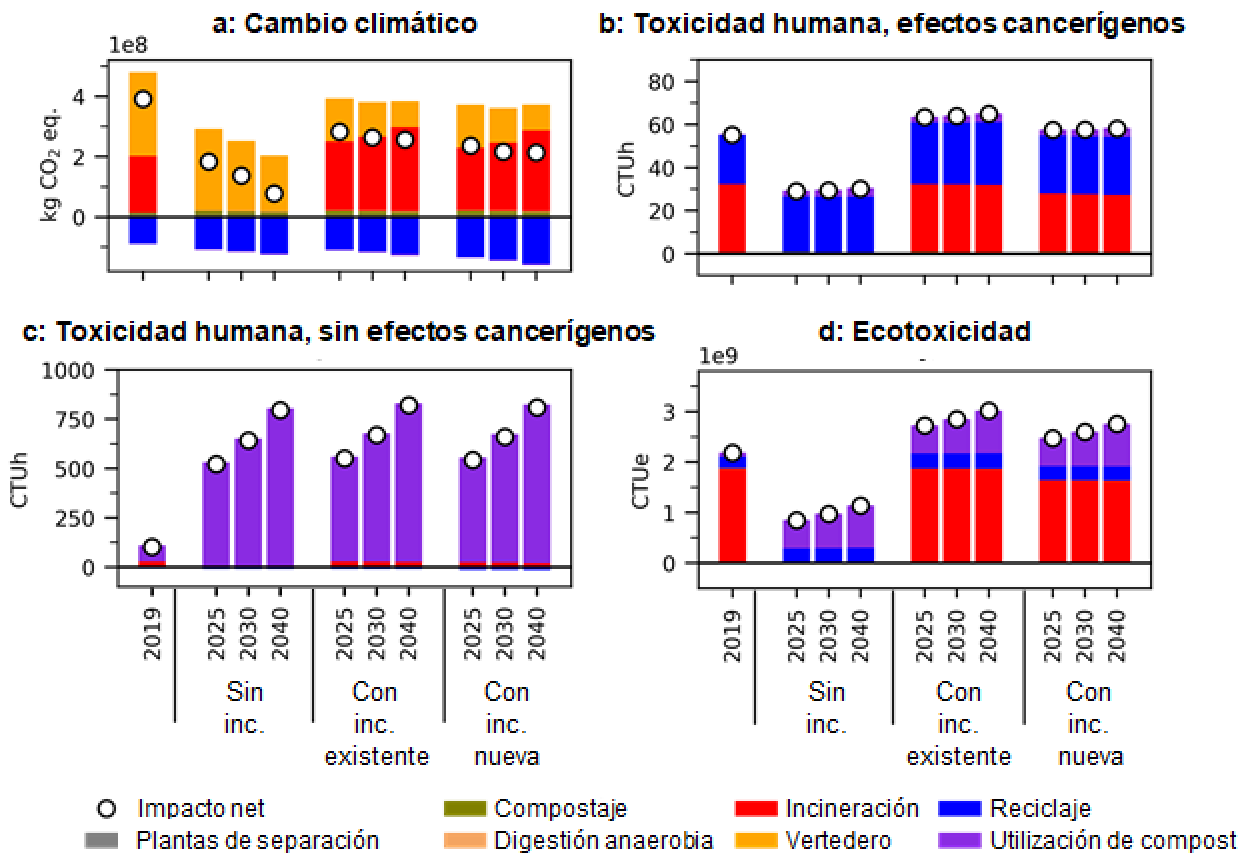

Figura 0-4. Evaluación del impacto de ciclo de vida del sistema de gestión de residuos sólidos urbanos (RSU) de Madrid en 2019 (escenario de referencia), 2025, 2030 y 2040. Los escenarios futuros se han evaluado con y sin incineración (abreviado como "inc." en la figura). Los valores positivos representan impactos generados por el sistema, mientras que los valores negativos representan impactos evitados. La unidad funcional es el tratamiento de la cantidad de RSU generada en el año estudiado.

Vale la pena señalar que la incineración tiene un impacto neto positivo en cambio climático en todos los escenarios estudiados. Esto significa que las emisiones de gases de efecto invernadero evitadas gracias a la sustitución del mix eléctrico no compensan las emisiones producidas por la quema de materiales de origen fósil, principalmente plásticos. Esto se debe en gran medida a la baja huella de carbono del mix eléctrico español, dominado por energías renovables. Esto también explicaría los reducidos beneficios climáticos asociados a la sustitución de la incineradora actual por una nueva planta, ya que la principal diferencia sería una mayor recuperación energética.

El impacto en toxicidad humana con efectos cancerígenos generado por el sistema de gestión de RSU está asociado casi exclusivamente con la incineración y el reciclaje (Figura 0-4b). Además, la incineración es el principal contribuyente al impacto en ecotoxicidad (Figura 0-4c). El alto impacto de la incineración en estas dos categorías se debe a la disposición de las cenizas volantes (residuos del sistema de limpieza de gases), que provocan la migración de metales pesados hacia agua subterráneos y superficiales a largo plazo. En este contexto, la eliminación de la incineración reduce 1,2 veces el impacto en toxicidad humana con efectos cancerígenos y 1,7-2,2 veces el impacto en ecotoxicidad. Por el contrrio, la incineración tiene una contribución insignificante al impacto en toxicidad humana sin efectos cancerígenos, que está dominado por el uso del compost debido a la adición de metales pesados al suelo agrícola (Figura 0-4d).

Estudios previos de ACV de incineración de RSU identificaron las emisiones de dioxinas, furanos y metales pesados como la principal fuente de impacto en la salud humana (Eisted and Christensen, 2013; Rajaeifar et al., 2015; Sharma and Chandel, 2017). Los resultados obtenidos en este trabajo, sin embargo, apuntan a la gestión de las cenizas volantes como el principal riesgo para la salud humana. El impacto en toxicidad humana con efectos cancerígenos generado por la incineración se debe casi exclusivamente (99% del impacto) a la emisión de cromo VI a las aguas superficiales. Por otro lado, el impacto en ecotoxicidad se debe principalmente a la emisión de antimonio a las aguas superficiales (82%) y, en menor medida, la emisión de cromo VI (17%). Vale la pena señalar que los inventarios de la disposición de las cenizas se obtuvieron de la base de datos Ecoinvent v3.7.1 debido a la falta de datos consistentes para modelar estos inventarios. Por lo tanto, los resultados están directamente influenciados por las suposiciones adoptadas en Ecoinvent. Los principales supuestos son las concentraciones de lixiviados y la contabilización de la generación y las emisiones de lixiviados a largo plazo (Doka, 2009). Si bien otros trabajos también han señalado el alto potencial de toxicidad de la gestión de las cenizas volantes (Beylot et al., 2018; Lausselet et al., 2016), las incertidumbres en torno a este tema siguen siendo grandes.

0.4.3 Desarrollo de un modelo de optimización para la planificación de rutas de valorización energética de RSU

Introducción

Los resultados obtenidos en los apartados anteriores proporcian algunas ideas sobre el futuro de la valorización energética de RSU en el contexto de una economía cada vez más circular. Se concluyó que los objetivos de recogida selectiva y reciclaje no necesariamente comprometen el potencial de valorización energética y que la eliminación de la incineración podría reducir el impacto en las categorías de cambio climático, toxicidad humana y ecotoxicidad a expensas de poner en riesgo la consecución de los objetivos de vertedero. En base a estos antecedentes, este apartado se centra en (1) determinar las rutas óptimas de valorización energética que podrían priorizarse en el futuro y (2) evaluar prospectivamente el desempeño económico y climático de estas rutas (**PI3**).

Metodología

Para realizar este estudio, se desarrolló un modelo de optimización multiperiodo que determina el diseño óptimo del sistema de gestión de RSU de acuerdo con objetivos económicos o climáticos y sujeto a la disponibilidad y composición de los residuos, las limitaciones y restricciones del sistema (p. ej., balances de masa y energía y restricciones de capacidad) y los objetivos de las políticas de residuos (p. ej., objetivos de vertedero). Para determinar el diseño óptimo del sistema, el modelo toma las siguientes decisions:

- Utilizar, mejorar o desmantelar las instalaciones existentes.
- Invertir en nuevas instalaciones (y con qué capacidades).
- Flujos de residuos hacia y entre las instalaciones de tratamiento.

Estas decisiones se toman en cada período a lo largo del horizonte temporal estudiado. Por defecto, el horizonte temporal abarca desde 2020 hasta 2040 dividido en intervalos de 5 años.

La función objetivo económica consiste en maximizar el valor actual neto (VAN) del sistema. El VAN se calcula a partir del flujo de caja neto descontado a lo largo del horizonte temporal. Además, el flujo de caja neto en cada período se calcula como la diferencia entre los ingresos generados por el sistema gracias a la venta de materiales y energía y los costes de capital y operación de las plantas de tratamiento (Martinez-Sanchez et al., 2015; Taelman et al., 2020). Por otro lado, la función objetivo climática se mide mediante el impacto de ciclo de vida en cambio climático. Para calcular este impacto se han seguido las mismas suposiciones metodológicas del apartado anterior (es decir, límites del sistema, multifuncionalidad, etc.).

El modelo de optimización desarrollado en esta tesis aborda los tres aspectos identificados en el apartado de Metodología. En primer lugar, el modelo incluye tanto las plantas de tratamiento existentes, así como la posibilidad de mejorar o desmantelar estas plantas y de invertir en nuevas instalaciones. En segundo lugar, el modelo incorpora el efecto de la economía de escala. Para ello, se han implementado funciones de costes que captan la relación entre los costes de capital y operación y la capacidad de las nuevas plantas. Por último, el desempeño tecnoeconómico y ambiental de los procesos de tratamiento de residuos se han modelado en base a la composición y propiedades fisicoquímicas de los residuos. Para ello, se implementaron los inventarios desarrollados en el modelo de ACV en el modelo de optimización, dando lugar a un modelo no lineal y no convexo. Para resolver este modelo, se ha implementado un algoritmo heurístico que tiene la ventaja de generar una solución óptima en un período reducido de tiempo.

El modelo de optimización se ha aplicado para evaluar las rutas óptimas de valorización energética en Madrid para el periodo 2020-2040. Se han evaluado tres escenarios que difieren en cuanto a la función objetivo y las restricciones del sistema. El escenario *Business-as-Usual* (BAU) tiene como objetivo maximizar el VAN del sistema de gestión de RSU, utilizando las instalaciones existentes sin restricciones de vertedero. En este escenario, se permite la inversión en nuevas instalaciones siempre y cuando la capacidad existente no sea suficiente para tratar todos los residuos generados. No obstante, en este escenario no se considera la instalación de nueva capacidad de incineración. Los otros dos escenarios analizados tienen por objetivo maximizar el VAN (escenario Max. Valor Actual Neto) o minimizar el impacto en cambio climático (escenario Min. Impacto en Cambio Climático) con total libertad de elección de las instalaciones de tratamiento de residuos. No obstante, el número de nuevas instalaciones de incineración se ha restringido a una, ya que es poco probable que se vaya a instalar más de una nueva incineradora. Además, en estos dos escenarios se ha inpuesto la restricción de alcanzar un 10% de verdero para el año 2035, en línea con la Directiva 2018/850.

Resultados y discusión

La Figura 0-5 muestra la optimización de las cantidades anuales de residuos destinadas a cada planta de tratamiento en cada período en los escenarios analizados. Además, la Tabla 0-3 muestra las capacidades óptimas de las plantas de valorización energética (digestión anaerobia e incineración).

El escenario BAU muestra un escenario hipotético en el cual la situación actual se extiende hasta 2040. En este escenario, la fracción restos de hogares y comercios (entre 781.207 y 583.251 t/año) se separa en las plantas de separación existentes. Las fracciones envases, papel/cartón y vidrio también se mandan a las plantas de separación correspondientes. Debido al aumento de la recogida selectiva de envases (desde 107.636 hasta 118.120 t/año durante 2020-2040), una nueva planta de separación con capacidad para 25.000 t/año se instala en el periodo 2030-2035. La materia orgánica proveniente de la recogida selectiva es tratada en las plantas de digestión anaerobia existentes (entre 157.180 y 201.220 t/año). Sin embargo, la capacidad de estas plantas resulta insuficiente para hacer frente al aumento de la recogida selectiva, por lo que el modelo añade una nueva planta de compostaje en el período 2030-2035 con una capacidad de 62.001 t/año. La materia orgánica residual separada de la fracción restos en las plantas de separación se trata principalmente mediante digestión anaerobia (entre 163.963 y 93.676 t/año) y, en menor medida, mediante compostaje (media de 33.608 t/año). La incineradora actual se utiliza durante todo el horizonte temporal para tratar una media de 298.935 t/año de rechazos de las plantas de separación. Por último, entre 626.860 y 503.910 t/año de residuos se mandan a vertedero. La capacidad del vertedero actual se supera en el primer período, por lo que el modelo añade un nuevo vertedero con capacidad para 8,2 millones de toneladas de residuos. Las tasas de reciclaje y vertedero en el escenario BAU alcanzan 37% y 34% para 2035-2040, mientras que la tasa de incineración se mantiene constante en 23%.

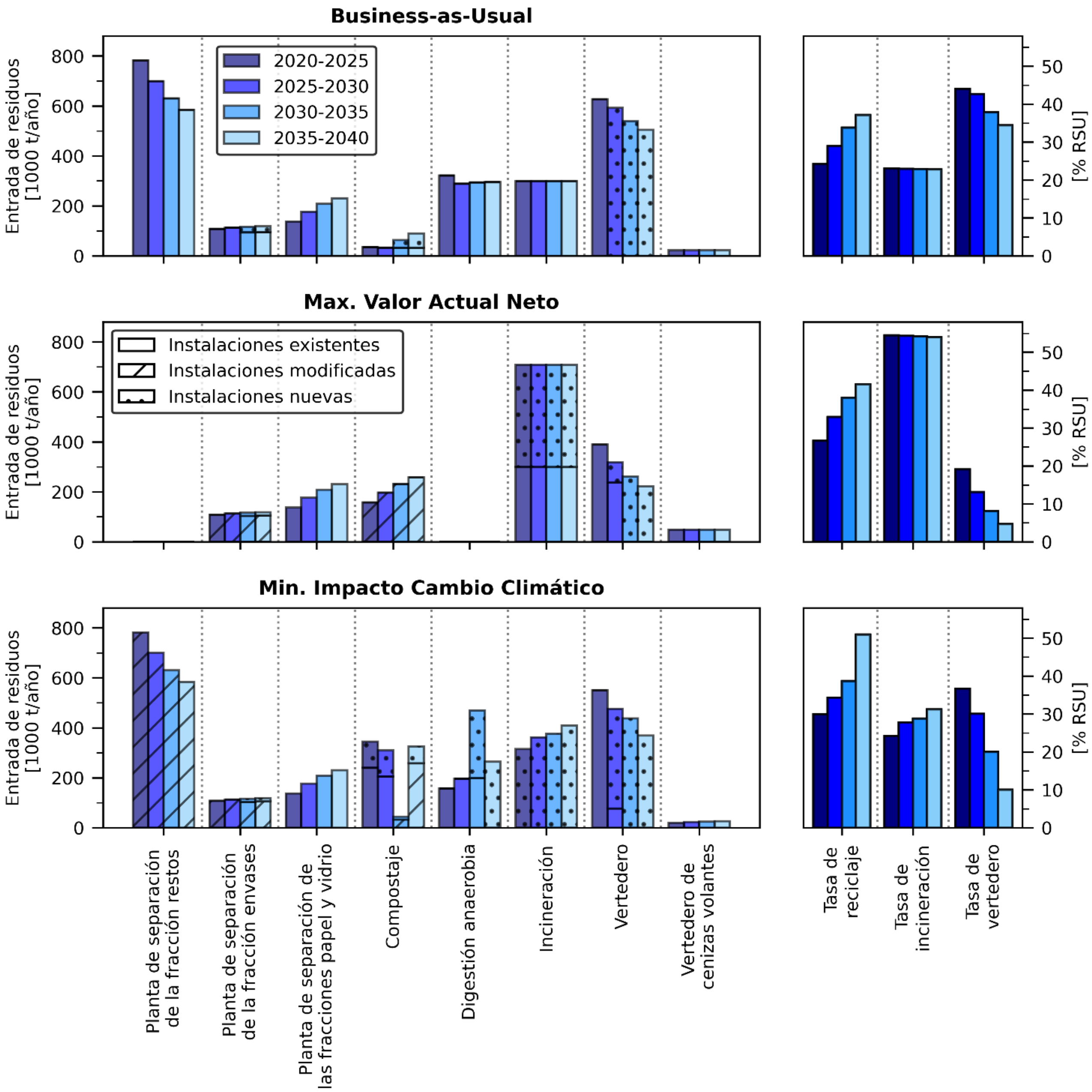

Figura 0-5. Cantidades anuales de residuos destinadas a cada tratamiento en cada período en los escenarios *Business-as-Usual*, Max. Valor Actual Neto y Min. Impacto Cambio Climático. Las tasas de reciclaje, incineración y vertedero en cada período se muestran a la derecha.

Tabla 0-3. Capacidades de tratamiento (t/año) óptimas de las plantas de valorización energética (digestión anaerobia e incineración) en cada periodo en los escenarios *Business-as-Usual*, Max. Valor Actual Neto y Min. Impacto Cambio Climático.

Escenario	**Planta de valorización energética**	**2020-2025**	**2025-2030**	**2030-2035**	**2035-2040**
Business-as-Usual	Existente digestión anaerobia para la materia orgánica separada en origen	161.000	161.000	161.000	161.000
	Existente digestión anaerobia para la materia orgánica residual	108.175	108.175	108.175	108.175
	Existente incineración	328.500	328.500	328.500	328.500
	Nueva digestión anaerobia para la materia orgánica residual	–	–	–	–
	Nueva incineración	–	–	–	–
Max. Valor Actual Neto	Existente digestión anaerobia para la materia orgánica separada en origen	D	D	D	D
	Existente digestión anaerobia para la materia orgánica residual	D	D	D	D
	Existente incineración	328.500	328.500	328.500	328.500
	Nueva digestión anaerobia para la materia orgánica residual	–	–	–	–
	Nueva incineración	450.000	450.000	450.000	450.000
Min. Impacto Cambio Climático	Existente digestión anaerobia para la materia orgánica separada en origen	161.000	161.000	161.000	D
	Existente digestión anaerobia para la materia orgánica residual	D	D	D	D
	Existente incineración	D	D	D	D
	Nueva digestión anaerobia para la materia orgánica residual	–	–	173.862	173.862
	Nueva incineración	450.000	450.000	450.000	450.000

Nota: 'D' indica planta desmantelada

Los escenarios Max. Valor Actual Neto y Min. Impacto Cambio Climático muestran dos vías diferentes que permitirían a Madrid alcanzar el objetivo de 10% vertedero para 2035. En el escenario Max. Valor Actual Neto, la tasa de vertedero alcanza el 5% en el periodo 2035-2040. Esto se logra en gran medida gracias a un uso más intensivo de la incineración. En este escenario, se incineran una media de 708.435 t/año de residuos en cada periodo. La mayor diferencia con respecto al BAU es que la fracción restos de hogares y comercios se incineran directamente, mientras que las plantas de separación se desmantelan. Dado que la capacidad de la incineradora actual no es suficiente, se instala una nueva incineradora con una capacidad de 450.000 t/año. La capacidad total de incineración alcanzaría así las 778.500 t/año, mientras que la tasa de incineración aumentaría hasta el 54%. Esta tasa de incineración es similar a la de otros países de la UE que logran una tasa de vertedero inferior al 1%, es decir, Finlandia (56% incineración), Suecia (53%) y Dinamarca (48%).

Además de la incineración directa de la fracción restos, el escenario Max. Valor Actual Neto también implementa los siguientes cambios: (1) las plantas de separación de envases se mejoran en el primer período, lo que aumenta la recuperación de materiales y los ingresos, (2) se instala una nueva planta de separación de envases con una capacidad de 25.000 t/año en el período 2030-2035 (como en el BAU), (3) las plantas de digestión anaerobia existentes se desmantelan debido a sus altos costes en comparación con el compostaje, (4) la planta de compostaje existente se adapta para recibir la materia orgánica proveniente de la recogida selectiva (actualmente recibe materia orgánica residual), y (5) se abre un nuevo vertedero con capacidad para 2,8 millones de toneladas de residuos en el segundo periodo.

Por ultimo, el escenario de mínimo impacto en cambio climático alcanza un 10% de vertedero en 2035. Este escenario se basa en opciones tecnológicas que maximizan el reciclaje debido a los altos impactos evitados por la sustitución de materiales primaries. Por lo tanto, este escenario muestra la tasa de reciclaje más alta, con un 51% para 2035. Las plantas de separación de restos y envases se mejoran desde el primer período para maximizar la recuperación de materiales. La incineradora actual se elimina en el primer período y se sustituye por una nueva instalación con una capacidad de 450.000 t/año. Esta sustitución viene motivada por la mayor eficiencia de la nueva instalación (26%) en comparación con la existente (12%) pero también porque la nueva instalación recupera metales no ferrosos (aluminio) de las cenizas.

Como se puede observar, la incineración juega un papel fundamental en el diseño de un sistema óptimo de gestión de RSU. Sin embargo, los tres escenarios analizados difieren en cuanto a los flujos de residuos destinados a incineración y sus cantidades. Por ejemplo, en el escenario BAU se incineran los rechazos de las plantas de separación, mientras que en el escenario de mínimo coste se incinera la fracción restos. Estos cambios pueden tener grandes implicaciones para el desempeño económico y climático de la incineración. En este sentido, la Figura 0-6 muestra el coste y la huella de carbono de la incineración en cada escenario.

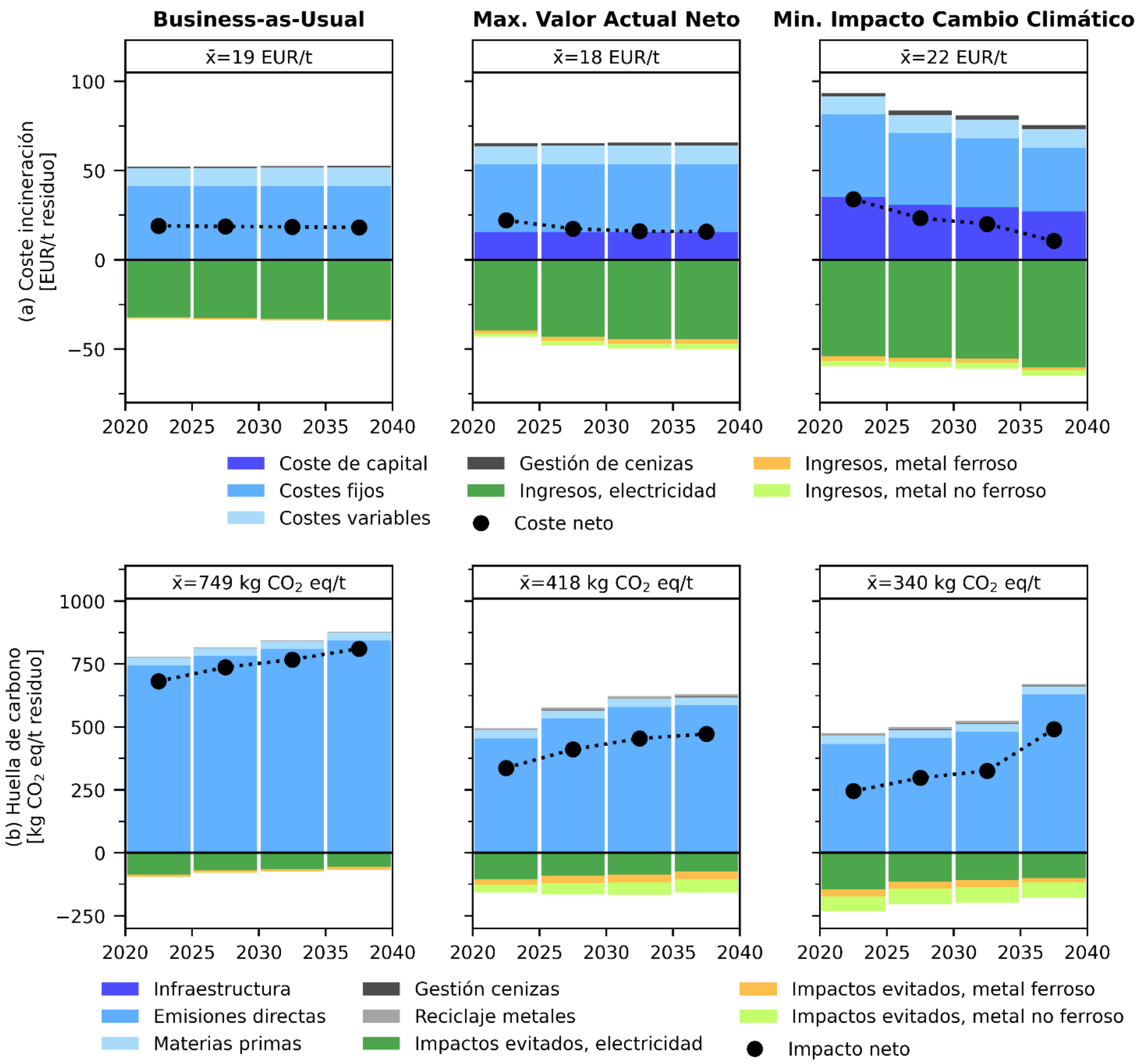

Figura 0-6. Desglose de costes (a) y huella de carbono (b) de la incineración en los escenarios Business-as-Usual, Max. Valor Actual Neto y Min. Impacto Cambio Climático. El cuadro de texto muestra los promedios durante 2020-2040. Los datos económicos están armonizados a EUR2019.

El coste bruto promedio de la incineración durante el horizonte temporal (es decir, el coste sin considerar los ingresos) oscila entre 53 EUR/t en el escenario BAU, 67 EUR/t en el escenario Max. Valor Actual Neto y 86 EUR/t en el escenario Min. Impacto Cambio Climático (Figura 0-6a). Los dos últimos escenarios muestran un coste bruto superior debido a la inversión en una nueva planta de incineración. A pesar de esto, los tres escenarios son comparables en cuanto al coste neto promedio (es decir, el coste considerando los ingresos), que va desde 20 EUR/t en los escenarios BAU y Max. Valor Actual Neto hasta 25 EUR/t en el escenario Min. Impacto Cambio Climático. Esto se debe a que los dos escenarios que tienen una nueva planta de incineración generan entre un 31% (escenario Max. Valor Actual Neto) y un 71% (escenario Min. Impacto Cambio Climático) más electricidad por tonelada de residuo que el escenario BAU. Los ingresos por venta de electricidad compensan entre el 62% y el 67% del coste bruto, dependiendo del escenario. En el escenario BAU, el coste neto se mantiene prácticamente constante durante el periodo 2020-2040, mientras que en los otros dos escenarios tiende a disminuir, desde 24 hasta 18 EUR/t en el escenario Max. Valor Actual Neto (disminución del 25%) y desde 37 hasta 13 EUR/t en el escenario Min. Impacto Cambio Climático (67% de disminución).

La huella de carbono promedio de la incineración va desde 749 kg CO_2 eq/t en el escenario BAU hasta 418 kg CO_2 eq/t en el escenario Max. Valor Actual Neto y 340 kg CO_2 eq/t en el escenario Min. Impacto Cambio Climático (Figura 0-6b). El escenario BAU muestra la mayor huella de carbono debido a unas mayores emisiones directas de CO_2. Estas emisiones se deben a una mayor proporción de plástico en los residuos incinerados en este escenario. Además, la nueva planta de incineración instalada en los escenarios Max. Valor Actual Neto y Min. Impacto Cambio Climático reduce aún más los impactos debido a su mayor eficiencia eléctrica y la recuperación de aluminio de las cenizas. La huella de carbono aumenta con el tiempo en los tres escenarios, desde 682 hasta 811 kg CO_2 eq/t en el BAU (aumento del 19%), desde 337 hasta 472 kg CO_2 eq/t en el escenario Max. Valor Actual Neto (40% de incremento) y desde 245 ahasta 491 CO_2 eq/t en el escenario Min. Impacto Cambio Climático (aumento del 100%). El aumento de los impactos en cambio climático de la incineración se debe a dos factores: (1) el aumento de las emisiones directas de CO_2 debido a una mayor proporción de plásticos incinerados y (2) la disminución de los impactos evitados por la sustitución de electricidad a medida que aumenta el porcentaje de energías renovables en el mix eléctrico.

0.5 Conclusiones

A continuación, se presentan los principales resultados y conclusiones para las preguntas de investigación planteadas en esta tesis.

PI1 ¿Cuál será el potencial de valorización energética de los RSU en escenarios futuros de mayor recogida selectiva y reciclaje?

Una mayor recogida selectiva y reciclaje de RSU no compromete necesariamente el potencial de valorización energética. Para el caso de estudio de Madrid, el potencial bruto de valorización energética varía entre 12.436 TJ en 2019 y 8.942-11.327 TJ en 2040, dependiendo del escenario. Por lo tanto, incluso en el escenario más prometedor para el reciclaje, el potencial bruto de valorización energética disminuye en un 28%. Este impacto relativamente bajo puede atribuirse a las ineficiencias en la separación en origen y la eficiencia limitada de las plantas de separación y clasificación. En general, la disponibilidad y las características de los flujos de residuos en los escenarios futuros analizados permitiría la operación de grandes instalaciones de valorización energética (p. ej., incineración y digestion anaerobia). Sin embargo, la implementación de medidas de prevención de residuos o esquemas de recolección alternativos, como los esquemas de depósito y reembolso, tendrá un efecto en el potencial de valorización energética.

PI2 ¿Cuáles son las posibles consecuencias ambientales de la eliminación de la incineración del sistema de gestión de RSU?

La eliminación de la incineración del sistema de gestion de RSU de Madrid podría reducir los impactos ambientales, incluido cambio climático, acidificación, eutrofización terrestre y de agua dulce, formación de ozono fotoquímico, toxicidad humana y ecotoxicidad. La sustitución de la incineradora actual por una planta nueva de mayor eficiencia energética traería escasos beneficios climáticos. Esto se debe en gran medida a los reducidos impactos evitados mediante la recuperación de electricidad dada la baja huella de carbono del mix eléctrico español. La eliminación de la incineración traería grandes beneficios en términos de toxicidad humana (con efectos cancerígenos) y ecotoxicidad. Estos beneficios se deben principalmente a que las cenizas volantes (residuos del sistema de limpieza de gases) son una gran fuente de toxicidad debido a la posible lixiviación y, en consecuencia, migración de metales pesados hacia aguas superficiales y subterráneos. Cabe señalar que la eliminación de la incineración aumentaría significativamente la tasa de vertedero y pondría en riesgo el objetivo de 10% de vertedero para 2035. Otros riesgos ambientales asociados con los vertederos, como la posible contaminación de ríos y océanos con plásticos, no han sido investigados en esta tesis. Por lo tanto, los objetivos de reducción de vertedero no deben ser dejados de lado, aunque lograr estos objetivos puede requerir cambios estructurales en el sistema de gestión de RSU más allá de la utilizaicón o no de la incineración.

PI3 ¿Cuáles son las rutas de valorización energetica óptimas desde el punto de vista económico y/o ambiental que podrían priorizarse en el futuro dada la disponibilidad y características de los RSU?

La aplicación del modelo de optimización al caso de estudio de Madrid reveló que alcanzar un 10% de vertedero para 2035 requiere un uso más intensivo de la incineración. Si bien, desde un punto de vista económico, la incineración podría volverse aún más atractiva en el futuro, es probable que su huella de carbono aumente con el riesgo de alejar esta opción de los beneficios climáticos que a menudo se afirman. Por otro lado, la digestión anaerobia resulta atractiva para minimizar los impactos en cambio climático, pero no para minimizar el coste del sistema, para lo cual el compostaje tiene ventaja. El despliegue de la digestion anaerobia podría verse afectado por las restricciones de vertedero debido a la gran cantidad de rechazos producidos durante el pretratamiento, especialmente si la recogida selectiva de materia orgánica es relativamente ineficiente.

En base a los resultados obtenidos en esta tesis, la visualización de un futuro con un papel relevante para la valorización energética parece muy probable incluso en el contexto de una economía cada vez más circular. Por lo tanto, el trabajo futuro debe priorizar el enriquecimiento de la cartera tecnológica del modelo de optimización para incluir rutas de valorización energética emergentes, en particular la gasificación y la pirólisis. Es probable que estas tecnologías adquieran un papel cada vez más importante en las próximas décadas y deben ser consideradas. El desarrollo de inventarios (balances de materia y energía) predictivos que tengan en cuenta la respuesta de estas tecnologías a los cambios en la composición de los residuos es clave. Además, los siguientes dos aspectos merecen especial atención dado su alto impacto en el potencial de valorización energética: (1) mejorar la proyección de la generación de RSU y (2) un modelado más exhaustivo de la recogida selectiva, incluyendo la consideración de sistemas de recogida alternativos.

Chapter 1
Background and motivation

1.1 Municipal solid waste management

The unprecedented economic growth experienced over the last decades relied on an intensive use of easily accessible and cheap resources. Global consumption of raw materials has never stopped growing while projections represent a doubling of demand by 2060 (OECD, 2019). Yet, the dependence on increasingly scarce natural resources along with the deterioration of the environment have alerted decision-makers to the necessity of a paradigm shift in the economic model (Ellen MacArthur Foundation, 2015). In light of this, the circular economy continues to gain traction in the political agenda for a competitive and climate-neutral economy (European Commission, 2020). Circular economy is broad in scope (Kirchherr et al., 2017), but most frequently is framed as a way of aligning economic growth with environmental constraints through the efficient use of resources (McDowall et al., 2017). In a circular economy, the value of resources is retained within the economic system for as long as possible through waste prevention, re-use, recycling, and recovery. Consequently, waste management is called upon to play a relevant role in a circular economy (European Commission, 2020).

Over 2,000 million tonnes of municipal solid waste (MSW) were generated globally in 2016, and this figure is expected to increase up to 3,400 million tonnes by 2050 (Kaza et al., 2018). The Europe and Central Asia region is currently the second largest MSW generator, concentrating 20% of the global generation. The European Union (EU) generates over 226 million tonnes of MSW per year, which is equivalent to 505 kg MSW per capita per year (Eurostat, 2020). The EU Waste Framework Directive 2008/98/EC defines MSW as "*waste from households and waste from other sources, such as retail, administration, education, health services, accommodation and food services, and other services and activities, which is similar in nature and composition to waste from households*". Generally speaking, MSW consists of a heterogeneous mixture of food waste, paper, cardboard, plastic, metal, glass, textiles, wood, and other materials.

Turning the vast amount of MSW into new materials and energy seems a sensible path to provide EU industry with domestic resources, to reduce the imports of raw materials and, ultimately, to accelerate the transition to a more circular and resource-efficient economy (European Commission, 2011). Currently some 48% of the MSW generated in the EU is recycled and/or composted, 27% is incinerated for energy recovery, and 23% is disposed of in landfills (Eurostat, 2020). Yet, there are notable cross-country differences. Landfilling is the predominant MSW disposal method in almost half of the Member States, with some countries such as Greece, Croatia, Spain, Slovakia, and Hungary still landfilling more than 50% of their MSW (Figure 1-1).

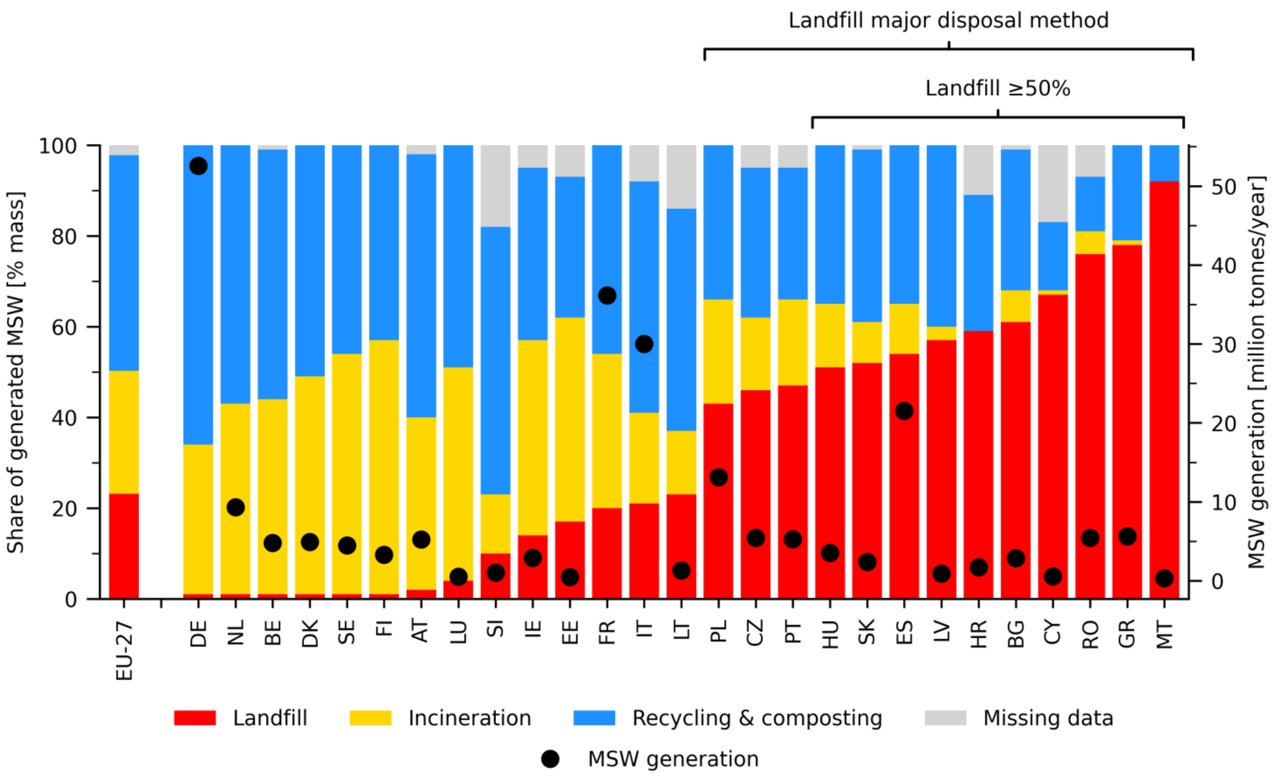

Figure 1-1. Municipal solid waste generation (right *y*-axis) and treatment shares (left *y*-axis) in the EU in 2019, based on Eurostat (2020) and CEWEP (2021a).

Accelerating the transition to a more circular economy calls for substantial changes in the MSW management system. The EU Waste Framework Directive establishes that waste management must be consistent with the waste hierarchy, which ranks prevention and preparation for re-use as the preferred options, followed by recycling and energy recovery with landfill being the very last option. Within this context, actions oriented to prevent, re-use, and recycle waste are at the top of the EU waste policy. Directive 2018/851 obliges Member States to prepare for re-use and recycling at least 55% of MSW by 2025, 60% by 2030, and 65% by 2035. Directive 2018/852 establishes a recycling target for packaging waste of 65% by 2025 and 70% by 2030. Furthermore, Directive 2018/852 sets recycling targets for specific packaging materials by 2030 as high as 85% for paper/cardboard, 55% for plastic, 80% for ferrous metal, 60% for aluminum, 75% for glass, and 30% for wood. Since the heterogeneous nature of MSW makes its direct recycling hardly feasible, the EU waste policy also mandates its separate collection. Separate collection is mandatory as of 2015 for paper, metal, plastic, and glass, by 2023 for organic waste (bio-waste), and by 2025 for textile and hazardous household waste (Dubois et al., 2020). In the same vein, Directive 2019/904 establishes a 90% separate collection of single-use plastics by 2029, while the new Circular Economy Action Plan proposes halving the amount of residual waste (remaining waste after separate collection of recyclables and organic waste) by 2030 (European Commission, 2020).

In the meanwhile, energy recovery, or waste-to-energy (WTE), seems to have received rather scarce attention in the political agenda (Malinauskaite et al., 2017). While traditionally associated with incineration, WTE is a general term that embraces thermochemical (e.g., incineration, gasification, and pyrolysis) and biochemical (e.g., anaerobic digestion (AD) and landfill gas (LFG) collection) conversion processes that can exploit the energy content of waste. Electricity, heat, and biomethane are the typical products obtained from MSW, while the production of biofuels or hydrogen is at an early development stage (Foster et al., 2021; Mukherjee et al., 2020). The interest in WTE, notably incineration and AD, has become ever more prominent due to an increased generation of MSW and the potential environmental risks that arise from its disposal in landfills (UNEP, 2015). Besides being a big loss of valuable resources, landfills have been recognized as one of the largest sources of anthropogenic methane emissions (Bogner et al., 2007) and can represent a risk for human health due to groundwater and soil contamination (WHO, 2016) as well as a source of marine plastic pollution (UNEP, 2018). In this regard, WTE offers the dual benefits of (1) preventing landfilling and the associated environmental and human health risks and (2) providing a domestic source of energy with the potential of reducing fossil fuels consumption and the associated greenhouse gas (GHG) emissions (World Energy Council, 2016).

While the existing waste policy and circular economy initiatives do not establish specific targets for energy recovery, WTE is expected to gain momentum in view of the Directive 2018/850, amending Directive 1999/31/EC on the landfilling of waste, which limits the share of MSW landfilled to 10% by 2035. According to the European Commission, "*WTE processes can play a role in the transition to a circular economy provided that the EU waste hierarchy is used as a guiding principle and that choices made do not prevent higher levels of prevention, re-use and recycling*" (European Commission, 2017). Yet, the transition to a circular economy is inherently intended to cause large-scale effects in the WTE sector that remain poorly understood. For example, many questions arise on how an increased recycling of combustible materials such as plastic, paper, and textile would affect incineration. To date, much effort has been devoted to assessing the circularity of key materials, including plastic (Eriksen et al., 2020, 2018), paper (Van Ewijk et al., 2018), and aluminium (Sevigné-Itoiz et al., 2014). Meanwhile, the long-term challenges faced by the WTE sector in the context of a circular economy have not been fully explored yet. In light of this, this thesis addresses the potential consequences that the transition to a more circular economy can have on the energy recovery potential of MSW and the deployment, use, and economic and environmental sustainability of WTE technologies.

1.2 Energy recovery from municipal solid waste

Within the European context, MSW has an average calorific value of 9 MJ/kg (Saveyn et al., 2016). This energy content is low compared with other solid fuels such as coal (≈ 29 MJ/kg) and wood (≈ 15 MJ/kg). Furthermore, MSW is a very heterogeneous feedstock whose composition (i.e., the share of food waste, paper, plastic, etc.) and physicochemical properties (e.g., energy, moisture, and carbon content) may vary considerably across space and time. Such variability reflects seasonality, changes in consumption patterns, level of urbanization, etc. The complexity of MSW makes its exploitation a challenging task, but despite this, it continues to gain traction as a valuable source of energy (World Energy Council, 2016).

When focusing on energy recovery from MSW, it is important to note that MSW is a very regional issue and that the design and performance of the MSW management system can vary substantially from one region to another. Furthermore, modern integrated management systems encompass a complex network in which waste is initially separated by materials type and nature (e.g., packaging, paper/cardboard, glass, and organic waste) and then treated through a broad range of interconnected and interrelated facilities (see Box 1.1). Thus, MSW management involves numerous waste streams with variable composition and properties and, consequently, susceptible to different treatment pathways. For example, the organic fraction of MSW can be collected separately and treated through AD, while the rejects from sorting residual waste at a material recovery facility (MRF) are more suitable for incineration. The composition and properties of the waste received at a treatment facility (e.g., rejects received at incineration facilities) may depend entirely on the performance of previous stages (e.g., the efficiency of separate collection and the sorting efficiency of the MRF). All in all, it is only through a regional system perspective that a holistic picture of energy recovery from MSW can be obtained. In this thesis, WTE is addressed from a broader system perspective as opposed to the technology perspective.

Box 1.1 Terms and concepts: integrated MSW management system

An integrated MSW management system encompasses a complex network of interconnected processes including waste collection, treatment, recycling, and final disposal. The county or municipality is generally responsible for the collection, treatment, and final disposal of the MSW generated within its geographical boundary. Recycling, on the other hand, usually takes place elsewhere in a global market. MSW is generally collected separately by materials type and nature, e.g., packaging, paper/cardboard, glass, and organic waste, with the remaining waste collected in the so-called residual waste. Following separate collection, MSW streams are treated through a broad range of interconnected facilities, such as material recovery facilities (MRFs), anaerobic digestion (AD), composting, and incineration. Treatment aims at maximizing the recovery of resources and ensuring that the remaining residues are disposed of in a safe manner.

Box 1.1 ***(continued)***

Packaging waste, paper/cardboard, and glass streams are typically delivered to MRFs for additional sorting. MRFs are automatized facilities for sorting the heterogeneous input waste into single material type streams, such as paper, cardboard, polyethylene terephthalate (PET), high-density polyethylene (HDPE), and ferrous metal. The sorted materials are baled and sold to recyclers. The organic waste collected separately is sent to AD and/or composting to reduce the organic degradable matter and recover energy and/or nutrients. The residual waste can be incinerated for energy recovery and/or disposed of in a sanitary landfill where the generated landfill gas (LFG) is partially collected for energy recovery. Alternatively, the residual waste can be sorted at MRFs either to enhance the recovery of materials for recycling or to produce a refuse-derived fuel (RDF). Sometimes MRFs also separate organic waste from the residual waste, obtaining the so-called residual organic waste which is treated through composting/AD and/or disposed of in landfill. Since the efficiency of MRFs is limited, an output stream of rejects is always generated. Rejects follow the same treatment pathways as the residual waste.

1.2.1 Overview of waste-to-energy pathways

WTE is most often used in reference to incineration, the most widespread technology for the recovery of energy from waste. However, a broad range of technologies, such as AD, gasification, pyrolysis, and LFG collection, can be included under the umbrella of WTE. Furthermore, the MSW management system involves a range of waste streams susceptible to energy recovery, giving rise to numerous possible WTE pathways. The term “WTE pathway” is defined here as the entire chain of processes used to recover energy from waste, including from separate collection to the thermochemical or biochemical conversion of waste to energy (Figure 1-2). The following paragraphs synthesize the state-of-the-art of WTE pathways for MSW treatment. A detailed review of the technical features of WTE technologies is out of the scope of this work but has been provided in the best available techniques (BAT) documents for waste treatment (Neuwahl et al., 2019; Pinasseau et al., 2018) and the existing literature (Kumar and Samadder, 2017; Lombardi et al., 2015; Mukherjee et al., 2020; Tong et al., 2018).

Incineration pathways

Incineration is by far the most widespread WTE technology. Currently, about 27% of the MSW generated in the EU is incinerated for energy recovery, while countries such as Sweden, Denmark, and Finland incinerate more than 50% (Eurostat, 2020). There were 492 MSW incineration facilities operating in Europe in 2018, with a total annual treatment capacity of 96 million tonnes of waste (CEWEP, 2021b). In the United States, incineration with energy recovery accounts for 12% of MSW (US EPA, 2021), while in China the annual capacity of MSW incineration has increased from approximately 5.47 million tonnes in 2003 to 84.53 million tonnes in 2015 (Lu et al., 2017).

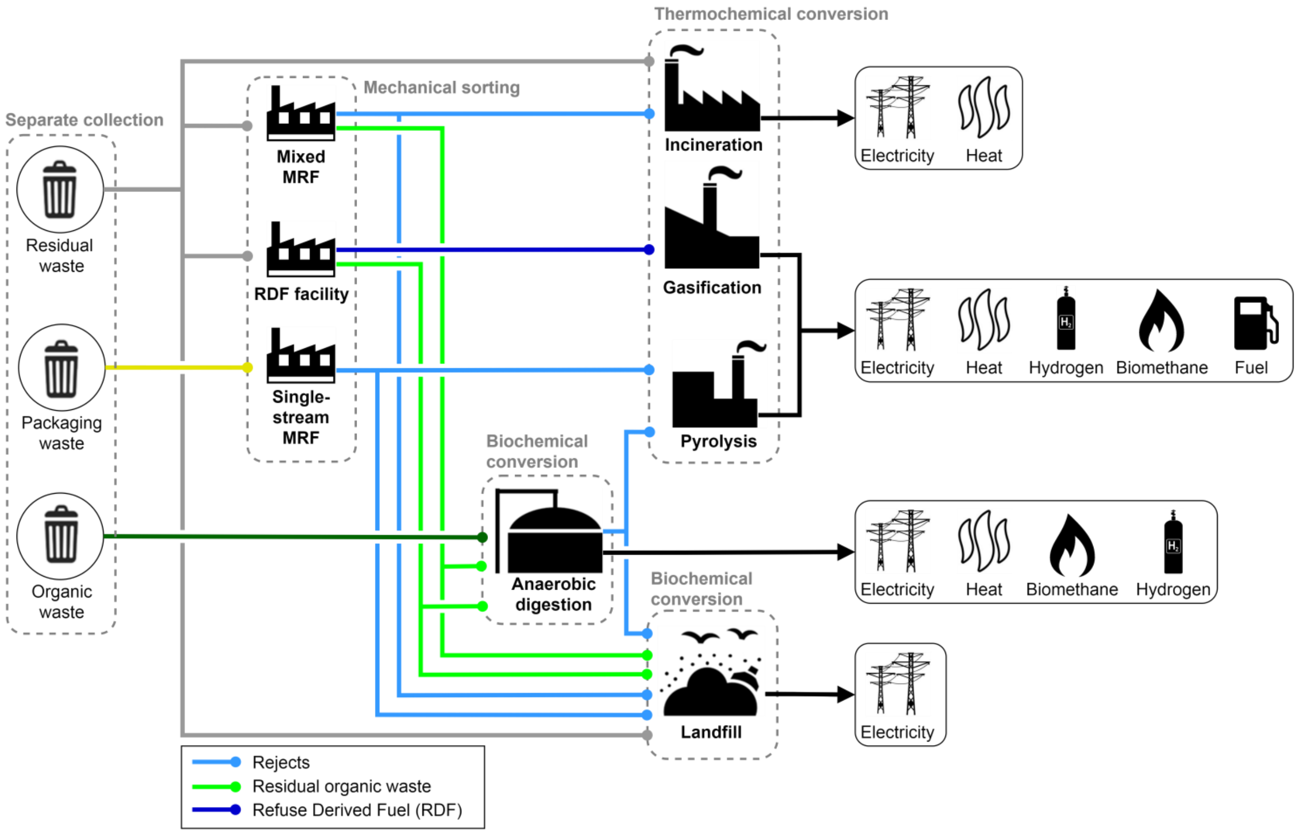

Figure 1-2. Overview of waste-to-energy pathways for municipal solid waste management.

Incineration consists of the complete oxidation of waste with a surplus of air at a temperature of 850 ºC or more. The process is exothermic, and most of the energy content of the waste is released as heat. Self-sustained combustion (i.e., without auxiliary fuels) is possible for waste streams with a minimum lower heating value (LHV) ranging from 5 to 7 MJ/kg (Hulgaard and Vehlow, 2011), while the typical LHV of the waste received at incineration facilities in Europe is in the range 10-12 MJ/kg (Reimann, 2012; Saveyn et al., 2016). The hot combustion gases are passed through a boiler, where their energy content is converted into superheated steam that is fed to a steam turbine connected to a power generator. The exhaust gases are passed through the air pollution control (APC) system before being released into the environment. The APC system can involve a variety of techniques, such as bag filters, cyclones, and electrostatic precipitators for dedusting, dry or wet scrubbing for acid gases abatement (HCl, HF, and SO_2), selective catalytic or noncatalytic reduction of NO_x, and reagents addition for dioxins abatement (Neuwahl et al., 2019). The solid residues (bottom ash produced from inert materials and fly ash from the APC system) are disposed of in landfills or treated for recycling as construction aggregates (only the bottom ash). Before that, ferrous and non-ferrous metals can be recovered from the bottom ash for recycling.

Incineration offers a valorization pathway to the residual waste and rejects generated at, for example, MRFs (Figure 1-2). Alternatively, the residual waste may undergo sorting, dehydration, and shredding to produce a refuse-derived fuel (RDF) with enhanced combustion properties. The RDF can be incinerated in dedicated facilities (Berardi et al., 2020) or co-incinerated in other installations such as cement kilns (Galvez-Martos and Schoenberger, 2014). In any case, incineration is intended as a straightforward pathway to reduce landfilling (Makarichi et al., 2018). Incineration is very efficient for this purpose since it reduces the weight of the waste by 75-90% (Hulgaard and Vehlow, 2011). This is reflected in Figure 1-1, where the lowest landfill rates are achieved by countries with relatively high incineration shares.

The incineration of MSW has also emerged as a domestic source of energy (World Energy Council, 2016). Incineration facilities are optimized for the supply of electricity, heat, or both in a combined heat and power (CHP) setting. In 2013, MSW incineration produced 110 PJ of electricity, which represents about 1% of the gross electricity produced in the EU in that year (Saveyn et al., 2016). In Northern Europe, where district heating networks are well-established, incineration satisfies a large proportion of the heat demand, for example, 20% in Denmark (Persson and Münster, 2016).

The electrical efficiency achieved at MSW incineration facilities is relatively low due to the corrosion nature of combustion gases, which limits the maximum temperature of the boiler (Rigamonti et al., 2013). BAT facilities are expected to exhibit a gross electrical efficiency in the range of 25-35% (Neuwahl et al., 2019), but existing older facilities show lower efficiencies (the electrical efficiency is defined in Box 1.2). The last energy efficiency report produced by the Confederation of European Waste-to-Energy Plants (CEWEP) found an average gross electrical efficiency of 21.6% for facilities producing electricity only and 15% in the case of CHP generation (Reimann, 2012). The average gross heat efficiency was 77.2% for facilities producing heat only and 37.1% in the case of CHP generation. In general, the electrical efficiency is higher for large-scale facilities due to improved steam parameters. For example, the steam used in large facilities (>250,000 tonnes/year) may reach 60-70 bar and 450-500 ºC, while small facilities (<100,000 tonnes/year) typically use 40 bar and 400 ºC. Thus, the average gross electrical efficiency reported by CEWEP ranges from 12.1% for small facilities to 15.6% for large facilities.

Box 1.2 Terms and concepts: electrical efficiency of waste-to-energy facilities

Gross electrical efficiency	Amount of electricity generated (including the electricity used internally) divided by the energy content of the waste (in the case of incineration), the biogas (in the case of anaerobic digestion), or the landfill gas (in the case of landfill gas collection).
Net electrical efficiency	Amount of electricity delivered to the grid after subtracting the electricity used internally divided by the energy content of the waste (incineration), biogas (anaerobic digestion), or landfill gas (landfill gas collection).

Anaerobic digestion pathways

AD is a well-established technology for the treatment of the organic fraction of MSW. It consists of a biochemical conversion process in which microorganisms break down organic matter in the absence of free oxygen following four phases, namely hydrolysis, fermentation, acetogenesis, and methanogenesis (Angelidaki and Batstone, 2011). The process converts biodegradable materials to biogas, a mixture of CH_4 (50-70% v/v) and CO_2 (30-50% v/v), and digestate, a nutrient-rich liquid-solid. The biogas is typically burned for electricity or CHP generation or upgraded to biomethane, which can be injected into the natural gas network or used as a fuel (Scarlat et al., 2018). Alternatively, the production of hydrogen (through biogas reforming and purification) or bioethanol (through digestate fermentation) has been proposed elsewhere (Byun et al., 2021). The digestate can be directly applied on land as an organic fertilizer in substitution for mineral fertilizers or composted before its use on land. As it can be seen, AD combines energy and nutrient recovery and, in consequence, it can be listed as a recycling option under certain circumstances. Directive 2018/851 indicates that AD may count as recycling when the mass of digestate/compost produced and applied on land is similar to the mass of organic waste treated. In any case, AD is considered a WTE technology in this work independently of the fate of the digestate.

AD technologies are typically classified based on the temperature or the moisture content achieved in the reactor (Kirchmeyr et al., 2020). Mesophilic AD operates at 35 ºC, while thermophilic AD operates at 53-55 ºC. On the other hand, dry systems operate with less than 75% moisture content in the reactor, while wet systems have a moisture content above 90%. AD is restricted to biodegradable waste materials. Accordingly, the organic waste collected separately is the most suitable feedstock for AD (Figure 1-2). However, the separate collection of organic waste is still marginal in many EU regions (Favoino and Giavini, 2020). A common alternative is to mechanically separate organic waste from residual waste at MRFs or RDF facilities, obtaining the so-called residual organic waste (Carchesio et al., 2020). When mechanical sorting and AD are integrated into the same plant, the resulting facility receives the name of mechanical-biological treatment (MBT).

AD requires the availability of a relatively constant and homogeneous (i.e., low level of impurities) organic waste feedstock (De Clercq et al., 2017). The biochemical methane potential (BMP) of food waste reaches 450 m^3 CH_4/tonne volatile solids (IEA Bioenergy, 2018a), but the presence of impurities (e.g., plastic, metal, and glass) can reduce this potential by up to 28% (Bernstad et al., 2013). Thus, the organic waste feedstock received at AD facilities usually undergoes certain pre-treatment to remove unwanted materials. A review of 17 Swedish AD facilities treating organic waste collected separately showed that between 2% and 45% of the input waste was rejected (average of 20%)

(Bernstad et al., 2013). These rejects are commonly disposed of in landfills, but they could be incinerated for energy recovery (Carlsson et al., 2015).

Gasification and pyrolysis pathways

Gasification and pyrolysis are promising alternatives to incineration (Mukherjee et al., 2020). Gasification consists of the partial oxidation of waste in the presence of less oxidizing agent than that required for full combustion. The process converts waste to syngas, a mixture of CO and H_2 with traces of CH_4 and other undesired products, and a solid product, a mixture of char and ashes. Pyrolysis, on the other hand, consists of the thermal degradation of waste at high temperatures (between 400 ºC and 800 ºC) in the complete absence of oxidizing agent (i.e., no oxidation occurs). This process converts waste to gas, oil, and a solid product (char). Both gasification and pyrolysis offer high flexibility regarding end-use applications. The syngas and pyrolysis oil can be (1) burned for electricity generation or CHP or (2) used as feedstock to produce hydrogen, biofuels, or chemicals (Figure 1-2). The syngas can be used to produce hydrogen through water-gas-shift reaction and purification (Ng and Phan, 2021), upgraded to biomethane through methanation (Ren et al., 2020), or converted to biofuels through the Fischer-Tropsch synthesis (Shahabuddin et al., 2020). On the other hand, pyrolysis oil can be upgraded to synthetic gasoline or diesel (Chen et al., 2015).

Despite significant research efforts, both gasification and pyrolysis are still in their infancy regarding large-scale implementation for MSW treatment. Gasification is in a more advanced stage (TRL 8-9) while pyrolysis is currently at TRL 5-6 (Foster et al., 2021). Japan is the only country that has decidedly opted for MSW gasification to the detriment of incineration, largely due to the lack of space for landfills and the environmental concerns related to incineration ashes (IEA Bioenergy, 2018b). Besides this, a few gasification facilities are operating in Europe and North America, the majority producing electricity and heat. An example is the Kymijärvi II facility in Finland, which processes 250,000 tonnes/year of RDF and supplies 50 MW of electricity and 90 MW of district heating (Helanti, 2016). A gasification facility in Edmonton (Canada) is the first of its class to produce methanol and ethanol from 100,000 tonnes/year of RDF. Similar facilities have already been approved to be constructed in Rotterdam (Netherlands) and Tarragona (Spain) with treatment capacities of 360,000 and 400,000 tonnes/year (Enerkem, 2022).

The heterogeneous nature of MSW is a major challenge to the operation of gasification and pyrolysis. The relatively high moisture content of MSW (up to 50%) can reduce the temperature inside the reactor due to water evaporation and can increase the production of H_2 over CO because the latter is consumed in the water-gas shift reaction. This effect causes a reduction in the LHV of syngas (Ramos et al., 2019). An additional issue of concern is ash melting at >800 ºC, which may cause clogging, reduces the overall efficiency, and induces mechanical deterioration of the equipment (Arena, 2012). Overall, the lack of sufficient maturity and/or low readiness level and the limited large-scale implementation make gasification and/or pyrolysis unlikely to completely displace incineration in the short to medium term.

Other waste-to-energy pathways

Once landfilled, biodegradable waste materials such as food, paper, cardboard, and wood start decomposing under anaerobic conditions and generate LFG, a mixture of CH_4 (45-55% v/v) and CO_2 (45-55% v/v) (IPCC, 2006a). Due to its relatively high CH_4 content, the LFG represents a threat to climate change, but also an opportunity for energy recovery. In this regard, existing landfills in developed countries are typically equipped with an LFG collection system and engines for electricity generation (Sauve and van Acker, 2020). Heat recovery from LFG combustion is not a common practice since landfills are usually located far from heat demand sources. Furthermore, the upgrading of the LFG to biomethane has been proposed elsewhere (IEA Bioenergy, 2018c).

A wide range of other novel WTE pathways not yet established in the market has been proposed for the organic fraction of MSW, including hydrothermal carbonization to produce hydrochar (IEA Bioenergy, 2020), hydrothermal liquefaction to produce biofuels (Gollakota et al., 2018), and ethanol fermentation (Beyene et al., 2018).

1.2.2 Environmental performance of waste-to-energy pathways

The potential environmental benefits arise as the main driving force behind the growing interest in energy recovery from MSW (UNEP, 2015). WTE can provide environmental benefits by (1) reducing the amount of waste landfilled and (2) recovering energy that can substitutes conventional counterparts and avoids the associated emissions (e.g., electricity generation through MSW incineration could substitute electricity from fossil fuels). But at the same time, WTE can cause substantial environmental impacts due to emissions from waste conversion, disposal of rejects and other residues, and consumption of resources (e.g., electricity generation and chemicals production). It is thus of paramount importance to follow a holistic life cycle perspective when pursuing the assessment and comparison of alternative WTE pathways (Manfredi and Pant, 2011). In this regard, life cycle assessment (LCA) appears as the most appropriate tool to quantify the potential environmental impacts associated with MSW management (Finnveden et al., 2007). The LCA methodology is further explained later in Section 1.4.

Herein, a review of the LCA literature was performed to shed light on the environmental consequences of implementing WTE pathways for MSW treatment*. Following the system perspective adopted in this thesis, the review focused on LCA studies concerning real MSW management systems that involve at least one comparison between two scenarios, a reference scenario without energy recovery and an alternative scenario with the implementation of a WTE pathway. The environmental consequences of implementing the WTE pathway were evaluated through the relative difference in the environmental impacts between the two scenarios. In this context, a negative value means beneficial environmental consequences. Further details about the review procedure are provided in Box 1.3.

Box 1.3 Review of environmental life cycle consequences of WTE pathways on the MSW management system

The literature search was performed via Scopus with the keywords "*life cycle assessment*" and "*waste management*". This review was performed at the beginning of this thesis and, therefore, it comprised papers published from 2009 to 2018 (included) in English-language peer-reviewed scientific journals, which covers 10 years of scientific research and, in the case of Europe, coincides with key efforts towards the implementation of the objectives of the Waste Framework Directive. This preliminary screening led to a total of 1,821 articles. Then, a more detailed screening was performed to identify only studies that fulfill the purpose of the review. The fundamental criteria applied were: (1) comparative LCA studies concerning real MSW management systems that involve at least one comparison between two scenarios, a reference scenario without energy recovery and an alternative scenario with energy recovery, and (2) all the MSW streams are included unless the excluded streams are managed identically in all scenarios. Finally, 27 articles were selected for the review.

According to the review methodology, 30 case studies were retrieved for review, from which three WTE pathways were identified: (1) diversion of waste from landfilling to incineration, (2) implementation of separate collection and AD of organic waste, and (3) diversion of organic waste from composting to AD. The environmental consequences were calculated for the most common impact categories addressed in the case studies, namely climate change, acidification, eutrophication, photochemical oxidant formation, and human toxicity.

Diversion of waste from landfilling to incineration

Figure 1-3 shows the life cycle environmental consequences of diverting residual waste or rejects from landfilling to incineration in 15 case studies from the literature. All the case studies concluded that this WTE pathway is effective to reduce the carbon footprint of MSW treatment. They also agree that the main drivers for GHG emissions savings are (1) the reduction of CH_4 emissions from landfilling and

* Based on: Istrate, I.R., Iribarren, D., Galvez-Martos, J.L., Dufour, J., 2020. Review of life-cycle environmental consequences of waste-to-energy solutions on the municipal solid waste management system. Resources, Conservation & Recycling 157: 104778

(2) the impacts avoided due to the substitution of other sources of energy elsewhere. Most of the case studies also indicate a potential reduction in photochemical oxidant formation due to the reduction of CH_4 emissions from landfills. Conversely, several case studies suggest trade-offs between the potential climate benefits and an increased impact on acidification, eutrophication, and human health. The most relevant discrepancies appear in terms of human health, even though the results are inconclusive. The case studies differ not only on the sense of the human health consequences (40% of case studies show impact increase while 60% show impact reduction) but also on the source of the impact on human health. The sources of human health impact identified were leaching of heavy metals from landfill (Erses Yay, 2015), emission of LFG (Fernández-Nava et al., 2014), or emissions of dioxins, furans, and heavy metals from incineration (Eisted and Christensen, 2013; Rajaeifar et al., 2015; Sharma and Chandel, 2017).

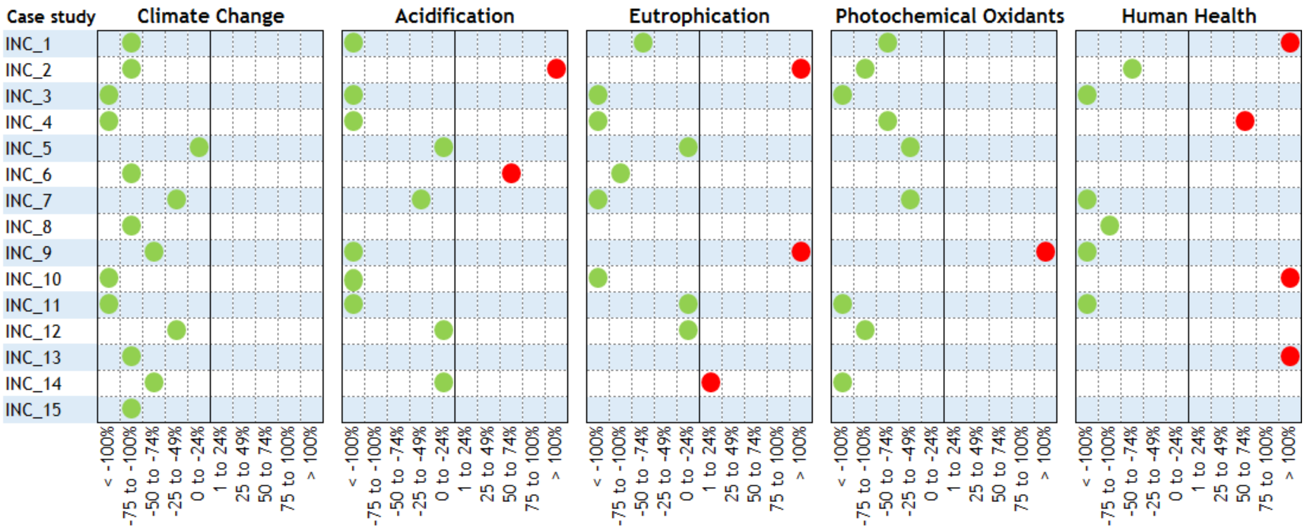

Figure 1-3. Environmental life cycle consequences of diverting residual waste and/or rejects from landfilling to incineration in different case studies. **Case studies**: INC_1 & INC_4: Eisted and Christensen (2013); INC_2: Erses Yay (2015); INC_3: Rajcoomar and Ramjeawon (2017); INC_5: Zhou et al. (2018); INC_6: Cherubini et al. (2009); INC_7: Belboom et al. (2013); INC_8: Fernández-Nava et al. (2014); INC_9: Coventry et al. (2016); INC_10: Sharma and Chandel (2017); INC_11: Starostina et al. (2018); INC_12: Chi et al. (2015); INC_13: Rajaeifar et al. (2015); INC_14: Yan Zhao et al. (2009); INC_15: Turner et al. (2016).

Implementation of separate collection and anaerobic digestion of organic waste

AD is becoming the preferred option to treat the organic fraction of MSW, chiefly due to its potential environmental benefits. In this regard, Figure 1-4 shows the environmental life cycle consequences of the simultaneous implementation of separate collection of organic waste followed by its AD in 7 case studies from the literature. In general, this WTE pathway can reduce the environmental impacts of MSW treatment. This is particularly true when the reference scenario consists of the disposal of organic waste in landfills (case studies SSAD_1 and SSAD_2). In this context, savings are due to (1) the reduction of CH_4 emissions from landfills and (2) energy recovery from biogas. Several case studies, however, concluded marginal benefits or even adverse impacts if the reference scenario consists of the collection of the organic waste along with residual waste and its incineration in a highly efficient facility (case studies SSAD_3 to SSAD_7). The reasons behind this conclusion are that (1) the incineration of organic waste along residual waste generates more energy, thus avoiding more impacts through substitution, and (2) AD involves CH_4 emissions due to reactors' leakage and digestate storage and N_2O emissions due to digestate application on land. It is important to remark that the availability of a highly efficient incineration facility may not be representative for all the geographical contexts.

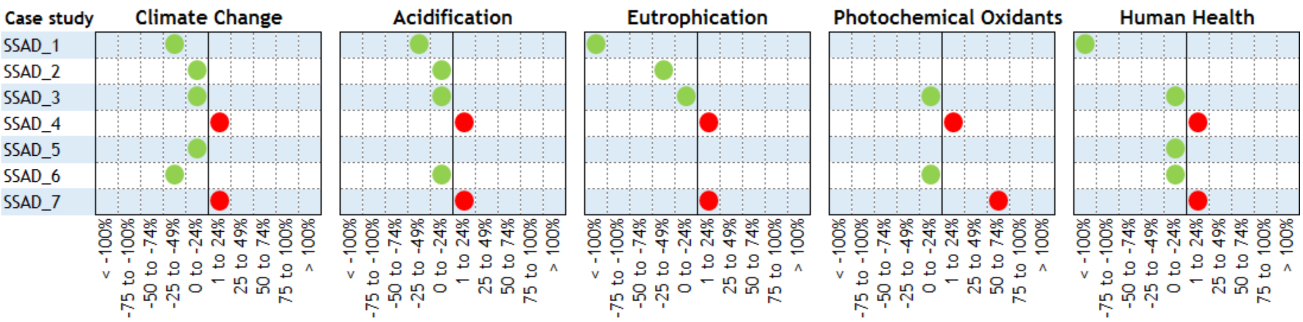

Figure 1-4. Environmental life cycle consequences of implementing the separate collection of organic waste followed by anaerobic digestion in different case studies. **Case studies**: SSAD_1: Sharma and Chandel (2017); SSAD_2: Liikanen et al. (2018); SSAD_3: Slagstad and Brattebø (2012); SSAD_4: Belboom et al. (2013); SSAD_5: Fernández-Nava et al. (2014); SSAD_6: Grosso et al. (2012); SSAD_7: Stanisavljevic et al. (2017).

<u>Diversion of organic waste from composting to anaerobic digestion</u>

Among the alternative treatments for organic waste, composting is the main competitor of AD. In this regard, Figure 1-5 shows the environmental life cycle consequences of diverting organic waste from composting to AD in 8 case studies. Practically all the case studies agree that this WTE pathway would reduce the environmental impacts associated with MSW treatment. The environmental benefits come primarily from the use of biogas for energy recovery. In this regard, the major difference between the two options is that both involve nutrient recovery (and substitution of mineral fertilizers) but only AD involves energy recovery.

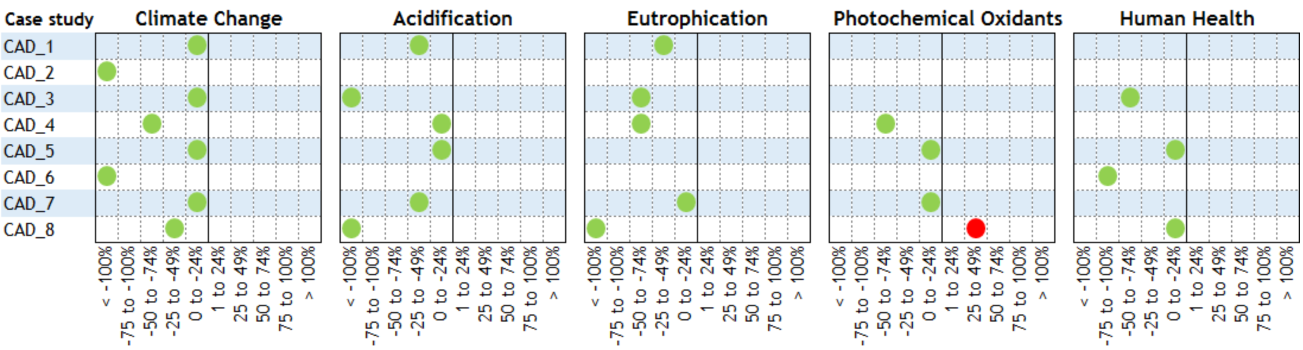

Figure 1-5. Environmental life cycle consequences of diverting organic waste from composting to anaerobic digestion in different case studies. **Case studies**: CAD_1: Liikanen et al. (2018); CAD_2: Friedrich and Trois (2016); CAD_3: Sharma and Chandel (2017); CAD_4: Chi et al. (2015); CAD_5: Giugliano et al. (2011); CAD_6: Rajaeifar et al. (2015); CAD_7: Bovea et al. (2010); CAD_8: Stanisavljevic et al. (2017).

Gasification and pyrolysis pathways

None of the reviewed LCA studies addresses the implementation of emerging gasification and pyrolysis WTE pathways. This is understandable since the review was restricted to real case studies and the penetration level of gasification and pyrolysis in the MSW management system is still low. There are however a few LCA studies addressing MSW gasification and pyrolysis from a technology perspective. Some of these studies reported an overall reduction in the environmental impacts of these technologies compared with incineration (Dong et al., 2018b; Voss et al., 2021). Others found that MSW gasification is an effective pathway to reduce the impacts on acidification, eutrophication, and photochemical ozone formation, but not the climate change impacts (Tang et al., 2020). Others, in contrast, concluded that modern incineration facilities equipped with advanced APC systems outperform both gasification and pyrolysis (Arena et al., 2015; Dong et al., 2018a). These discrepancies have been also highlighted in the review carried out by (Dong et al., 2018a; Mayer et al., 2019), who concluded that there is not yet sufficient evidence about the environmental preference of gasification and pyrolysis over incineration.

General overview of the life cycle environmental consequences of WTE pathways

The case studies reviewed show that energy recovery generally has large consequences for the environmental performance of the MSW management system. The environmental outcome can vary substantially among case studies, either in terms of magnitude or sense (i.e., impact increase or reduction). For example, the average reduction in GHG emissions due to the diversion of waste from landfilling to incineration equals 117%, with a range from 6% to 489% (Figure 1-3). This variability is even more pronounced in the case of human health, ranging from an impact increase of 125,490% to a reduction of 946%. The disparity between case studies reflects the high dependence of the environmental performance of MSW treatment on issues such as scenario formulation, waste composition, technology, energy system, climatic conditions, and methodological choices and assumptions. Overall, regionalization is paramount to determine the environmental performance of MSW management, and caution is needed when extrapolating results to other geographical contexts (Bernstad and Jansen, 2012; Cleary, 2009; Laurent et al., 2014a).

Among all the factors influencing the environmental performance of WTE, the composition and physicochemical properties of the waste are often identified as the most critical. Lausselet et al. (2016), for example, found that the environmental performance of incineration is primarily driven by the waste LHV and its content on biogenic carbon and heavy metals. In fact, Bisinella et al. (2017) demonstrated that waste composition influences the environmental performance of incineration more than technology parameters. AD is also especially affected by waste composition, in particular the level of impurities, i.e., the contamination of the organic waste with unwanted materials such as plastic and metal (Di Maria et al., 2020a). High levels of impurities result in poor methane yield, low energy recovery, low quality of digestate/compost, and, overall, reduced environmental benefits.

Another determinant factor is the magnitude of the impacts generated by the substituted energy source (Fruergaard et al., 2009). The most common practice is to assume that electricity from MSW substitutes an equivalent amount of electricity from the national mix (Anshassi et al., 2021). Thus, the impacts avoided by WTE can vary substantially and tend to decrease in countries with high shares of renewables. In the EU, for example, the carbon footprint of electricity ranges from 0.765 kg CO_2/kWh in Greece to 0.033 kg CO_2/kWh in Sweden (Scarlat et al., 2022). Alternatively, electricity from MSW is assumed to substitute individual sources, such as electricity from natural gas or coal (Jeswani and Azapagic, 2016). This decision could entail large implications if the impacts of the individual source deviate significantly from those of the average national mix. For example, the carbon footprint of coal-based electricity ranges from 0,675 to 1.689 kg CO_2/kWh (Whitaker et al., 2012) while the average carbon footprint of electricity in the EU is 0,296 kg CO_2/kWh (Scarlat et al., 2022). In the case of CHP, the heat recovered is usually assumed to substitute an equivalent amount of heat produced with natural gas. In this regard, WTE facilities with CHP tend to outperform facilities producing electricity only (Eriksson and Finnveden, 2017).

1.2.3 Economics of waste-to-energy

The planning of MSW management is carried out by or for municipalities usually under a constrained budget (UNEP, 2015). Thus, although WTE has the potential of reducing environmental impacts, the relatively high costs are often a major barrier.

Incineration is generally more expensive than landfilling, MRFs, composting, and AD, but cheaper than gasification (Table 1-1). The main contributor to the costs of incineration is the advanced APC system. According to the BAT document, the capital costs[†] of incineration range from 382 EUR/annual tonne for large-scale facilities (600,000 tonnes/year) to 726 EUR/annual tonne for medium-scale facilities (100,000 tonnes/year) (Neuwahl et al., 2019). In absolute terms, the capital costs range from 73 million EUR for a medium-scale facility to 229 million EUR for a large-scale facility. Other sources reported similar capital costs: 512-853 EUR/annual tonne (Kaza and Bhada-Tata, 2018), 542-813 EUR/annual tonne (World Energy Council, 2016), and 368-547 EUR/annual tonne (Yassin et al., 2009). Meanwhile, the capital costs of the few commercial gasification facilities dedicated to

[†] All the economic data presented in this section were harmonized to EUR2019.

generating electricity from MSW range from 746 to 1,561 EUR/annual tonne (IEA Bioenergy, 2018b). The net operating costs of incineration (including revenues from the sale of energy) were reported in the range from 21 to 26 EUR/tonne (Kaza and Bhada-Tata, 2018).

Table 1-1. Capital and operating costs and gate fees of municipal solid waste treatment options within the European context.

Technology	**Capital costs [EUR/annual tonne] (1)**	**Operating costs [EUR/tonne] (2)**	**Gate fees [EUR/tonne] (3)**
Sanitary landfill	28–55 (a)	6–26 (a)	4–150 (i)
			2–84 (j)
Material recovery facilities	206–287 (b)	26–33 (b)	21–137 (b)
			33–99 (j)
Composting	73–114 (c)	10 (a)	
Anaerobic digestion	294–667 (d)	26–49 (a)	0–63 (j)
	294–512 (a)		
Incineration	382–726 (e)	21 – 26 (a)	38–350 (e)
	512–853 (a)		42–132 (j)
	542–813 (f)		
	368–547 (g)		
Gasification	746 – 1,561 (h)	NA	NA
Pyrolysis	NA	NA	NA

Note: [1] Capital costs refer to the amount of money divided by the annual processing capacity of the facility; [2] Operating costs refer to the amount of money per tonne of waste treated, including revenues; [3] Gate fees refer to the amount of money charged per tonne of waste received at the facility, excluding taxes.
Source: [a] Kaza and Bhada-Tata (2018); [b] Cimpan et al. (2016); [c] Edwards et al. (2018); Mayer et al. (2020); Shahid and Hittinger (2021); Slorach et al. (2020); van Haaren et al. (2010); [d] Angelonidi and Smith (2015); [e] Neuwahl et al. (2019); [f] World Energy Council (2016); [g] Yassin et al. (2009); [h] IEA Bioenergy (2018b); [i] European Environment Agency (2022); [j] WRAP (2021)

AD is more expensive than its principal competitor, namely composting. Angelonidi and Smith (2015) found that the capital costs of nine European AD facilities range from 294 to 667 EUR/annual tonne. Other sources, such as The World Bank, reported similar capital costs (295-512 EUR/annual tonne) (Kaza and Bhada-Tata, 2018). Meanwhile, the capital costs of composting as reported in the literature barely reach 73-114 EUR/annual tonne (Edwards et al., 2018; Mayer et al., 2020; Shahid and Hittinger, 2021; Slorach et al., 2020; van Haaren et al., 2010). AD is also more expensive in terms of net operating costs, between 26 and 49 EUR/tonne against 10 EUR/tonne for composting (Kaza and Bhada-Tata, 2018). The higher costs of AD are due to the pre-treatment needed to remove unwanted materials, the energy conversion system (i.e., CHP units or upgrading), the higher consumption of raw materials and energy, the higher maintenance cost, and, overall, the higher complexity of the process.

The gate fee charged by WTE facilities generally exceeds those charged by landfills, making the former economically less attractive. The median gate fee reported for incineration in the EU is 80 EUR/tonne (range 38-350 EUR/tonne) (Neuwahl et al., 2019), which is very similar to the 82

EUR/tonne (42-132 EUR/tonne) reported in the UK (WRAP, 2021). For AD, the median gate fee reported by UK authorities is 31 EUR/tonne (0-63 EUR/tonne). Meanwhile, the median gate fee of landfills ranges from 22 EUR/tonne (2-84 EUR/tonne) in the UK to 44 EUR/tonne (4-150 EUR/tonne) in EU (European Environment Agency, 2022).

Many factors determine the economic feasibility of WTE projects. Feedstock availability is critical. As explained above, incineration is more attractive at high treatment capacities due to economies of scale (Tsilemou and Panagiotakopoulos, 2006). To ensure the economic feasibility of an incineration facility, the supply of a minimum of 100,000 tonnes/year of feedstock over the facility's lifetime is generally recommended (World Energy Council, 2016). For example, the average MSW incineration capacity in Europe ranges from 60,000 tonnes/year (Norway) to 488,000 tonnes/year (The Netherlands), with an overall average of 193,000 tonnes/year (Neuwahl et al., 2019). Feedstock availability is also a major challenge faced by AD facilities. Nowadays, a major barrier to the economic feasibility of AD is the lack of access to large quantities of relatively pure organic waste feedstocks, chiefly due to the low rate of separate collection (De Clercq et al., 2017; Edwards et al., 2015; Satchwell et al., 2018).

The revenues received for the sale of energy have also a large impact on the economic feasibility of WTE projects. The revenues generated depend on multiple factors, including the amount of energy delivered, the energy products obtained (e.g., electricity, heat, or both), and the selling price (Haraguchi et al., 2019). The amount of energy delivered depends on the process energy recovery efficiency, but also on the availability and properties (energy content) of feedstock.

1.3 The role of waste-to-energy in a circular economy

The previous section discussed the pivotal role of energy recovery for MSW treatment. Special emphasis has been placed on the decisive role of feedstock availability and composition in the environmental sustainability (Section 1.2.2) and techno-economic feasibility (Section 1.2.3) of WTE technologies. Looking to the future, the rise of the circular economy, where waste materials tend to a higher recirculation through the separate collection, re-use, and recycling, is likely to bring long-term challenges for the WTE sector (European Commission, 2017). This thesis is motivated by the need to carry out a robust analysis of such challenges. More precisely, this thesis points to:

(1) understanding the future energy recovery potential of MSW,

(2) understanding the environmental consequences of phasing-out incineration in the MSW management system, and

(3) prioritizing economic and/or environmental optimal WTE pathways in the context of a circular economy.

1.3.1 Energy recovery potential of municipal solid waste

The economic and environmental performance of WTE technologies is highly sensitive to the availability and characteristics of feedstocks. Furthermore, feedstock determines facility location, treatment capacity, and technology design among others. The causal loop diagram in Figure 1-6 shows some of the potential interrelationships between the circular economy and energy recovery.

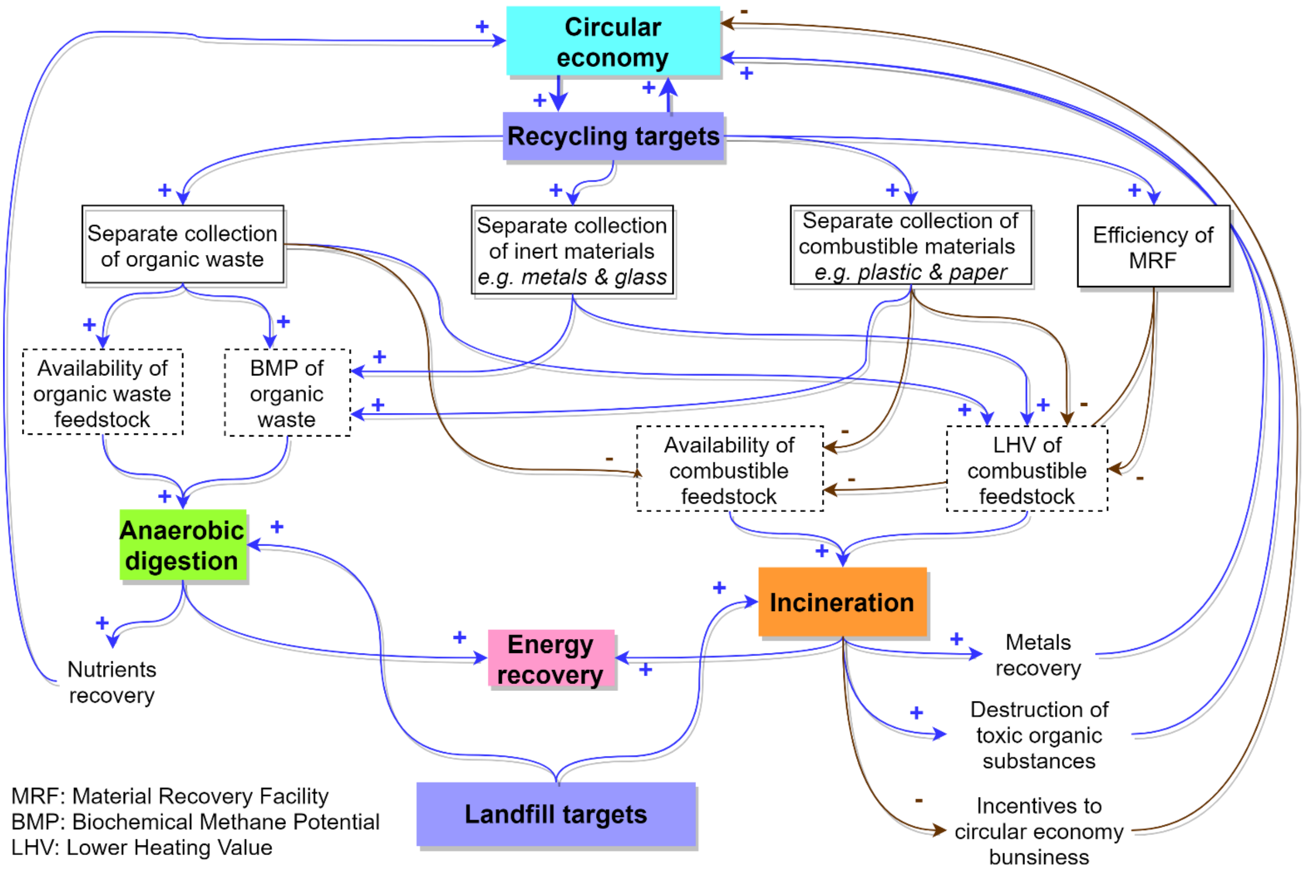

Figure 1-6. Causal loop diagram illustrating some potential interrelationships between the circular economy and energy recovery (anaerobic digestion and incineration). Positive links [+] indicate that an increase/decrease in the causal variable results in an increase/decrease in the effect variable. Negative links [-] indicate that an increase/decrease in the causal variable results in a decrease/increase in the effect variable.

Recycling targets and mandatory separate collection are the main EU waste policy instruments to support the transition to a circular economy (*cf.* Section 1.1). One could easily imagine that the realization of these policy goals will have a certain impact on the energy recovery potential of MSW. For example, the diversion of an ever-increasing proportion of combustible materials such as paper, cardboard, plastic, and textile towards recycling (through the separate collection and more efficient MRFs) can reduce the amount of residual waste and rejects and their energy content. This may in turn negatively affect incineration due to a decrease in the availability and quality (energy content) of feedstock (Saveyn et al., 2016). In contrast, the obligation to separate organic waste may encourage the expansion of AD since more feedstock will become available (De Clercq et al., 2017; Satchwell et al., 2018; Scarlat et al., 2018). There arise many other cross-linked effects. Increasing the separate collection of non-biodegradable materials, such as plastic, glass, and metal, prevents these from contaminating the organic waste feedstock received at AD facilities. At the same time, increasing the separate collection of non-combustible materials (e.g., glass and metal) and wet organic waste prevents these from lowering the energy content of the waste feedstocks directed to incineration. All in all, understanding the future energy recovery potential of MSW streams, and how this potential is affected by a higher circularity of materials, is paramount to define a role for WTE in the circular economy.

1.3.2 To incinerate, or not to incinerate

The convenience of incineration within the circular economy remains highly controversial. Some hold that incineration is compatible and complementary with the circular economy (Brunner and Rechberger, 2015; ESWET, 2020; Van Caneghem et al., 2019). The main argument in support of incineration is that this is the only mature and competitive technology to valorize the residual waste and rejects that otherwise would end up in landfills (ESWET, 2020). There are three undisputed facts about waste prevention and recycling: (1) economic activity will always generate waste no matter waste prevention measures, (2) materials cannot be recycled infinitely due to quality losses and cross-contamination, and (3) not all the materials can be recycled (Arena, 2015; Quicker et al., 2020; Van Caneghem et al., 2019). In consequence, neither separate collection nor MRFs is likely to achieve 100% recovery efficiency. The average separate collection rate of recyclable materials (paper/cardboard, plastic, glass, and metal) in European capitals barely reached 25% in 2015 (Seyring et al., 2016). The estimated separate collection rates required to realize the 2030 recycling targets for packaging materials are approximately 86% for paper/cardboard, 76% for plastic, 64% for metal, and 94% for glass (Tallentire and Steubing, 2020). MRFs also show limited efficiency. European MRFs for plastic packaging waste reject on average 19% of PET, 24% of HDPE, 42% of films, and 43% of PP (Antonopoulos et al., 2021). Furthermore, MRFs for residual waste can reject up to 89-93% of the input waste (Cimpan et al., 2015a). Within this context, some studies determined that incineration would continue to be necessary to realize the 10% landfill goal by 2035 (CEWEP, 2019; Di Foggia and Beccarello, 2021).

Conversely, others express concern that incineration may prevent recycling and the development of circular economy business models. For example, a report alerted that Nordic countries are unlikely to meet the EU recycling targets if they maintain the current high dependence on incineration (Papineschi et al., 2019). The European Commission has also stressed that "*such high rates of incineration are inconsistent with more ambitious recycling targets*" (European Commission, 2017). Furthermore, incineration is continuously linked to impacts on the environment and human health, despite a drastic reduction in emissions over the last decades due to more stringent regulation and continuous improvements in the APC system (Ardolino et al., 2020; Damgaard et al., 2010). In particular, incineration has been associated with potential impacts on human health due to the release of toxic compounds (Allsopp et al., 2001). Several of the LCA case studies reviewed in Section 1.2.2 identified the release of dioxins, furans, and heavy metals from incineration as the main source of impacts on human health.

Within this context, an increasing number of organizations and groups, such as the Global Alliance for Incinerator Alternatives (GAIA, 2013) and Zero Waste Europe (Zero Waste Europe, 2022), consider the phase-out of incineration as a way to protect the environment and human health and to promote the transition to a more circular economy. However, the potential environmental benefits of eliminating incineration remain highly uncertain considering that such a decision could intensify landfilling. Thus, phasing-out incineration will certainly avoid the associated life cycle environmental impacts but may lead to an increase in the impacts from landfilling. Furthermore, it is still unclear to what extent improvements in separate collection and recycling can mitigate the negative impact that incineration phase-out might have on the landfill rate. All in all, phasing-out MSW incineration is a strategy that is gaining momentum in the circular economy debate, but whose environmental consequences have not been fully assessed yet.

1.3.3 Optimal waste-to-energy pathways in a circular economy

Potential changes in waste flows resulting from an increased circularity of materials will affect WTE at all levels, from capacity deployment and use to economic feasibility and environmental sustainability. Thus, WTE pathways should be carefully planned considering the future availability and characteristics of waste streams as well as changes in the background system (i.e., the surrounding economic sectors, such as power generation) to avoid potentially bad decisions and to guarantee the overall best environmental outcome (European Commission, 2017).

Particular attention should be paid to the availability of feedstock to sustain the operation of existing and/or new incineration facilities over their entire lifetime (between 20 and 30 years). Concern has been expressed about the risk of incineration overcapacity at the local level due to a shortage in feedstock availability (European Commission, 2017; Jofra and Ventosa, 2013; Wilts and Von Gries, 2015). In this context, investment in extra incineration capacity should only be done in duly justified situations. Furthermore, the waste hierarchy establishes that stakeholders must opt for technologies that maximize the recovery of materials and energy. This calls for a comprehensive analysis of the potential replacement of older facilities with new more efficient facilities that combine energy with ferrous (steel) and non-ferrous (aluminium) metals recovery (Abis et al., 2020; Brunner and Rechberger, 2015). Metals recovery is increasingly viewed to play a pivotal role in the future economics and environmental performance of incineration (IEA Bioenergy, 2019).

Future changes in the composition of waste streams will also affect the economic and environmental performance of incineration. Section 1.3.1 and Figure 1-6 have focused primarily on illustrating the potential impact of separate collection and recycling targets on the energy content of waste streams. Not only the energy content may be altered, but also other physicochemical properties that determine emissions, such as the carbon content. In particular, the relatively high share of biogenic carbon in the waste fed to incineration brings significant environmental benefits, since biogenic CO_2 is usually considered climate neutral (Christensen et al., 2009). Currently, up to 85% of the carbon content in the waste fed to incineration is considered biogenic (Astrup et al., 2009). However, an increasing separate collection of biogenic-based materials, such as food, paper, cardboard, and textile, can reduce this share in favor of fossil carbon (Bisinella et al., 2021). This will in turn afect the potential climate benefits which traditionally have been assigned to incineration with energy recovery (*cf.* Figure 1-3). The transition to a more renewable energy system is likely to further exacerbate the potential environmental impacts of incineration (Istrate et al., 2019; Pizarro-Alonso et al., 2018).

Regarding AD, the implementation and/or improvement of separate collection of organic waste should result in greater availability of feedstock (European Commission, 2017). The current average rate of food waste separate collection in Europe is 26%, while the maximum technical potential was estimated to be 85% (Favoino and Giavini, 2020). Thus, the potential for energy recovery through AD can increase substantially in the future. However, since AD is a relatively expensive technology (*cf.* Section 1.2.3), the deployment of new capacity requires a holistic perspective to guarantee the most sustainable choice. An assessment of the long-term optimal AD pathways should include competing technologies, such as composting and incineration (Slorach et al., 2020), alternative energy recovery systems, such as electricity generation and biomethane upgrading (Pérez-Camacho et al., 2019), the impact of energy system decarbonization (Styles et al., 2022), and the disposal of rejects from pre-treatment (Carlsson et al., 2015).

1.4 Methodological background

Developing MSW management plans is a challenging task for decision-makers (UNEP, 2015). The heterogeneous nature of MSW, the complexity of the MSW management system, and the lack of reliable monitoring practices often hamper the identification of the solutions best suited to each situation (Dri et al., 2018; Zeschmar-Lahl et al., 2016). The transition to a more circular economy is likely to amplify the complexity of the problem. In this context, addressing the long-term challenges faced by the WTE sector requires developing tailored tools capable of tackling the increasing complexity of MSW and anticipating system-wide consequences. A broad variety of decision support tools exist for the assessment of MSW management systems, including material flow analysis (MFA), life cycle assessment (LCA), cost-benefit analysis, life cycle costing, risk assessment, environmental impact assessment, multi-criteria analysis, and mathematical optimization (Chang et al., 2011). In this thesis, MFA, LCA, and mathematical optimization were found to be suitable to address the points presented in Section 1.3.

1.4.1 Material flow analysis

MFA is a methodology to perform a systematic inventory of the flows and stocks of materials and/or substances in a system within defined spatial and temporal boundaries (Brunner and Rechberger, 2017). MFA follows the law of conservation of mass: input quantities equal output quantities plus stocks. MFA has been widely applied to MSW management to quantify aggregated indicators such as recycling rate (Turner et al., 2016), to track the fate of key substances including carbon, nutrients, and heavy metals (Allesch and Brunner, 2017; Jensen et al., 2017), and to perform energy balances (Tonini et al., 2014). Most MFA studies on MSW management address past and/or current situations (Elgie et al., 2021; Vujovic et al., 2020). MFA combined with scenario analysis has been suggested elsewhere to systematically evaluate future changes in waste flows (Allesch and Brunner, 2017; Eriksen et al., 2020; Klotz et al., 2022). In this context, the parametrization of the MFA model is a key modelling feature to allow predicting future mass and energy flows. This requires the development of a mathematical framework to capture all the relationships between material and substance flows. Only a limited number of studies have developed parametrized MFA models for waste treatment systems, including MRFs (Kleinhans et al., 2020) and waste electrical and electronic equipment management (De Meester et al., 2019). The combination of parametrized MFA and scenario analysis is particularly suitable to quantify the future energy recovery potential of MSW.

1.4.2 Life cycle assessment

LCA is by far the most popular tool for supporting MSW management decisions (Allesch and Brunner, 2014; Cobo et al., 2018a; Pires et al., 2011). LCA is a standardized (ISO, 2006a, 2006b) and well-established methodology to quantify the potential environmental impacts of products, systems, and services over their entire life cycle, from raw materials extraction to end-of-life (Hellweg and Canals, 2014). The LCA methodology consists of four phases: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation (Box 1.4) (European Commission, 2010). A review of LCA studies on MSW management systems is provided in Section 1.2.2. Here, existing LCA tools for waste assessment are reviewed and modelling gaps are identified.

Box 1.4 Terms and concepts: phases of life cycle assessment

(1) **Goal and scope definition**: The objective, context, intended application, and the targeted audience must be defined in the first phase of an LCA study. Scope definition involves the description of the system under analysis, system boundaries, the functional unit (reference to which all the inputs and outputs are mathematically referred), allocation, impact categories, impact assessment method, data requirements, assumptions, and limitations.

(2) **Inventory analysis**: The second phase involves developing a life cycle inventory (LCI) of all relevant inputs and outputs of the system under analysis. Thus, data must be collected (e.g., from measurements, calculations, and literature) for each unit process within the system boundaries.

(3) **Impact assessment**: During the life cycle impact assessment (LCIA) phase, the environmental flows included in the LCI are aggregated into impact categories.

(4) **Interpretation**: In the last phase, the results from the LCI analysis and the LCIA are interpreted and discussed in relation to the goal and scope.

Several waste-LCA tools have been developed over the last two decades. Some examples are EASETECH developed at the Technical University of Denmark (Clavreul et al., 2014), SWEET developed by the U.S. Environmental Protection Agency (US EPA, 2018), WRATE developed by Golder Associates (Golder Associates, 2014), CO2ZW (Sevigné Itoiz et al., 2013), LCA-IWM (den Boer et al., 2007), and ORWARE (Eriksson et al., 2002). A complete review and comparison of available waste-LCA tools is available in Gentil et al. (2010) and Anshassi and Townsend (2021).

From a modelling perspective, the capability of connecting the life cycle inventory (LCI) of waste treatment processes (i.e., emissions, resources consumption, and materials and energy recovery) to the composition and physicochemical properties of the waste (e.g., energy, carbon, nutrients, and heavy metals content) is recognized as the cornerstone of any waste-LCA tool (Gentil et al., 2010). This sounds reasonable given the decisive influence of waste composition on the environmental performance of waste treatment processes (*cf.* Section 1.2.2) and its high variability across time and space. Waste-LCA tools and studies have a different degree of detail in modelling waste treatment processes. Often, they rely on simple black-box models where outputs (e.g., emissions and energy recovery) are linked by fixed ratios to the total mass of waste treated (Friedrich and Trois, 2016; Ripa et al., 2017). This approach, however, neglects any dependence on waste composition and properties and may not be sufficient to perform assessments in the context of increasingly circular MSW management systems. Recently, increased attention is being devoted to the development of waste-LCA frameworks that allow tracking the flow of individual waste materials and their physicochemical properties and linking these to the LCI (Lodato et al., 2021, 2020). This can be achieved by integrating LCA and MFA. In such an integrated framework, MFA provides the appropriate mathematical relationships that describe the flow of materials, chemicals, and energy through the MSW management system, from collection to final disposal. Thus, MFA provides the basis to operationalize LCIs that are fully responsive to the composition and physicochemical properties of the waste.

MFA and LCA were combined by Turner et al. (2016), to evaluate the national MSW management strategy of Wales, by Haupt et al. (2018a) to evaluate alternative MSW management scenarios for Switzerland, by Wang et al. (2022), to retrospectively evaluate the evolution of MSW management in Nottingham (UK), and by Sevigné-Itoiz et al. (2015a, 2015b, 2014), to evaluate the performance of aluminium, paper, and plastic recycling in Spain. In general, MFA and LCA are combined in a sequential process: a pre-defined MSW management scenario is first assessed with MFA, and the results are used to perform the LCA. More advanced tools can be developed if MFA fundamentals are fully integrated into the LCA framework. (Lodato et al., 2021, 2020) have recently presented the integration of advanced MFA approaches into the EASETECH waste-LCA tool.

1.4.3 Mathematical optimization

MFA and LCA are powerful descriptive tools that can inform decision-makers about the performance of alternative waste management scenarios. The scenarios must be defined a priori, and their number is generally limited between 3 and 24 (Tascione and Raggi, 2012). However, modern integrated management systems can involve many waste streams and treatment processes, resulting in a network with several hundreds of possible interconnections. Furthermore, system restrictions (e.g., capacity constraints), waste policy (e.g., landfill targets), and the dynamics of waste generation and composition make the assessment of all relevant scenarios practically infeasible. Strictly speaking, there are infinite possible scenarios. Within this context, mathematical optimization is a powerful tool that simultaneously addresses all possible waste treatment pathways to determine the best solution according to one or multiple criteria while satisfying the system’s constraints and restrictions.

Mathematical optimization consists of (1) developing a mathematical model that describes the system under analysis and (2) using an algorithm (solver) to determine the optimal solution to the problem. The mathematical model consists of an objective function (or several in the case of multi-objective optimization) and several constraints that impose bounds on the decision variables. The solution of the optimization model corresponds to the values of the decision variables (e.g., capacities of waste treatment facilities and mass of waste treated by each facility) that optimize the objective function (e.g., the cost of the system). A generic optimization model can be expressed in compact form as follows (Vanderbei, 2017):

$$\begin{aligned} \max_{x,Y} \quad & f(x,Y) \\ s.t. \quad & h(x,Y) = 0 \\ & g(x,Y) \leq 0 \\ & x \in \mathbb{R}, Y \in \{0,1\} \end{aligned}$$

where f is the objective function (e.g., costs or environmental impacts), x denotes continuous variables (e.g., mass and energy flows or capacities), and Y represents binary variables that describe discrete decisions, such as whether to invest in a specific technology or not. The inequality and equality constraints are the mathematical equations that describe the behavior of the system and include, for example, mass and energy balances, capacity constraints, economic assessment, and life cycle impact assessment.

Depending on the number of objective functions, optimization models can be classified as single-objective or multi-objective. Furthermore, depending on the nature of the objective function and constraints, optimization models can be classified as linear programming (LP) or nonlinear programming (NLP). LP involves only continuous variables and linear equations. NLP implies that the objective function and/or at least one constraint is a non-linear function, while all the variables are continuous. Optimization models that involve both continuous and discrete variables are classified as mixed-integer linear programming (MILP) or mixed-integer nonlinear programming (MINLP). Nonlinear models may be either convex or nonconvex. An optimization model is convex if the constraints are convex functions, and the objective function is convex if minimizing or concave if maximizing. Convex optimization models have a unique global optimum solution. On the other hand, an optimization model is nonconvex if the objective function or any of the constraints are nonconvex. Nonconvex optimization models may have multiple local optimum solutions, which makes the determination of the global optima a hard task (Floudas, 2000).

Several waste optimization models have been developed for a broad range of goals and scopes, including waste collection routes, waste treatment facilities location, capacity use and expansion, and waste allocation (Ghiani et al., 2014; Juul et al., 2013). The goal is typically to minimize the cost of the system (Garibay-Rodriguez et al., 2018; Rizwan et al., 2018; Shahid and Hittinger, 2021), but increased attention is being devoted to minimizing the environmental impacts through the integration of LCA and optimization (Cobo et al., 2019; Mavrotas et al., 2013). Several models incorporate a multi-period decision horizon to deal with the dynamics of waste generation (Batur et al., 2020; Roberts et al., 2018), landfill capacity (Garibay-Rodriguez et al., 2018; Santibañez-Aguilar et al., 2017), and the decarbonization of electricity generation (Levis et al., 2013).

Existing optimization models play an essential role in understanding the pathways towards more sustainable MSW management systems (Cobo et al., 2018a). However, the future poses new challenges that demand new modelling features. First, most models focus solely on new infrastructure, thus disregarding the extended use and/or retrofitting of existing facilities (Ng et al., 2014; Shahid and Hittinger, 2021; Tan et al., 2014). The extension of the lifetime of existing incinerators, for example, may present economic advantages, but at the same time lead to the operation of older, less efficient facilities (Levis et al., 2013; Roberts et al., 2018). The implications of these trade-offs remain largely unexplored. Secondly, economies of scale have been in general neglected. Capital and fixed operating costs were typically modelled with a fixed amount of money per unit of capacity (Levis et al., 2013; Ng et al., 2014). However, the capital and fixed unit operating costs of waste treatment processes tend to decrease at larger capacity (*cf.* Section 1.2.3). The energy efficiency of incineration also tends to increase at larger capacities (*cf.* Section 1.2.1). So, economies of scale can be of paramount importance if the future availability of feedstock forces to move towards smaller incineration facilities that might result economically uncompetitive.

Finally, just as in the case of waste-LCA tools, a recurring limitation of existing waste optimization models is the lack of response of treatment processes to waste composition and properties. Existing optimization models typically apply fix yields (e.g., kWh electricity per tonne waste) and emission factors (e.g., kg CO_2 per tonne waste), thereby ignoring any dependence on waste composition (Chang et al., 2012; Ooi et al., 2021; Yousefloo and Babazadeh, 2020). A more accurate representation would be achieved if processes outputs were endogenously calculated as a function of waste composition and the input-output relationships that govern the process. That is to say, integrating MFA principles into the optimization framework. While some work has already been done in this regard (a review is presented in Chapter 5), the incorporation of this fundamental feature into waste treatment optimization frameworks is still very scarce.

1.5 Case study of Madrid

Madrid is an interesting case study to address the long-term challenges faced by WTE since it is a representative example of a large and densely populated city with a modern and diverse MSW management system, but which is still struggling with low rates of separate collection and recycling and a high rate of landfilling.

In 2019, the municipality of Madrid treated approximately 1,394,105 tonnes of MSW, corresponding to 1,141,187 tonnes of household waste (82% of MSW), 116,140 tonnes of commercial waste (8%), and 136,777 tonnes of street cleaning waste (10%) (Figure 1-7a). With a population of 3.2 million (estimated at 3.4 million by 2030), these figures correspond to a per capita generation rate of 434 kg MSW/person-year. The composition of MSW in Madrid is similar to other cities within developed countries (Kaza et al., 2018), with a high share of organic waste (24%), paper/cardboard (24%), and plastic (14%) (Figure 1-7b).

Madrid established an integrated MSW management system (Madrid City Council, 2019). The household waste is collected separately into packaging (plastic, metal, and beverage cartons), paper/cardboard, glass, and organic waste, the rest being collected as part of the residual waste stream. The commercial waste managed by the municipality consists of organic waste collected separately and a stream of residual waste. The street cleaning waste consists of mixed waste (no separate collection). Following separate collection, the waste streams are treated through a broad range of interconnected facilities including MRFs, AD, composting, incineration, and landfilling. The packaging waste, paper/cardboard, and glass streams are delivered to MRFs for additional sorting before recycling. The organic waste collected separately is treated through AD. The household residual waste is delivered to MRFs where some materials are recovered for recycling and the food and green waste that remains in the residual stream is separated (i.e., residual organic waste) and distributed between AD, composting, and landfilling. The rejects generated by MRFs are incinerated and/or landfilled. Finally, the rejects generated by AD pre-treatment, the commercial residual waste, and the street cleaning waste are disposed of in landfill.

In 2019, only 30% of the household waste was collected separately, with the remaining 70% being collected in the residual waste. The percentage of MSW diverted to recycling, AD, and composting remained almost constant at 35% over the period 2015-2019, while the rate of incineration slightly increased from 21% to 24% and the rate of landfilling decreased from 44% to 42% (Figure 1-7c).

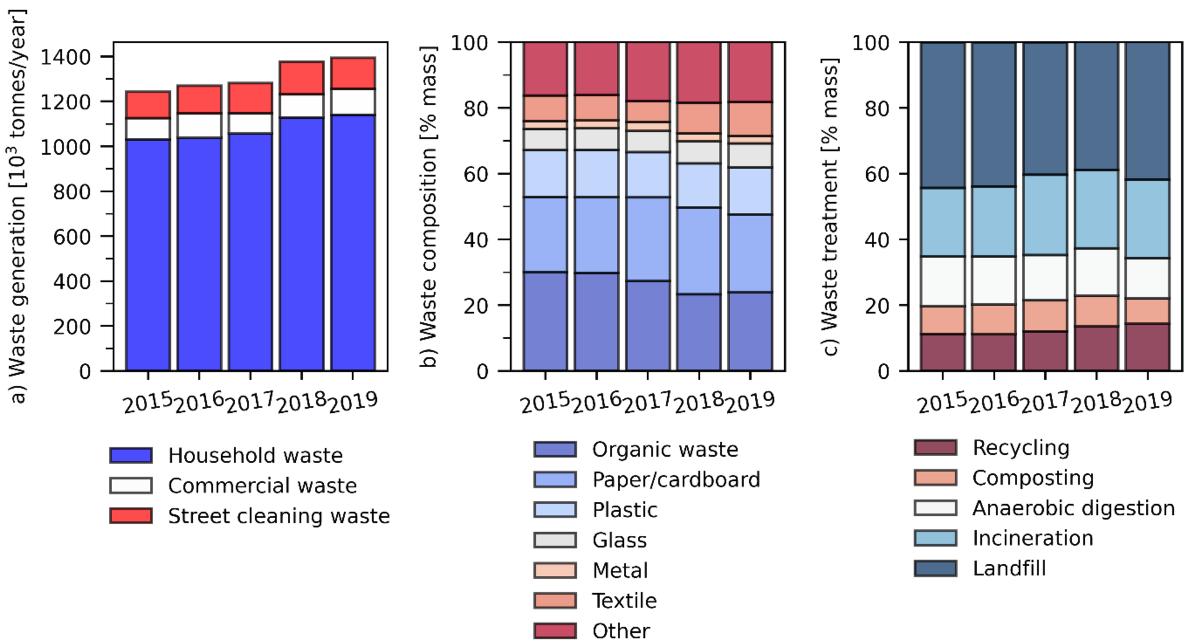

Figure 1-7. Municipal solid waste statistics in Madrid from 2015 to 2019. a) Mass of municipal solid waste treated, b) Waste composition, and c) Waste treatments shares.

Energy recovery is a fundamental part of MSW management in Madrid. The incineration facility of Madrid opened in 1993 and has a capacity of 328,500 tonnes waste/year. Over the period 2015-2019, this facility treated on average 300,000 tonnes/year of rejects from MRFs and fed into the electrical grid 146,000 MWh/year. There are two AD facilities, with capacity for 269,175 tonnes/year, dedicated to producing biomethane and electricity from the organic waste collected separately and some of the residual organic waste. Between 2015 and 2019, an average of 271,000 tonnes/year of organic waste was received at AD facilities, even though 32% (86,000 tonnes/year) was rejected at the pre-treatment. Therefore, 185,000 tonnes/year of organic waste was treated through AD producing between 5.8 and 9.3 million m^3 of biomethane which was injected into the natural gas network. Finally, some electricity is also generated through the collection and combustion of the LFG generated at the opened landfill site (7,962 MWh in 2019).

Madrid is a very interesting case study to the purpose of this thesis for many reasons. First and foremost, Madrid needs a notable effort to improve separate collection and recycling and reduce landfilling to meet the EU waste policy targets. This provides an excellent opportunity to evaluate the potential consequences of these objectives on energy recovery. Secondly, incineration is an essential part of the system, but it faces opposition from residents due to human health risk perception. In this regard, a new waste strategy published by the City Council in 2018 (declared invalid in 2019) planned to phase out the existing incineration facility by 2025. Thus, Madrid provides a real case study to assess the environmental consequences of eliminating incineration. Thirdly, the existing incineration facility was installed in 1993 and it is close to its end-of-life. Furthermore, the separate collection of organic waste has been recently implemented in 2017. This context provides the basis for the development of a framework to prioritize the optimal WTE pathways in the long term. Questions such as whether to decommission or replace the existing incineration facility or whether to expand the AD capacity have still to be addressed. Finally, the share of renewables in the Spanish electricity mix is expected to further increase in the future. This will significantly affect the potential environmental benefits of WTE technologies such as incineration and AD.

Chapter 2

Objectives

Objectives

Transitioning to a more circular economy calls for effective strategies to prevent municipal solid waste (MSW) generation, increase its re-use and recycling, and minimize landfilling. Realizing these objectives is inherently intended to cause large-scale effects on the waste-to-energy (WTE) industry that have still to be determined. The overarching aim of this thesis is to assess the energy recovery potential of MSW and the economic and environmental optimal WTE pathways within the context of a circular economy. This objective will be achieved by answering the following research questions (RQ):

RQ1 What will the energy recovery potential of MSW be under future scenarios of increased separate collection and recycling?

RQ2 What are the potential environmental consequences of phasing-out incineration in the MSW management system?

RQ3 What are the economic and/or environmental optimal WTE pathways that could be prioritized in light of the future availability and characteristics of MSW?

Giving scientifically sound answers to the stated research questions requires substantial modelling efforts. In this regard, this thesis develops new tailored tools capable of tackling the complexity of MSW and anticipating system-wide consequences. First, a material flow analysis (MFA) framework was developed to answer RQ1. Secondly, the MFA is combined with life cycle assessment (LCA) to answer RQ2. Finally, building on the previous tools, a comprehensive multi-period optimization framework for the systemic design of MSW treatment networks according to economic and environmental objectives is developed to answer RQ3. The research questions established are addressed through the case study of Madrid.

Chapter 3

Assessing the energy recovery potential of municipal solid waste under future scenarios

This chapter presents the work made to quantify the energy recovery potential of MSW under future scenarios of increased separate collection and recycling (**RQ1**)[‡]. Section 3.1 reviews previous works and methodological approaches in this field. Subsequently, Section 3.2 identifies the MSW streams susceptible to energy recovery in Madrid (hereinafter referred to as potential WTE feedstocks) and defines the scenarios for analysis. Section 3.3 proposes a framework for the systematic analysis of materials and energy flows through the MSW management system. Section 3.4 applies this framework to the case study of Madrid, while Section 3.5 contextualizes the main findings and outlines future research lines.

3.1 Introduction

The importance of understanding the future energy recovery potential of MSW has been highlighted in the first chapter. Several studies assessed the future energy recovery potential of MSW at the national (Burg et al., 2019; Gregg, 2010) and/or continental (Abis et al., 2020; CEWEP, 2019) scale. Within the European context, the Confederation of European Waste-to-Energy Plants (CEWEP) estimated an annual production of 46 million tonnes of household residual waste and 23 million tonnes of rejects by 2035 considering the 65% recycling target (CEWEP, 2019). Based on these results, many long-term opportunities for MSW incineration were envisioned. In a similar study, Abis et al. (2020) determined that the amount of MSW susceptible to incineration in Europe is not likely to drastically decrease by 2030 (3% decrease with respect to 2018).

While the studies mentioned above have some interesting findings, it is important to note that MSW is a regional issue. The availability and characteristics of MSW and the management system can vary significantly across continents and countries. In Spain, for example, municipalities use up to four alternative MSW collection schemes (Gallardo et al., 2010) and the rate of separate collection ranges from 9.6% to 33.6% (Sastre et al., 2018). Other local-specific variables are waste generation and composition and technological choices. For example, in Northern Europe the residual waste is typically incinerated, whereas in Southern Europe it is more frequently sorted at MRFs (Cimpan et al., 2015a). All in all, assessments based on national and/or continental aggregated data should not be extrapolated to decision-making at the municipal scale.

Focusing on the municipality/province level, some previous works have quantified the future availability of residual waste for energy recovery through incineration (Gu et al., 2021; Scarlat et al.,

[‡] Based on: Istrate, I.R., Medina-Martos, E., Galvez-Martos, J.L., Dufour, J., 2021. Assessment of the energy recovery potential of municipal solid waste under future scenarios. Applied Energy 293: 116915

2015; Shapiro-Bengtsen et al., 2020) and food waste for AD (Cudjoe et al., 2021). Nevertheless, modern integrated MSW management systems cover a broad range of potential WTE feedstocks, such as organic waste collected separately, residual organic waste, rejects, RDF, commercial waste, and street cleaning waste. Furthermore, the consequences of an increasing separate collection and recycling on the energy recovery potential has received only limited attention. Gu et al. (2021) is the only study that attempted to evaluate the impact of separate collection targets, focusing on the case study of Beijing. They found that the separate collection of all non-combustible materials and 90% of food waste reduces the mass of feedstocks available for incineration by 66%, while the LHV could increase from 5.7 to 14.5 MJ/kg. This study, however, focused exclusively on the residual waste and relied on theoretical assumptions, such as perfect separate collection.

The future energy recovery potential of MSW has been typically estimated from socio-economic trends. Scarlat et al. (2015), for example, relied on the methodology proposed by The World Bank, which projects MSW generation per capita based on the gross domestic product (GDP) (Kaza et al., 2018). Subsequently, the authors quantified the energy recovery potential by assuming a uniform waste LHV of 9 MJ/kg. Cudjoe et al. (2021) predicted food waste generation from population growth rate and GDP and assessed the energy recovery potential of AD with the Buswell equation, which correlates the theoretical methane potential to the chemical composition of the feedstock. Gu et al. (2021) proposed a multivariate analysis to predict MSW generation and computed the LHV from the composition of waste. Overall, previous works have relied on modelling approaches that, on the one hand, predict MSW generation from socio-economic trends and, on the other hand, estimate the energy recovery potential from waste generation and pre-defined energy content. These studies, therefore, do not involve a specific MFA model capable to track the flow of materials and energy while predicting the consequences of changes in the system.

In this chapter, the energy recovery potential of MSW in Madrid is quantified considering a reference scenario for the year 2019 and a set of plausible scenarios for the years 2030 and 2040. The aim is to evaluate to what extent an increase in separate collection and recycling could compromise energy recovery. To carry out this study, scenario analysis was combined with a parametrized MFA framework that facilitates a systematic analysis of materials and energy flows through the MSW management system. The MFA framework was applied to quantify the following indicators that allow getting a holistic picture of the energy recovery potential under each scenario:

- **Feedstock availability**: Available annual mass of potential WTE feedstocks (tonnes/year). Understanding this variable is paramount to determining treatment capacities.

- **Gross energy recovery potential**: Annual amount of energy contained in the WTE feedstocks (TJ/year) or maximum amount of energy that could be obtained.
- **Energy content**: Energy content per unit mass of WTE feedstock. The energy content of the feedstocks susceptible to thermochemical WTE conversion (e.g., incineration) is quantified through the LHV (MJ/kg feedstock), while the energy content of the organic feedstocks susceptible to undergo biochemical WTE conversion (i.e., AD) is quantified through the biochemical methane potential (BMP) per tonne of feedstock (m^3 CH_4/tonne feedstock). The BMP measures the maximum methane potential that could be achieved under optimum AD conditions (Naroznova et al., 2016).

Furthermore, prospects for WTE technologies in Madrid are discussed in relation to the availability and energy recovery potential of feedstocks, and the electricity generation potential is quantified based on technology performance.

3.2 Identifying waste-to-energy feedstocks in Madrid

As it has been described in Section 1.5, the municipality of Madrid is responsible for the collection and management of the waste generated by households, commercial activities (partially), and street cleaning. However, the system studied herein covers exclusively MSW management pathways to produce WTE feedstocks. The potential WTE feedstocks depend on the configuration of the MSW management system. The existing system configuration in 2019 is simulated in the Reference scenario, in which the following waste streams were identified as potential WTE feedstocks: organic waste collected separately, residual organic waste separated from the residual waste at MRFs, rejects from sorting residual waste at MRFs, rejects from sorting packaging waste at MRFs, rejects from AD pre-treatment, commercial residual waste, and street cleaning waste (Figure 3-1).

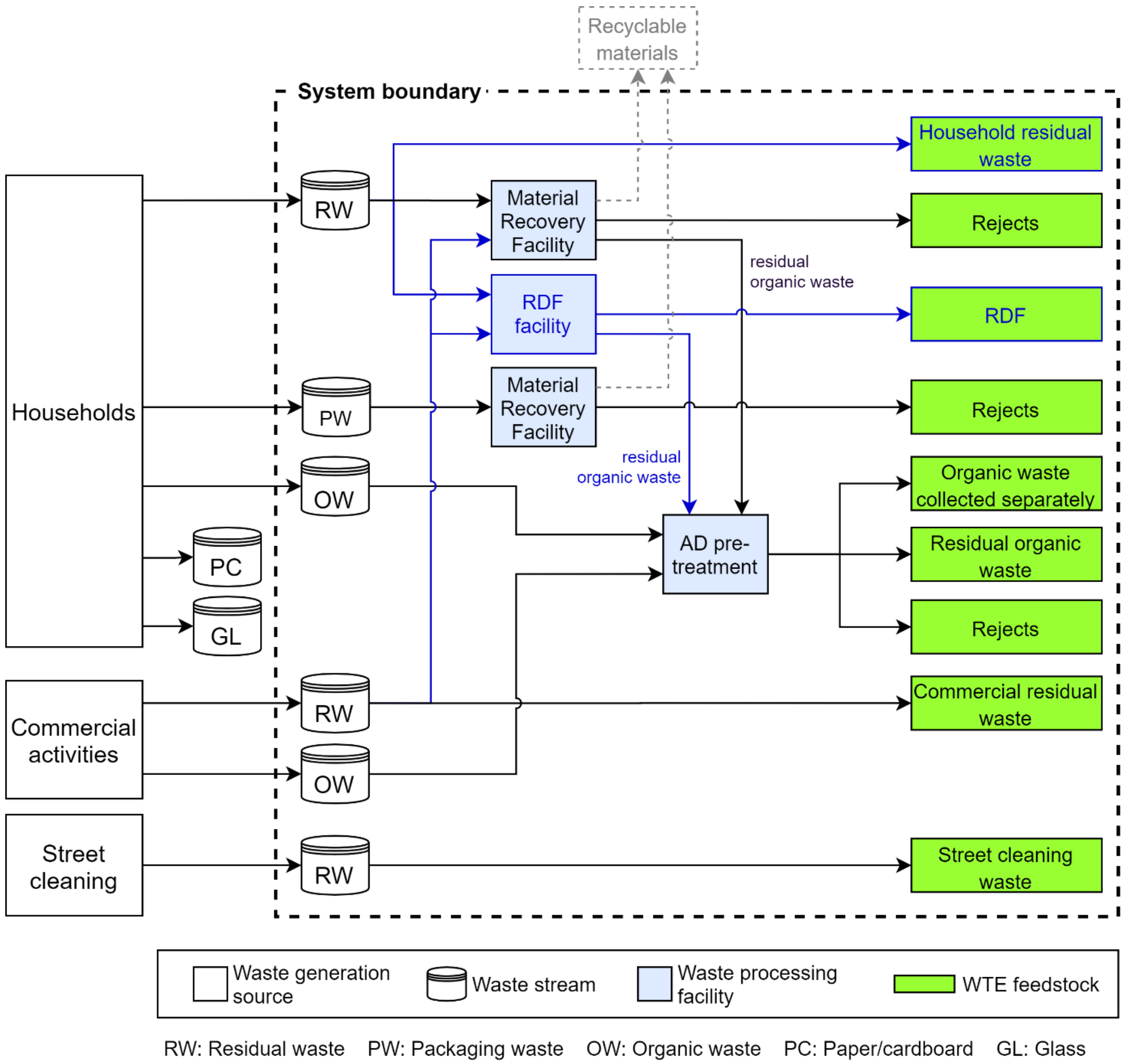

Figure 3-1. Municipal solid waste management pathways to produce waste-to-energy feedstocks in Madrid. Black lines and boxes represent existing pathways in Madrid in 2019. Blue lines and boxes represent alternative pathways investigated for the years 2030 and 2040. Thin dashed lines represent recyclable materials that are not addressed in this study. RDF: refuse-derived fuel.

For the years 2030 and 2040, a range of scenarios describing alternative system's configurations have been evaluated:

- <u>Business-as-Usual</u> (BAU): This scenario considers the current system's configuration for the years 2030 and 2040. Consequently, the BAU scenario involves the same potential WTE feedstocks as the reference scenario. Since the system remains unchanged, any variation in the energy recovery potential is exclusively associated with changes in waste generation and separate collection.
- <u>Improved MRFs</u> (I-MRF): This scenario simulates the substitution of the existing MRFs with new more efficient MRFs. This scenario involves the same potential WTE feedstocks as the

reference scenario, but it allows evaluating the joint effect of improving separate collection and material recovery efficiency.

- Refuse Derived Fuel (RDF): This scenario simulates the implementation of RDF production from the household and commercial residual waste. RDF is produced by removing non-combustible materials to increase the energy content and enhance the combustion properties. The RDF scenario involves the following potential WTE feedstocks: organic waste collected separately, residual organic waste separated from the residual waste at RDF facility, RDF, rejects from sorting packaging waste at MRFs, rejects from AD pre-treatment, and street cleaning waste.
- No MRFs (No-MRF): This scenario simulates a system without MRFs for residual waste. In consequence, the following potential WTE feedstocks were identified: organic waste collected separately, household residual waste, rejects from sorting packaging waste at MRFs, rejects from AD pre-treatment, commercial residual waste, and street cleaning waste.

3.3 Material flow analysis framework

An MFA framework was developed to facilitate a systematic analysis of materials and energy flows through the system illustrated in Figure 3-1. The fundamental modelling principle is that the mass flows and properties (e.g., energy content) of MSW streams are determined by the individual waste materials contained. MSW is typically disaggregated into the major material categories, namely organic waste, paper/cardboard, plastic, glass, metal, and others (Kaza et al., 2018). However, such a generic disaggregation does not account for the different properties and characteristics of materials included within the same category, e.g., food and green waste (organic waste category), different polymers (plastic category), steel and aluminium (metal category), or wood and textile (others category). Herein, MSW generation was further disaggregated into the most common 18 waste materials: food waste, green waste, cellulose (e.g., tissue paper and diapers), paper, cardboard, polyethylene terephthalate (PET), high-density polyethylene (HDPE), low-density polyethylene (LDPE), polypropylene (PP), other plastic packaging (e.g., polystyrene and polyvinyl chloride), non-packaging plastic (e.g., toys), cartons and alike (e.g., beverage cartons), glass, ferrous metal, non-ferrous metal (aluminium), textile, wood, and others. Each material was defined by a set of biological and physicochemical properties, such as moisture content, LHV, burnability factor, volatile solids (VS) content, and BMP (Table 3-1). The MFA model tracks the flow of each material through the MSW management system, from generation to separate collection and treatment. The mass and properties of any waste stream at any point across the system can be evaluated by combining data on the mass of waste materials contained with the material-specific properties (Table 3-1). Note that any change in the system –e.g., the improvement of the separate collection of food waste– would alter the flow of materials through the system and, in consequence, would have an impact on the mass and properties of waste streams. That is to say, the MFA framework allows predicting waste streams flow and composition based on structural changes in the MSW management system.

Table 3-1. Biological and physicochemical properties of waste materials. The data sources can be found in Appendix A.

Material	**Moisture** (% ww)	**LHV** (MJ/kg dw)	**Burnability** (% dw)	**VS** (% dw)	**BMP** (Nm^3 CH_4/t VS)
Food waste	69	15	100	83	390
Green waste	58	14	100	77	145
Cellulose	60	18	100	90	209
Paper	11	14	100	84	267
Cardboard	11	15	100	86	284
PET	6	27	100	93	0
HDPE	6	42	100	98	0
LDPE	22	34	100	94	0
PP	6	41	100	96	0
Other plastic packaging	8	28	100	94	0
Non-packaging plastic	5	31	100	95	0
Cartons and alike	15	20	95	94	211
Glass	2	0	0	1	0
Ferrous metal	4	0	0	1	0
Non-ferrous metal	3	0	0	3	0
Textile	14	19	100	87	225
Wood	12	17	100	88	82
Others	15	8	50	35	0

Note: LHV: lower heating value. VS: volatile solids. BMP: biochemical methane potential. ww: wet weight. dw: dry weight.

3.3.1 Mathematical formulation

The following paragraphs describe the mathematical formulation of the MFA framework.

Waste generation

The first step consists of projecting the generation of waste. The mass of each waste material k generated in year y ($W_{k,y}$) is computed from the mass generated in the reference year 2019 ($W_{k,2019}$) and the annual variation (δ_k) as follows:

$$W_{k,y} = W_{k,2019} \cdot (1 + \delta_k)^{y-2019}, \qquad \forall\, k \in K, y \in \{2030,2040\} \qquad \textit{Eq. 3-1}$$

Waste separate collection

In an integrated MSW management system, waste materials are separated into different bins (hereinafter also referred to as waste streams). In Madrid, for example, the household waste is collected separately into packaging, paper/cardboard, glass, and organic waste (set J^{SC} of waste streams collected separately), the rest being collected as part of the residual waste stream (set J^{RES} of residual waste streams). The separate collection rate of each waste stream $j \in J^{SC}$ collected in year y ($sc_{j,y}$) is defined as the yearly mass of j selectively collected relative to the yearly mass of target waste materials generated throughout that year (Dri et al., 2018; Zeschmar-Lahl et al., 2016). The target waste materials are those for which the bin is intended. Based on the MSW collection scheme implemented in Madrid, the waste materials targeted by each bin are established as follows:

- Packaging waste stream: PET, HDPE, LDPE, PP, other plastic packaging, cartons and alike, ferrous metal, and non-ferrous metal.
- Paper/cardboard stream: paper, cardboard.
- Glass stream: glass
- Organic waste stream: food waste, green waste.

Therefore, the organic waste stream, for example, targets food and green waste. Any non-target material placed in the organic waste stream, such as PET or textile, represents an impurity. The presence of impurities reduces the content on target materials, increases rejects, and can alter the properties of the waste stream (Andreasi Bassi et al., 2020). Note that, depending on the collection scheme, some materials are not targeted by any dedicated bin. In the collection scheme of household waste established in Madrid, these materials are cellulose, non-packaging plastic, textile, wood, and others. These materials must inevitably be placed into the residual waste stream.

Based on the above definitions, the mass of waste stream $j \in J^{SC}$ collected separately in year y ($\omega_{j,y}$) is computed as follows:

$$\omega_{j,y} = sc_{j,y} \cdot \sum_{k \in K} W_{k,y} \cdot \beta_{k,j}\,, \qquad \forall\, j \in J^{SC}, y \in \{2019, 2030, 2040\} \qquad \textit{Eq. 3-2}$$

where $\beta_{k,j}$ is a binary parameter with value 1 if waste material k is targeted by stream j and 0 otherwise.

Because waste separation is by nature imperfect, the waste streams collected separately are not only composed of target materials but also include a variety of non-target materials (impurities). The mass of each waste material k contained in waste stream j selectively collected in year y ($M_{k,j,y}$) is determined from the mass of stream collected ($\omega_{j,y}$) and its material composition ($\sigma_{k,j,y}$):

$$M_{k,j,y} = \omega_{j,y} \cdot \sigma_{k,j,y}, \qquad \forall\, j \in J^{SC}, k \in K, y \in \{2019,2030,2040\} \qquad \textit{Eq. 3-3}$$

The remaining waste materials that are not collected separately are placed into the residual stream. Accordingly, the mass of each waste material k contained in the residual waste stream collected in year y is determined by subtracting the mass collected separately from the mass generated ($\omega_{k,y}$):

$$M_{k,j,y} = \omega_{k,y} - \sum_{j \in J^{SC}} M_{k,j,y}\,, \qquad \forall\, j \in J^{RES}, y \in \{2019,2030,2040\} \qquad \textit{Eq. 3-4}$$

The model described by Eq. 3-2 to Eq. 3-4 produces a matrix with the mass of each waste material (rows) contained in each waste stream (columns) in each year (dimensions). The calculation is repeated for each waste generation source (i.e., households, commercial activities, and street cleaning) and each year.

Mechanical sorting

The only type of waste processing included in the studied system is the mechanical sorting carried out at MRFs, RDF facilities, and AD pre-treatment (Figure 3-1). Mechanical sorting involves a sequence of operations including manual separation, ballistic separation, air classification, optic sorting, magnets, and eddy currents (Cimpan et al., 2015a). The mass balance of mechanical sorting was modelled with material-specific transfer coefficients that describe the partitioning of each input material among the output streams (Tanguay-Rioux et al., 2021). Depending on the objective of the facility, the output streams may consist of several single recyclable materials (i.e., paper, PET, HDPE, ferrous metal, etc.), RDF, residual organic waste, and rejects. Herein, only the potential WTE feedstocks are assessed, i.e., RDF, residual organic waste, and rejects. Further details on transfer coefficients are provided in the data section 3.3.2.

Energy recovery potential indicators

The mass availability of each WTE feedstock is computed from the flow of materials as described above. The gross energy recovery potential, on the other hand, is calculated by combining data on the mass of waste materials contained in each WTE feedstock with the specific properties of waste materials (Table 3-1).

The gross energy recovery potential of the feedstocks susceptible to thermochemical WTE conversion $f \in F^{Thermochemical}$ (household residual waste, rejects from residual MRF, rejects from packaging MRF, rejects from AD pre-treatment, commercial residual waste, street cleaning waste) is quantified through the LHV as follows:

$$E_f = \sum_{k \in K} F_{k,f} \cdot (1 - moist_k) \cdot burn_k \cdot lhv_k\,, \qquad \forall\, f \in F^{Thermochemical} \qquad \textit{Eq. 3-5}$$

where E_f is the gross energy recovery potential of feedstock f [MJ/year], $F_{k,f}$ is the mass of waste material k contained in feedstock f [tonnes/year], $moist_k$ is the moisture content of waste material k [% mass], $burn_k$ is the burnability factor of waste material k [% dry mass], and lhv_k is the LHV of waste material k [MJ/tonne dry material]. The parameter $burn_k$ is applied to distinguish between inert (e.g., glass and metal) and burnable (e.g., plastic and wood) materials as recommended elsewhere (Hellweg et al., 2001). Only the energy content of burnable materials contributes to the gross energy recovery potential.

The gross energy recovery potential of the organic feedstocks susceptible to undergo AD $f \in F^{Biochemical}$ (organic waste collected separately and residual organic waste) is quantified through the BMP as follows:

$$E_f = \sum_{k \in K} F_{k,f} \cdot (1 - moist_k) \cdot vs_k \cdot bmp_k \cdot lhv\_CH_4, \qquad \forall\, f \in F^{Biochemical} \qquad \textit{Eq. 3-6}$$

where vs_k is the volatile solids (VS) content of waste material k [% dry mass], bmp_k is the BMP of waste material k [Nm^3 CH_4/tonne VS], and LHV_CH_4 is the LHV of CH_4 [35.8 MJ/m^3 CH_4].

Finally, the energy content of each WTE feedstock can be easily computed by dividing its gross energy recovery potential by the annual mass.

3.3.2 Case study data

This section summarizes the major assumptions and data used for the case study of Madrid.

Waste generation

Table 3-2 shows the amount of waste materials generated by households, commercial activities, and street cleaning in Madrid in the reference year 2019 and by 2030 and 2040.

MSW generation and composition in 2019 were obtained from the annual report elaborated by the Madrid City Council (2019). This report provides information for nine major material categories: food waste, green waste, cellulose, paper/cardboard, packaging, non-packaging plastic, glass, textile, and others. Several assumptions were made to further disaggregate these categories into the 18 materials considered in the MFA framework. The paper/cardboard category was disaggregated into 73% paper and 27% cardboard, based on forestry production and trade statistics provided by the Food and Agriculture Organization of the United Nations (FAO, 2022). The packaging category was disaggregated into 76% plastic packaging, 6% cartons and alike, 13% ferrous metal, and 5% non-ferrous metal, based on a characterization campaign carried out in Madrid in 2016 (Madrid City Council, 2017a). The polymer composition of the plastic packaging varies across waste streams. The plastic content of the residual waste streams consists of 19% PET, 6% HDPE, 55% LDPE, 15% PP, and 5% others, while in the packaging waste stream it consists of 29% PET, 11% HDPE, 40% LDPE, 15% PP, and 6% other (Madrid City Council, 2017a). The material category others account for up to 15% of the MSW generated in Madrid in 2019. This category includes waste materials that have not been identified during the sampling process and, consequently, further disaggregation was not possible.

Table 3-2. Municipal solid waste generation in Madrid in 2019, 2030, and 2040 (tonnes/year).

Material	**Households**			**Commercial activities**			**Street cleaning**		
	2019	**2030**	**2040**	**2019**	**2030**	**2040**	**2019**	**2030**	**2040**
Food waste	190,871	197,265	203,264	20,681	21,373	22,023	2,674	2,764	2,848
Green waste	70,063	72,410	74,612	2,621	2,709	2,791	45,931	47,470	48,914
Cellulose	60,187	62,204	64,095	3,535	3,653	3,764	881	911	939
Paper	151,211	156,276	161,029	25,901	26,769	27,583	17,779	18,375	18,934
Cardboard	57,238	59,155	60,954	9,804	10,133	10,441	5,421	5,602	5,773
PET	20,434	21,119	21,761	3,824	3,952	4,072	1,685	1,742	1,795
HDPE	6,238	6,447	6,643	1,141	1,180	1,216	503	520	536
LDPE	43,233	44,681	46,040	10,977	11,345	11,690	4,838	5,001	5,153
PP	12,534	12,954	13,348	2,923	3,021	3,113	1,288	1,331	1,372
Other plastic packaging	4,663	4,819	4,966	949	981	1,010	418	432	445
Non-packaging plastic	62,976	65,086	67,065	13,270	13,714	14,131	5,849	6,045	6,229
Cartons and alike	12,735	13,162	13,562	1,487	1,536	1,583	625	646	666
Glass	97,672	100,944	104,013	2,327	2,405	2,478	1,841	1,903	1,961
Ferrous metal	15,680	16,205	16,698	4,096	4,233	4,362	3,954	4,086	4,211
Non-ferrous metal	5,502	5,687	5,859	2,013	2,081	2,144	928	959	989
Textile	140,489	98,246	59,844	1,018	1,052	1,084	2,308	2,385	2,458
Wood	19,401	20,051	20,661	6,028	6,230	6,419	20,687	21,380	22,030
Others	170,059	175,756	181,101	3,546	3,665	3,776	19,165	19,807	20,409

MSW generation by 2030 and 2040 has been calculated with Eq. 3-1 and applying a 0.3% annual increase. The 0.3% annual increase corresponds to the default assumption for Spain adopted in the European Reference Model on Waste Generation and Management under the assumption that the economic recovery will not increase significantly in the near future (Hogg et al., 2015). The potential effects of the COVID-19 pandemic on waste generation have not been considered since long-term trends are still largely unknown (Naughton, 2020).

While the 0.3% annual increase has been uniformly applied to all the waste materials, particular attention has been paid to the future generation of textile waste due to its high energy content and the ongoing policy to increase its re-use and recycling. Textile is considered a key product value chain in the new Circular Economy Action Plan (European Commission, 2020). Furthermore, the Waste Framework Directive establishes the mandatory separate collection of textile waste by 2025. Within this context, a decreasing trend in the generation of textile waste is expected. In this work, it was assumed that the amount of textile waste generated by households drops 60% by 2040 due to re-use and recycling. This figure corresponds to the average of the two best-performing countries in textile re-use and recycling, namely Germany (70%) and Denmark (50%) (Sandin and Peters, 2018).

Separate collection rates

In 2019, about 80% of packaging waste, 35% of paper/cardboard, 60% of glass, and 42% of organic waste generated by households in Madrid were collected separately (Table 3-3). Furthermore, about 75% of the commercial organic waste was collected separately. The projection of the separate collection rates up to 2030 and 2040 follows the benchmark of excellence identified by the European Commission and the literature as summarized below:

- Packaging waste: The benchmark of excellence for the separate collection of co-mingled packaging waste is ≥ 65% (Dri et al., 2018; Zeschmar-Lahl et al., 2016). (Tallentire and Steubing, 2020) estimated that the average rate of separate collection to realize the 2030 EU recycling target for packaging is 72%. Madrid has already achieved about 80% separate collection of packaging waste in 2019. In consequence, it was assumed that this rate slightly increases up to 85% by 2030 and that it remains unchanged afterward.
- Paper/cardboard: The benchmark of excellence for the separate collection of paper/cardboard is 85%. (Tallentire and Steubing, 2020) calculated a rate of 86% to realize the 2030 EU recycling target. Since Madrid is still far from these figures, it was assumed that the rate of separate collection of paper/cardboard increases up to 70% by 2030 and up to 85% by 2040.
- Glass: The benchmark of excellence identified for the separate collection of glass is 90%. (Tallentire and Steubing, 2020) calculated a separate collection rate of 94% to realize the 2030 EU recycling target. Herein it was assumed that the separate collection of glass in Madrid increases from 60% in 2019 to 80% by 2030 and 90% by 2040.
- Organic waste: The separate collection of household organic waste was introduced in Madrid in 2017. By 2019, about 42% of the household organic waste was collected separately. The current average separate collection rate of organic waste in the European Union (EU) is 34% (Favoino and Giavini, 2020). The highest national rate is achieved by Cyprus (83%) followed by Finland (57%), Italy (55%), Hungary (55%), and Estonia (54%). Some regions in Italy have reached separate collection rates higher than 65% (Di Maria et al., 2020b). In Ghent (Belgium), the rate of separate

collection of organic waste was 60% in 2016 (Sanjuan-Delmás et al., 2021). Favoino and Giavini (2020) established that the maximum operational separate collection rate of organic waste should be around 85%. Herein, it was assumed that the rate of separate collection of household organic waste increase from 42% in 2019 up to 70% by 2030 and up to 85% by 2040. Furthermore, the separate collection of commercial organic waste increases from 74.9% in 2019 up to 85% by 2030 and remains constant afterwards.

Table 3-3. Rates of separate collection of municipal solid waste in Madrid in 2019, 2030, and 2040.

Stream	Households			Commercial activities			Street cleaning		
	2019	2030	2040	2019	2030	2040	2019	2030	2040
Packaging waste	79.5%	85.0%	85.0%	–	–	–	–	–	–
Paper/cardboard	35.2%	70.0%	85.0%	–	–	–	–	–	–
Glass	60.2%	80.0%	90.0%	–	–	–	–	–	–
Organic waste	42.3%	70.0%	85.0%	74.9%	85.0%	85.0%	–	–	–

Composition of waste streams collected separately

Table 3-4 shows the composition of the packaging waste, paper/cardboard, glass, and organic waste collected separately from households. These figures correspond to the average composition for the period 2015-2019 and were calculated using data from the annual reports elaborated by the Madrid City Council (2015, 2016, 2017b, 2018, 2019).

Table 3-4. Composition of the household waste streams collected separately in Madrid (% of mass).

Material	Packaging	Paper/cardboard	Glass	Organic waste
Food waste	4.62	0.20	0.00	61.73
Green waste	0.73	0.00	0.00	8.77
Cellulose	4.06	0.10	0.00	4.33
Paper	7.08	73.18	0.06	3.24
Cardboard	2.17	22.76	0.01	1.09
PET	8.99	0.09	0.07	1.11
HDPE	3.27	0.00	0.00	0.03
LDPE	12.21	0.36	0.07	0.54
PP	4.53	0.08	0.00	0.05
Other plastic packaging	1.69	0.01	0.00	0.36
Non-packaging plastic	11.06	0.79	0.07	0.43
Cartons and alike	6.35	0.10	0.00	0.93
Glass	2.98	0.20	99.09	2.13
Ferrous metal	5.37	0.10	0.07	0.98
Non-ferrous metal	1.92	0.00	0.00	0.34
Textile	7.65	0.69	0.07	2.67
Wood	0.88	0.50	0.00	1.97
Others	14.46	0.85	0.49	9.29
Targeted materials	44%	96%	99%	71%
Non-targeted materials	56%	4%	1%	30%

As it can be observed, the level of impurities (non-targeted materials) is high in the packaging waste (56% of mass) and organic waste (30% of mass) streams, but negligible in the paper/cardboard (4% of mass) and glass (1% of mass) streams. The organic waste collected separately from commercial activities was assumed to consist of 100% food waste (i.e., 0% impurities). The same composition was assumed for all years. Note that the composition of the residual waste is adjusted by the MFA framework to account for the increased separate collection.

Transfer coefficients

Transfer coefficients were defined for each process involved in the studied system: existing and new MRFs for sorting residual waste, existing and new MRFs for sorting packaging waste, RDF facility, and AD pre-treatment (Table 3-5).

MRFs treating residual waste recovers paper, cardboard, plastic, cartons and alike, glass, and metal. Average recovery efficiencies of existing MRFs in Madrid were calculated from on-site data for the period 2015-2019. The new MRFs have relatively higher recovery efficiencies, based on average data from Cimpan et al. (2015a) and Montejo et al. (2013). RDF facilities separate 86% of ferrous metal and 81% of non-ferrous metal for recycling (Sardarmehni and Levis, 2021). Average transfer coefficients of waste materials into the RDF were obtained from Stegmann (2011), Montejo et al. (2011), Nasrullah et al. (2015), and Sardarmehni and Levis (2021). Both MRFs and RDF facilities are equipped with a trommel for separating the residual organic waste from the residual waste stream. Transfer coefficients for trommel screens are based on average data from Stegmann (2011), Pressley et al. (2015), and Cobo et al. (2019).

MRFs treating packaging waste recover paper, cardboard, plastic, cartons and alike, glass, and metal. Recovery efficiencies of existing MRFs in Madrid were also calculated from on-site data for the period 2015-2019, while average efficiencies for new MRF were obtained from Cimpan et al. (2016), Ip et al. (2018), and Kleinhans et al. (2020). AD pre-treatment uses a trommel to separate the input organic waste feedstock into two streams: a purified organic waste stream that is fed to AD digesters and a stream of rejects. Transfer coefficients into the purified organic waste stream are the same as for the trommel section of MRFs and RDF facilities. Finally, transfer coefficients into the rejects were calculated for all processes by subtracting the transfer coefficients into the remaining output streams.

Table 3-5. Transfer coefficients of existing and new material recovery facilities (MRFs), refuse-derived fuel (RDF) facility, and anaerobic digestion (AD) pre-treatment (% input mass).

Material	**MRFs for residual waste [existing / new]**			**MRF for packaging waste [existing / new]**		**RDF facility**			**AD pre-treatment**	
	Recyclable materials	Residual organic waste	Rejects	Recyclable materials	Rejects	Recyclable materials	RDF	Residual organic waste	Organic waste	Rejects
Food waste	– / –	90 / 90	10 / 10	– / –	100 / 100	–	10	90	90	10
Green waste	– / –	63 / 63	37 / 37	– / –	100 / 100	–	4	96	63	37
Cellulose	– / –	60 / 60	40 / 40	– / –	100 / 100	–	60	40	60	40
Paper	14 / 29	35 / 35	51 / 36	45 / 60	55 / 40	–	82	18	35	65
Cardboard	14 / 29	35 / 35	51 / 36	45 / 60	55 / 40	–	82	18	35	65
PET	8 / 51	30 / 30	62 / 19	57 / 85	43 / 15	–	85	15	30	70
HDPE	7 / 51	30 / 30	61 / 20	57 / 85	43 / 15	–	85	15	30	70
LDPE	6 / 40	30 / 30	64 / 30	39 / 59	61 / 41	–	85	15	30	70
PP	– / –	30 / 30	70 / 70	35 / 64	65 / 36	–	85	15	30	70
Other plastic packaging	2 / 10	30 / 30	68 / 60	35 / 73	65 / 27	–	85	15	30	70
Non-packaging plastic	2 / 10	30 / 30	68 / 60	35 / 73	65 / 27	–	78	22	30	70
Cartons and alike	6 / 77	17 / 17	77 / 7	63 / 80	37 / 20	–	77	23	17	83
Glass	1 / 20	10 / 10	89 / 70	23 / –	77 / 100	–	7	93	10	90
Ferrous metal	72 / 72	10 / 10	18 / 19	92 / 92	8 / 8	86	9	5	10	90
Non-ferrous metal	22 / 58	10 / 10	68 / 32	44 / 76	56 / 24	81	9	10	10	90
Textile	– / –	5 / 5	95 / 95	– / –	100 / 100	–	95	5	5	95
Wood	– / –	5 / 5	95 / 95	– / –	100 / 100	–	86	14	5	95
Others	– / –	50 / 50	50 / 50	– / –	100 / 100	–	5	95	50	50

3.4 Energy recovery potential assessment for Madrid

This section presents the application of the MFA framework to quantify the energy recovery potential of MSW in Madrid under the scenarios defined above. First, Section 3.4.1 projects the mass flows of MSW streams collected in each scenario. Subsequently, Section 3.4.2 presents the yearly mass flows of WTE feedstocks, while Section 3.4.3 and Section 3.4.4 focus on their yearly gross energy recovery potential and energy content, respectively. Finally, prospects for WTE technologies in Madrid are discussed in Section 3.4.5.

3.4.1 Projecting municipal solid waste collection

Table 3-6 summarizes the mass flows of MSW streams collected in Madrid in 2019, 2030, and 2040 based on the assumed waste generation, composition, and separate collection rates.

Table 3-6. Mass flows of municipal solid waste streams collected in Madrid in 2019, 2030 and 2040 (tonnes/year) and the corresponding rate of separate collection (% of MSW).

Waste stream	2019	2030	2040
Household waste			
Residual waste	802,346	605,825	497,477
Packaging	96,231	106,313	109,546
Paper/cardboard	73,475	150,802	188,685
Glass	58,809	80,755	93,612
Organic waste	110,327	188,773	236,195
Commercial waste			
Residual waste	100,653	101,863	104,961
Organic waste	15,487	18,167	18,720
Street cleaning waste			
Mixed waste	136,777	141,359	145,658
Separate collection rate			
Only households	30%	47%	56%
All generators	25%	39%	46%

In 2019, only 30% of the household waste was collected separately (338,841 tonnes), with the remaining 70% being collected as part of the residual waste stream (802,346 tonnes). If also considering commercial activities and street cleaning, about 75% of the total MSW treated in 2019 consisted of residual/mixed waste (1.04 million tonnes) and only 25% was collected separately (354,328 tonnes). The rate of separate collection increases up to 39% and 46% by 2030 and 2040, respectively. Compared with 2019, the yearly amount of packaging waste, paper/cardboard, glass, and organic waste collected separately from households in 2040 increase by 0.14, 1.57, 0.59, and 1.14.

However, despite these improvements, the results for the years 2030 and 2040 suggest that residual/mixed waste is likely to continue to be a large share of the MSW. The amount of residual/mixed waste reaches 849,047 tonnes/year by 2030 (61% of MSW) and 748,095 tonnes/year by 2040 (54% of MSW), compared with 1.04 million tonnes in 2019 (75% of MSW).

The large flow of commercial residual waste and street cleaning waste found by 2030 and 2040 could be expected since separate collection was considered negligible and/or null. Conversely, the large flow of household residual waste may appear rather surprising since as high as 85-90% separate collection has been established for packaging, paper/cardboard, glass, and organic waste, while the diversion of textile towards re-use and recycling was assumed to increase up to 60%. There are two fundamental reasons for this trend. On the one hand, the MFA framework proposed in this work accounts for the presence of non-target materials (impurities) in the waste streams. This means that, for example, an 85% separate collection rate does not necessarily imply that 85% of the packaging waste generated is collected separately. In fact, as high as 56% of the packaging waste collected separately consists of non-packaging materials, based on data from Madrid (*cf.* Table 3-4). On the other hand, most important is that not all the waste materials are targeted by the separate collection scheme. For the case study of Madrid, it was established that cellulose, non-packaging plastic, textile, wood, and others are not subject to separate collection at households. These materials sum 421,343 and 392,765 tonnes/year in 2030 and 2040, respectively. This means that 35-37% of the MSW consists of materials that are not targeted by the separate collection scheme.

These findings reveal the limitations of separate collection as an indicator to measure the efficiency of diverting MSW towards recycling. At the same time, these limitations imply the production of large flows of potential WTE feedstocks, as demonstrated in the following sections.

3.4.2 Availability of waste-to-energy feedstocks

Figure 3-2a presents the yearly mass flows of WTE feedstocks in Madrid under each scenario. The MSW streams identified as potential WTE feedstocks represent approximately 86% of the MSW generated in the reference scenario 2019 (about 1.20 million tonnes). The major WTE feedstock is the flow of rejects from MRFs (about 502,737 tonnes/year; 36% of MSW). Other potential feedstocks susceptible to thermochemical WTE conversion include rejects from AD pre-treatment (168,905 tonnes/year), street cleaning waste (136,777 tonnes/year), and commercial residual waste (100,653 tonnes/year). The potential feedstocks susceptible to energy recovery through AD include 92,576 tonnes/year of organic waste collected separately and 199,324 tonnes/year of residual organic waste (after subtracting the amount of pre-treatment rejects). All in all, in Madrid in 2019, about 909,072 tonnes/year of waste (65% of MSW) could be susceptible to thermochemical WTE conversion and about 291,899 tonnes/year of organic waste (21% of MSW) could be susceptible to AD. However, in 2019 only about 333,000 tonnes/year of rejects from MRFs were incinerated and about 172,000 tonnes/year of organic waste were treated through AD (Madrid City Council, 2017b). This sums up to 505,000 tonnes/year of MSW dedicated to energy recovery which means that only 42% of the mass potential calculated herein was exploited in 2019.

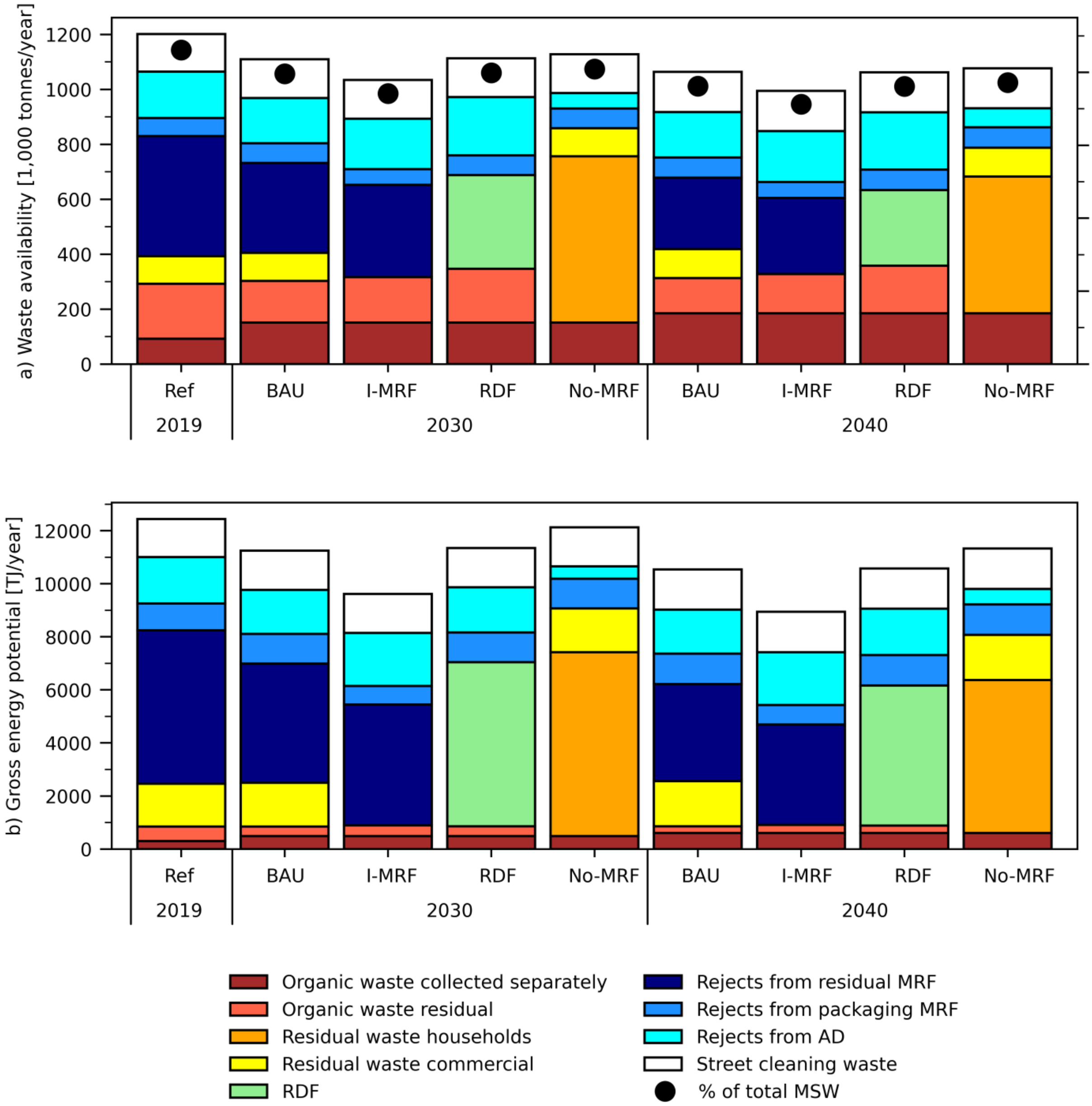

Figure 3-2. a) Yearly mass flows of potential waste-to-energy feedstocks available in Madrid under each scenario. The black dots (right y-axis) represent the percentage of waste-to-energy feedstocks with respect to the total mass of municipal solid waste. b) Yearly gross energy recovery potential under each scenario.

The scenarios for the years 2030 and 2040 depict a small decrease in the availability of WTE feedstocks. The percentage of potential WTE feedstocks with respect to the total mass of MSW drops from 86% in 2019 to 74-81% by 2030 and 71-77% by 2040, depending on the scenario. The BAU scenarios suggest that the improvement of separate collection reduces the availability of WTE feedstocks by only 8% by 2030 and 12% by 2040 over the reference scenario 2019. The improvement of separate collection together with the implementation of more efficient MRFs (scenarios I-MRF) reduces the availability by 14% by 2030 and 17% by 2040. The production of RDF (scenarios RDF) and the non-utilization of MRFs (scenarios No-MRF) have less impact on the availability of WTE feedstocks, with reductions ranging from 6% to 7% by 2030 and from 10% to 12% by 2040.

The future yearly amount of WTE feedstocks susceptible to thermochemical WTE conversion totals between 718,026 and 976,907 tonnes/year by 2030 and between 666,751 and 891,815 tonnes/year by 2040, compared with 909,072 tonnes/year in 2019. The largest flow is found in the scenarios No-MRF, where the household and commercial residual waste streams are readily available to undergo thermochemical conversion. The lowest flow is observed in the scenarios I-MRF, where the household and commercial residual waste streams are directed to new more efficient MRFs. The RDF scenarios suggest that up to 341,113 and 275,984 tonnes/year of RDF could be produced by 2030 and 2040, respectively.

Regarding the WTE feedstocks susceptible to undergo AD, the amount of organic waste collected separately increases from 92,576 tonnes/year in 2019 up to 150,901 and 185,198 tonnes/year by 2030 and 2040, respectively (after subtracting pre-treatment rejects). Conversely, the amount of residual organic waste that can be obtained from sorting residual waste at MRFs drops. This decreasing trend is motivated by the fact that less organic waste ends up in the residual waste stream as the rate of separate collection increases. Under the BAU scenarios, the mass of residual organic waste decreases from 199,324 tonnes/year in 2019 to 152,029 and 128,110 tonnes/year by 2030 and 2040, respectively. The trend is less pronounced under the I-MRF and RDF scenarios since here the commercial residual waste is mechanically sorted as well. By 2040, the residual organic waste reaches 142,140 tonnes/year in the scenarios I-MRF and 172,541 tonnes/year in the RDF scenarios.

3.4.3 Gross energy recovery potential

The gross energy recovery potential of the WTE feedstocks available in Madrid in the reference scenario 2019 achieves 12,436 TJ/year (Figure 3-2b). About 55% of this potential is contained in the rejects from MRFs (6,794 TJ/year), 14% in the rejects from AD pre-treatment (1,755 TJ/year), 13% in the residual commercial waste (1,610 TJ/year), and 11% in the street cleaning waste (1,426 TJ/year). Overall, the gross energy recovery potential through thermochemical WTE conversion achieves 11,585 TJ/year (93% of total potential). The gross energy recovery potential through AD is much lower, accounting for 852 TJ/year distributed between organic waste collected separately (304 TJ/year) and residual organic waste (548 TJ/year). This reflects the lower mass of organic waste feedstocks and the higher moisture content of these streams. In Madrid in 2019, about 73% of rejects from MRFs were incinerated and 55% of organic waste was treated through AD. If these rates were applied to the potentials calculated herein, it would result that only 44% of the gross energy recovery potential (5,454 TJ/year) was exploited in 2019.

Focusing on the future scenarios, the gross energy recovery potential equals between 9,621 and 11,346 TJ/year by 2030 and between 8,942 and 11,327 TJ/year by 2040. The potential continues to be largely concentrated in the feedstocks susceptible to thermochemical WTE conversion (91-96%). Therefore, the improvement of separate collection reduces the gross energy recovery potential an 10% by 2030 and 15% by 2040 (scenarios BAU), chiefly due to the increasing diversion of combustible materials (i.e., paper, cardboard, plastic, and textile) towards recycling. The maximum reduction is observed in the scenarios I-MRF (23% and 28% by 2030 and 2040, respectively), where the implementation of new more efficient MRFs further reduces the proportion of combustible materials transferred into the WTE feedstocks. Conversely, the minimum reduction is observed in the scenarios No-MRF; 2% and 9% reduction by 2030 and 2040, respectively. In this scenario, the household and commercial residual waste streams are not sorted at MRFs and, in consequence, all the combustible materials (e.g., paper and plastic) remain in the residual waste. Finally, the RDF scenarios show a similar gross energy recovery potential as the BAU and lower than the No-MRF scenarios.

3.4.4 Energy content of waste-to-energy feedstocks

The suitability of a feedstock for energy recovery largely depends on its energy content per unit of mass. In this regard, Figure 3-3 presents the energy content of the potential WTE feedstocks available in Madrid under each scenario.

The BMPs of the organic waste collected separately and the residual organic waste in the reference scenario 2019 are 92 and 77 m^3 CH_4/tonne, respectively (after subtracting pre-treatment rejects). The residual organic waste has lower BMP due to its higher level of impurities; 40% compared with 12% in the organic waste collected separately. The BMP of the organic waste collected separately is the same in all the scenarios since the composition of this stream was assumed constant across years and this feedstock is not affected by changes in the system’s configuration. Conversely, the BMP of the residual organic waste decreases to 53-63 m^3 CH_4/tonne by 2030 and to 46-60 m^3 CH_4/tonne by 2040. This decreasing trend is primarily caused by the improvement of the separate collection of food and green waste. The share of food and green waste in the residual organic waste decreases from 60% in the reference scenario 2019 to 50-54% by 2030 and to 45-49% by 2040. At the same time, the share of non-biodegradable materials (i.e., plastic, glass, metal, and others) increases from 23% in 2019 to 30-40% by 2030 and 35-45% by 2040. In other words, up to 30-45% of the residual organic waste fed to the digesters would consist of non-biodegradable materials.

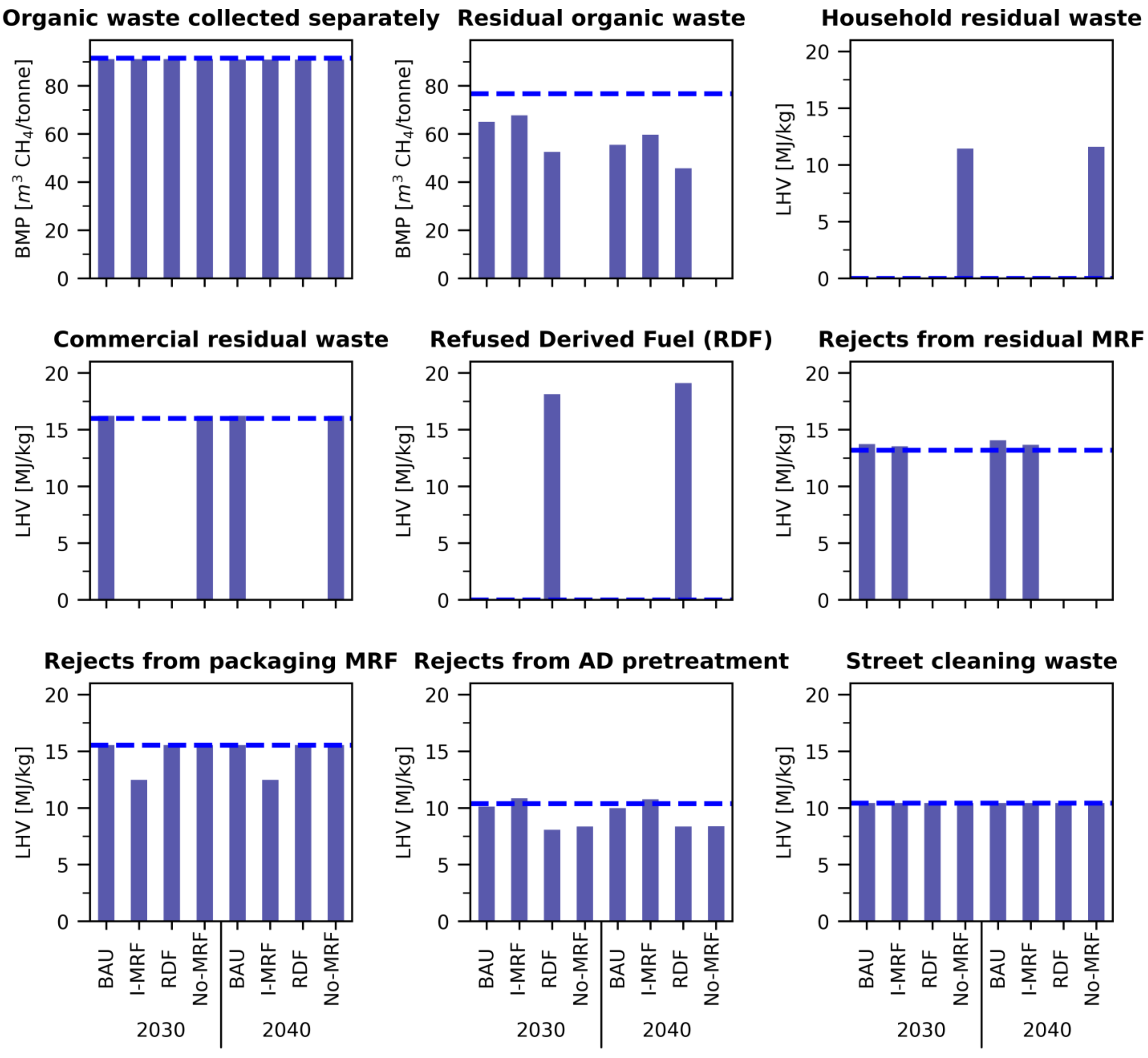

Figure 3-3. Energy content of potential waste-to-energy feedstocks available in Madrid under each future scenario. The horizontal blue dashed line shows the results for the 2019 reference scenario.

The average LHV of the feedstocks susceptible to thermochemical WTE conversion reaches 13 MJ/kg in the reference scenario 2019. Rejects from sorting packaging waste at MRFs and the commercial residual waste has the highest LHV with 16 MJ/kg each. Rejects from sorting household residual waste at MRFs have an LHV of 13 MJ/kg, while AD pre-treatment rejects and street cleaning waste have an LHV of 10 MJ/kg each. The results for the years 2030 and 2040 suggest a negligible impact of separate collection and recycling on the LHVs. The LHVs found in the future scenarios are generally similar to those observed in the reference scenario and fall within the range of 10 to 12 MJ/kg used at incineration facilities in Europe (Reimann, 2012).

3.4.5 Waste-to-energy technologies prospects

The gross energy recovery potential quantified in Section 3.4.3 gives an overview of the maximum amount of energy that could be obtained from the feedstocks. This potential is an inherent property of the feedstocks, while the actual amount of useful energy that could be obtained (e.g., electricity) will depend on the WTE technology used and its performance. Herein, future prospects for AD and incineration in Madrid are discussed and the electricity generation potential is quantified.

Biochemical waste-to-energy conversion

The energy content of the organic feedstocks can be exploited through AD for electricity generation from the combustion of the biogas. AD achieves between 70% and 75% of the feedstock's CH_4 potential (Banks et al., 2011; Møller et al., 2009; Smith et al., 2021; U.S. Environmental Protection Agency, 2008). The upper bound fits better to a relatively homogeneous feedstock (e.g., organic waste collected separately), while the lower bound could correspond to a more heterogeneous feedstock (e.g., residual organic waste). It is important to note that about 2.5% of the CH_4 produced by AD is typically lost due to leakage. Furthermore, AD facilities typically flare a non-negligible share of the biogas (up to 15% on a yearly basis) due to storage limitations and/or low methane content (Smith et al., 2021)[§]. All in all, only between 59% and 62% of the gross energy recovery potential is eventually available for electricity generation. Biogas engines have a gross electrical efficiency of about 37.5%, but parasitic loads (electricity used to satisfy internal demand) account for up to 10%. Therefore, biogas combustion engines reach a net electrical efficiency of about 27.2%. Furthermore, the oxidation efficiency of CH_4 in the combustion process is about 99% (Cobo et al., 2019; US EPA, 2011). After applying all these losses, it results that only 16-17% of the gross energy recovery potential contained in the organic feedstock is converted to electricity through AD.

Based on the technical considerations described above and the BMP of the organic waste collected separately (91 m^3 CH_4/tonne), its electricity production rate equals 152 kWh/tonne. This figure falls within the lower bound of the range 130-430 kWh/tonne reported in Di Maria et al. (2018). The electricity production rate of the residual organic waste drops to 71-119 kWh/tonne, as this feedstock has higher heterogeneity and lower BMP. Considering the yearly mass of organic feedstocks available

[§] The fugitive emissions and biogas flaring rate used in this analysis correspond to an AD facility where the biogas is burned for electricity generation. An AD facility equipped with biomethane upgrading would exhibit higher fugitive emissions and a lower flaring rate, as shown in Chapter 4.

in each scenario (Figure 3-2a), the electricity generation potential of AD ranges from 37,890 MWh in the reference scenario 2019 to 22,886-40,326 MWh by 2030 and 28,030-41,211 MWh by 2040. The great variation in the future scenarios is caused by the configuration of the MSW management system. In particular, the lowest electricity generation potential is achieved in the scenarios No-MRF, since the residual organic waste is not separated at MRFs.

Thermochemical waste-to-energy pathways

The existing incineration facility of Madrid treats on average 300,000 tonnes/year of waste. However, the MFA results revealed that the mass of feedstocks susceptible to thermochemical WTE conversion reaches 909,072 tonnes/year in the reference scenario 2019 and between 666,751 and 891,815 tonnes/year by 2040. This means a treatment capacity gap of 609,072 tonnes/year in 2019 and between 366,751 and 591,815 tonnes/year by 2040. These flows of waste per se would contribute to the landfill rate with 44% in 2019 and with 26-42% by 2040.

The incineration facility of Madrid dates from 1993 and its gross and net electrical efficiencies are relatively low; about 16.7% and 12.2% with respect to the waste LHV, respectively (see Table B-1 in Appendix B). This efficiency is within the observed range for existing facilities in Europe. For example, Doka (2013) reported that on average the Swiss MSW incineration facilities convert 16% of the waste LHV to electricity, while Beylot et al. (2017) found that the average electrical efficiency of French MSW incineration facilities is 15%. Meanwhile, modern large-scale incineration facilities reach net electrical efficiencies as high as 26% (Ardolino et al., 2020; Dong et al., 2018a; Lombardi et al., 2015; Turconi et al., 2011), which can further increase up to 30% in the case of RDF incineration (Lombardi et al., 2015). In this context, the electricity generation potential of incineration has been quantified considering three alternative incineration pathways.

In the first pathway, all the available feedstocks but RDF are hypothetically incinerated in the existing facility (the RDF is incinerated in a dedicated new facility). For the reference scenario 2019, this pathway results in the generation of 392,596 MWh. The incineration of all the feedstocks available by 2040 in the existing facility would generate between 272,305 and 363,440 MWh. If the production of RDF is implemented by 2040, the electricity generation potential would increase up to 589,304 MWh due to the higher efficiency of a new RDF incineration facility.

The first pathway is based on an unrealistic assumption since the capacity of the existing facility is not enough to treat all the available feedstocks. In this regard, the second pathway considers the simultaneous use of the existing facility plus a new facility. This pathway would increase the electricity generation potential in the reference scenario by 76% up to 690,130 MWh. Focusing on the future scenarios, the electricity generation potential would reach 550,894 MWh in the BAU, 441,742 MWh in the I-MRF, 640,019 MWh in the RDF, and 636,253 MWh in the No-MRF. This means an increase with respect to the first pathway ranging from 9% in the RDF scenario to 75% in the No-MRF scenario.

Lastly, it should be noted that the existing incineration facility dates from 1993 and, therefore, it is nearing the end of its lifetime. In this regard, the third pathway considers the incineration of all the available feedstocks in new facilities with higher efficiency. This would further increase the electricity generation potential in the reference scenario by 21% up to 836,681 MWh. For the future scenarios, the electricity generation potential would range from 580,323 MWh in the I-MRF scenario to 758,534 MWh in the RDF scenario.

Electricity generation potential

Figure 3-4 represents the accumulated electricity generation potential in the reference scenario 2019 and the 2040 scenarios considering AD and the three incineration pathways. The black horizontal line represents the actual amount of electricity fed into the electrical grid by the existing incineration facility in 2019 (166,607 MWh).

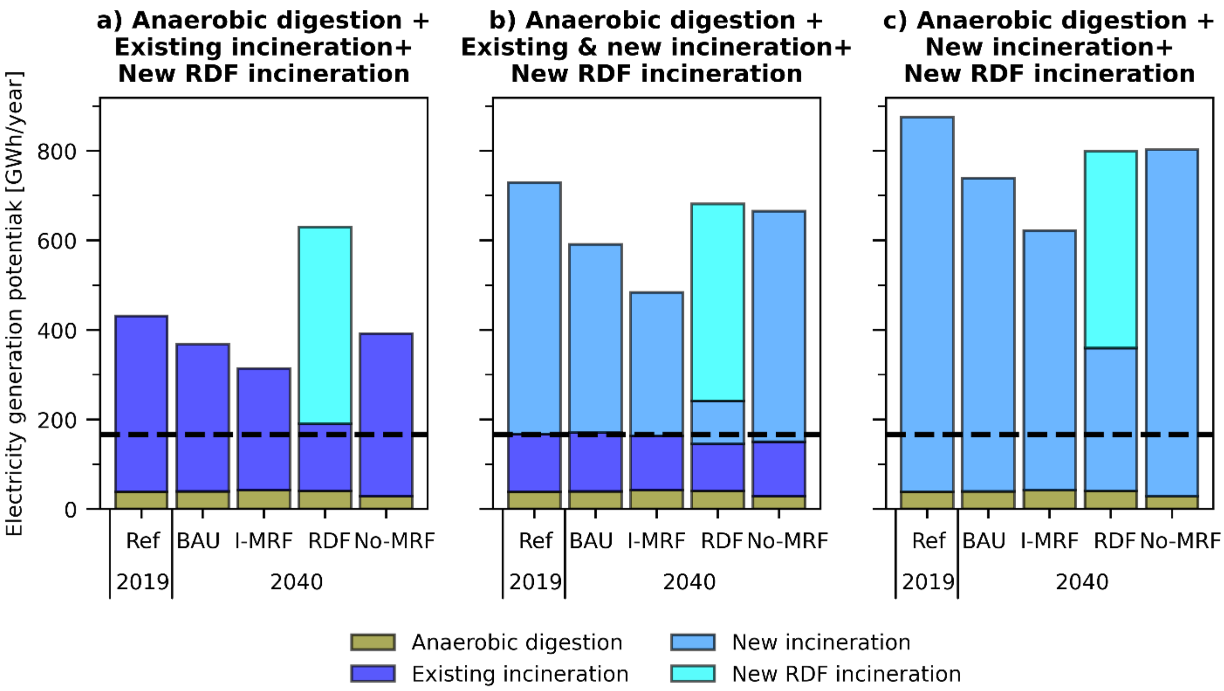

Figure 3-4. Electricity generation potential of municipal solid waste in Madrid under the reference scenario 2019 and the 2040 scenarios. Three incineration pathways are represented. The black horizontal dashed line represents the amount of electricity fed into the electrical grid by the existing incineration facility in 2019.

It can be observed that the contribution of AD is minor, ranging from 3% to 13% depending on the scenario. This could be expected since the organic feedstocks contain less than 10% of the gross energy recovery potential of MSW. Therefore, the electricity generation potential is largely driven by the incineration pathway chosen. In this regard, the investment in a new more efficient incineration facility would substantially increase electricity generation (pathways b and c in Figure 3-4). All in all, the electricity generation potential reaches between 430,867 and 874,571 MWh in the reference scenario 2019 and between 313,516 and 802,575 MWh in the 2040 scenarios.

3.5 Final remarks and future work

Mandatory separate collection and more ambitious recycling targets will certainly have an impact on the flow of waste materials through the MSW management system. However, this work quantitatively demonstrated that a higher separate collection and recycling do not necessarily compromise energy recovery (**RQ1**). For the specific case study of Madrid, it was found that under the most promising scenario for recycling as high as 71% of MSW represents potential WTE feedstocks. Furthermore, the gross energy recovery potential decreases only 28%, while the energy content of WTE feedstocks is only marginally affected. Within this context, envisioning a future with a relevant role for energy recovery appears highly likely.

The numeric results presented herein reflect the case study of Madrid and are based on a number of assumptions. Since MSW is a very regional issue, these numbers should not be directly extrapolated to other contexts. However, this work reveals some trends that are generally applicable. The waste streams susceptible to energy recovery can be classified into organic feedstocks, suitable for biochemical WTE conversion (e.g., AD), and combustible feedstocks, suitable for thermochemical WTE conversion (e.g., incineration). The availability of WTE feedstocks depends on the design of the MSW management system and can vary across space and even time. In any case, the combustible feedstocks concentrate most of the gross energy recovery potential, in part due to the high energy content of combustible materials (e.g., plastic) and in part due to the high moisture content of organic waste feedstocks. Consequently, a relatively higher energy recovery potential can be envisioned for WTE thermochemical pathways (e.g., incineration, gasification, and pyrolysis). The gross energy recovery potential of combustible feedstocks will certainly fall in the future as a result of the increased recycling of paper, cardboard, plastic, and textile. However, this decrease is unlikely to be of such magnitude to have a large-scale impact on the WTE sector. This is largely explained by inefficiencies in separate collection and materials recovery. First, there is a maximum feasible rate of separate collection, which for paper/cardboard, plastic packaging, and textile was set in this work at 85%, 85%, and 60%, respectively. Furthermore, some waste streams collected separately, such as the packaging waste, may contain high levels of impurities. Secondly, not all waste materials are subject to separate collection. In Madrid, for example, non-packaging plastic should not be separated with packaging materials. Finally, the efficiency of materials recovery through mechanical sorting at MRFs is also limited, even for new modern facilities.

A major conclusion drawn from this work is that the design and performance of the separate collection scheme have a large influence on the energy recovery potential of MSW. This work focused on Madrid, where separate collection of household waste is based on five street bins for plastic and metal packaging waste, paper/cardboard, glass, organic waste, and residual waste. While this collection scheme is representative for many municipalities across Spain (Gallardo et al., 2010) and other European countries (Seyring et al., 2016), alternatives exist that could further incentive separate collection thus affecting the energy recovery potential. Some examples are economic instruments such as deposit-refund and “pay-as-you-throw” (PAYT) schemes (Dubois et al., 2020). For example, the implementation of a PAYT scheme in the County of Aschaffenburg in Germany allowed this municipality to collect separately 86% of the recyclable materials, while the residual waste decreased from 380 to 55 kg per capita per year between 1995 and 2013 (Morlok et al., 2017). All in all, the impact of alternative collection schemes, such as deposit-refund and PAYT, on WTE remains largely unexplored.

The parametrized MFA framework presented in this chapter is simple to communicate and apply and, contrary to previous approaches, its basis is a detailed modelling of the flow of individual waste materials rather than aggregated socio-economic data. This represents a big advantage in the context of the circular economy since the consequences of changes in the flow of specific waste materials such as PET, aluminum, or textile can be predicted. The framework could be further improved in several ways. First, MSW generation has been projected in a rather simplistic way, assuming an annual variance that can be specified for each waste material individually. More complex forecasting methods can be implemented in the future to increase the robustness of the framework. For example, Peeters et al. (2015) proposed a method that uses market trends, including sales and products lifetime, to forecast plastic waste generation. Naturally, such a forecasting method would require a large volume of additional information (e.g., time- and site-specific sales for 18 materials). Secondly, the composition of the waste streams collected separately (e.g., packaging, paper/cardboard, glass, and organic waste) was modelled exogenously and assumed fixed across years. This was a major assumption since it implies that the separate collection behavior remains the same over time. Evidences from literature seem to indicate that the composition of waste streams is affected by the rate of separate collection. Eriksen et al. (2018) and Haupt et al. (2018c), for example, argue that the level of impurities in plastic waste streams is likely to increase with the rate of separate collection. Conversely, Puig-Ventosa et al. (2013) found a lower presence of impurities in the organic waste stream at higher rates of separate collection. Modelling the evolution of waste streams composition over time seems a hard task. This would require, for example, determining the time-dependent share of unwanted materials such as green waste, paper, textile, or wood in the packaging waste stream. Thirdly, the framework applies linear mathematical relationships to model complex mechanical, biochemical, and physicochemical relationships. For example, MRFs were modelled with static transfer coefficients. However, the transfer coefficients of a specific material (e.g., PET) vary nonlinearly with waste composition and other operation conditions (Tanguay-Rioux et al., 2021). Lastly, it is important to note that many characteristics that determine the suitability of a feedstock to undergo a specific WTE technology can hardly be captured in a prospective MFA model. An example is the particle size distribution, which is an important characteristic of RDF (Velis et al., 2013).

All in all, the analysis carried out in this chapter revealed that the future availability and characteristics of feedstocks would be adequate to sustain the operation of large-scale WTE facilities. In this regard, the following chapters will focus on developing assessment frameworks to analyze pathways to exploit this potential.

Chapter 4

Environmental assessment of phasing-out incineration in the municipal solid waste management system

This chapter addresses the potential life cycle environmental consequences of phasing-out incineration in Madrid (**RQ2**)[**]. The following Section 4.1 delves into the motivations to phase-out incineration and the importance of performing a holistic assessment of such a decision. Section 4.2 presents an integrated MFA and LCA framework for the systematic assessment of the environmental performance of MSW management systems. Section 4.3 applies the framework to the case study of Madrid. Finally, Section 4.4 synthesizes the main findings and conclusions.

4.1 Introduction

Incineration is a fundamental part of modern integrated MSW management systems. About 27% of the MSW generated in the EU in 2019 was incinerated for energy recovery (*cf.* Figure 1-1). In Chapter 1 it has been highlighted the role of incineration in reducing the landfill rate, avoiding CH_4 emissions from landfills, and providing a domestic source of energy. Despite these apparent benefits, the human health risk perception has contributed to consolidating a strong public opposition to incineration across many places. For example, Subiza-Pérez et al. (2020) found a low public acceptance of a new incineration facility in the city of San Sebastian (Spain) due to the relatively high health risk perception. Similar findings were found in the Yangtze River Delta region in China (Liu et al., 2021). Bena et al. (2019) surveyed the risk perception of incineration in Turin (Italy), and they found a higher concern among residents living close to the facility. Even people who support incineration would prefer that the facility will not be located close to their residential area (Huang et al., 2015). In Madrid, residents call for the immediate closure of the incineration facility, even though a recent study commissioned by the City Council concluded that there are no higher health risks in the vicinity of the facility (Pérez et al., 2018).

Within this context, moving away from incineration is increasingly claimed to protect the environment and human health (GAIA, 2013; Zero Waste Europe, 2022). In Barcelona (Spain), for example, an incineration facility was closed in 2004 after years of public protests (Subiza-Pérez et al., 2020). Focusing on Madrid, a new waste strategy for the city published in 2018 planned to close the existing incineration facility by 2025. Even though this new waste strategy was declared invalid in 2019, the phase-out of incineration continues to gain traction in the public debate, albeit the environmental consequences have not been fully explored yet.

[**] Based on: Istrate, I.R., E., Galvez-Martos, J.L., Dufour, J., 2021. The impact of incineration phase-out on municipal solid waste landfilling and life cycle environmental performance: Case study of Madrid, Spain. Science of the Total Environment 755: 142537

The business-as-usual scenario analyzed in Chapter 3 indicates that the residual/mixed waste and rejects could represent up to 58% of the MSW collected in Madrid by 2030. This would mean about 806,485 tonnes/year of waste ending up in a landfill in the absence of alternative treatments. Even the most promising scenario for recycling results in the production of 718,026 tonnes/year of residual/mixed waste and rejects by 2030. As emphasized in Chapter 1, the environmental consequences of future MSW management scenarios are not obvious. Indeed, they must be holistically assessed considering the future generation and composition of waste and other structural changes in the system (e.g., the improvement of separate collection) and the background system (e.g., the decarbonization of the electricity mix). This demands a comprehensive modelling framework that can predict the overall environmental performance of the MSW management system.

This chapter presents a quantitative assessment of the potential life cycle environmental consequences of phasing-out incineration in Madrid. A reference scenario describing the MSW management system of Madrid in 2019 was compared against a range of plausible scenarios describing the system in the years 2025, 2030, and 2040, with and without incineration. A major question was raised during the research performed in this chapter: Would a new incineration facility provide environmental benefits? In this regard, the future scenarios with incineration were assessed considering both the existing facility as well as a new facility with higher energy efficiency and improved metals recovery. To perform this analysis, the MFA principles adopted in Chapter 3 were embedded into an integrated MFA and LCA scenario analysis framework that allows quantifying the environmental impacts and benefits associated with the MSW management system. The framework presented in this chapter involves detailed MFA-based modelling of the life cycle inventory (LCI) of waste treatment processes. Thus, the LCIs and impact assessment are responsive to changes in waste composition and properties that inevitably arise under future scenarios.

4.2 Integrated material flow analysis and life cycle assessment framework

The integrated MFA and LCA scenario analysis framework builds on the MFA framework presented in Chapter 3. On the one hand, the MFA framework from Chapter 3 was used to quantify the mass and composition of the waste streams collected from households, commercial activities, and street cleaning. On the other hand, the fundamental modelling principle adopted in Chapter 3 (e.g., tracking the flow of each waste material) was applied here to develop predictive inventory models for waste treatment processes. More precisely, outputs from waste treatment processes (e.g., emissions) were linked to the mass and physicochemical properties of the 18 waste materials contained in the waste streams. In this regard, the waste materials properties used in Chapter 3 to assess the energy recovery potential (*cf.* Table 3-1) were complemented with the additional properties needed to quantify emissions, such as carbon, nutrients, and heavy metals content among others. The complete list of properties and sources can be found in Appendix A. The goal and scope of the framework, the modelling of waste treatment processes, the calculation of waste management indicators, and the implementation of capacity restrictions are explained in the following.

4.2.1 Goal and scope

Goal and functional unit

The framework presented herein aims at evaluating the life cycle emissions and environmental performance of MSW management systems to provide scientifically sound analysis on the consequences of waste strategies (e.g., phasing-out incineration). MSW is defined as waste collected from households, commercial activities, and street cleaning. The function of the studied system is the treatment of the MSW collected within defined spatial and temporal boundaries. The spatial boundary is limited to the geographical entity responsible for the management of this waste (e.g., municipality or region), while the temporal boundary covers the waste collected along one year. It is worth noting that emissions from MSW treatment might last for a long period of time, such as in the case of the decomposition of biodegradable materials in landfills or the application of compost on land. In this regard, a 100-year time horizon was adopted, which means that emissions occurring over 100 years following waste disposal were accounted for (Gentil et al., 2009).

The functional unit was defined as "*treatment of the mass of MSW collected in the studied region for one year*". A waste-generation based functional unit was considered (instead of, e.g., 1 tonne of MSW) to provide a more comprehensive picture of the magnitude of the impacts generated by the MSW management system (Ekvall et al., 2007).

System boundaries

The system boundaries start when the household, commercial, and street cleaning waste enter the treatment system, as illustrated in Figure 4-1. The MSW management system operates as follows: waste treatment facilities (i.e., material recovery facilities (MRFs), composting, anaerobic digestion (AD), incineration, and landfills) receive the waste and convert it to intermediates, residues, and/or products while consuming commodities and services (e.g., electricity, chemicals, or transportation) and generating emissions to air, water, and soil (e.g., GHGs and heavy metals). The intermediates are further processed within the system. For example, the rejects generated by MRFs can be incinerated or disposed of in a landfill, while the residual organic waste sorted at MRFs can be treated through AD or composting or disposed of in landfill. Residues, such as incineration ashes, are landfilled. The products recovered by the system include recyclable materials, compost, electricity, and biomethane. These products may require certain processing downstream to fulfill the same function as the market counterparts that they aim to substitute (see multifunctionality section below). Recycling is needed to convert recyclable materials into secondary materials, compost is spread on agricultural land, while biomethane is combusted in central or small-scale heating systems. Importantly, waste collection and transportation are not included within the system boundaries.

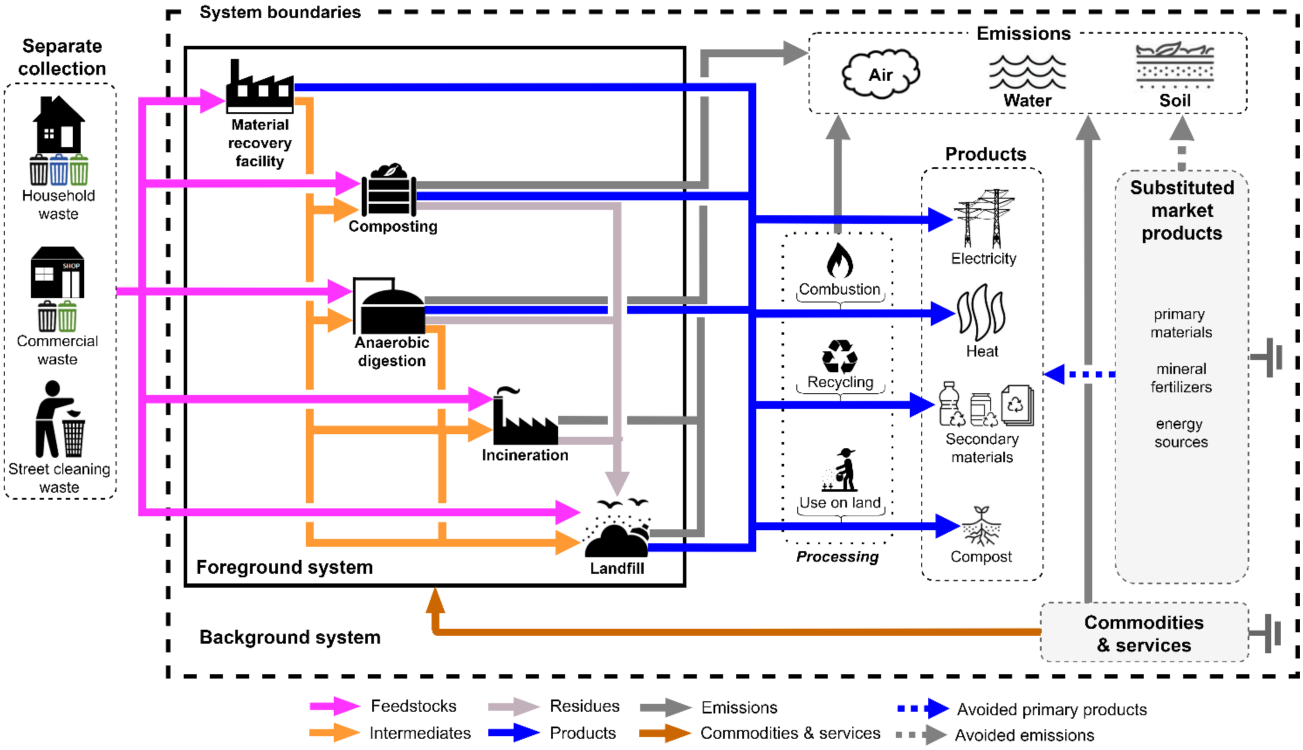

Figure 4-1. System boundaries of the municipal solid waste treatment system.

The system boundaries are divided into the foreground and background systems (Figure 4-1). The foreground system comprises the interconnected and interrelated processes engaged in the treatment of MSW that are located within the municipality (i.e., MRFs, composting, AD, incineration, and landfilling). The background system comprises the surrounding economic sectors that interact with the MSW management system, such as the supply of commodities and services (e.g., electricity) and the production of the substituted market products (e.g., primary materials or mineral fertilizers). Recycling, application of compost on land, and the use of the energy recovered are also part of the background system since these activities take place elsewhere outside the MSW management system.

Multifunctionality

The MSW management system is a multifunctional system since, in addition to treating waste as a service, it also supplies secondary materials (i.e., paper, plastic, metal, etc.), organic fertilizers (i.e., compost), and energy (e.g., electricity and biomethane). Since the goal is to evaluate the specific emissions and impacts associated with MSW management, the multifunctionality issue must be solved. The ISO 14044:2006 (and its amendment in 2020) establishes a hierarchy of three approaches for solving multifunctionality: (1) subdivision of the multifunctional system into monofunctional sub-systems, (2) system expansion –expanding the system boundaries to include the co-functions– or substitution –expanding the system boundaries to include the counterfactual market products displaced by the co-functions–, and (3) allocation, which is the partitioning of the inputs and outputs between the co-functions (European Commission, 2010; Heijungs et al., 2021). When focusing on MSW management, the substitution approach (also called "*avoided burdens approach*" or "*crediting*") is by far the most popular choice (Astrup et al., 2015; Laurent et al., 2014b; Viau et al., 2020). Accordingly, the materials and energy recovered from MSW substitute other functionally equivalent market products, and the MSW management system is credited for the impacts avoided.

Life cycle inventory modelling

The LCI modelling follows an attributional approach. This implies that average market data was used to model the environmental impacts of the foreground and background systems. In this context, secondary materials were assumed to substitute the corresponding average market of primary materials (e.g., secondary PET substitutes the market of primary PET based on a substitution ratio), compost substitutes the average market of mineral fertilizers, electricity substitutes an equivalent amount of electricity generated with the national mix, and biomethane substitutes the extraction, distribution, and combustion of an equivalent volume of natural gas. It is worth noting that substitution is usually applied in consequential LCA, where the substituted market products are included using marginal technologies or marginal market mixes (Schrijvers et al., 2016). The marginal suppliers are those most likely to be affected by a change in demand (Weidema et al., 1999). In the case of electricity supply, for example, natural gas power plants typically adapt their production to changes in demand (Mathiesen et al., 2009). Herein, attributional LCA is applied in combination with substitution under the premise that the recovery of materials and energy from MSW at a municipal scale will not trigger large-scale consequences on other markets, such as electricity and primary materials (European Commission, 2012, 2010; Laurent et al., 2014b).

The inventory of the foreground system was modelled based on parametrized mass and energy balances for each process as described later in Section 4.2.2. The inventories of background processes were obtained from the Ecoinvent database v3.7.1 cut-off system model (Wernet et al., 2016). The inventories of recycling, application of compost on land, and the use of the energy recovered were modelled following a hybrid approach that combines explicit modelling (e.g., considering own recycling efficiencies) with data from the Ecoinvent database (e.g., the impacts per kg of material recycled).

Life cycle impact assessment

The impact assessment was performed with the European Commission's recommended Environmental Footprint (EF) method 2.0 (Fazio et al., 2018). EF 2.0 includes a list of 16 impact categories, from which the following 13 were identified as relevant for MSW management: climate change (100-years timeframe), acidification, freshwater eutrophication, terrestrial eutrophication, marine eutrophication, photochemical ozone formation, ozone depletion, particulate matter formation, human toxicity cancer effects, human toxicity non-cancer effects, ecotoxicity, non-renewable energy depletion, and minerals and metals depletion (Laurent et al., 2014b; Mayer et al., 2019). The biogenic CO_2 was separately calculated and considered neutral with respect to climate change, following common practice in waste LCA (Christensen et al., 2009). The biogenic carbon that remains in landfills after 100 years was also considered neutral, while the biogenic carbon that remains stored in soil after 100 years of using compost was considered an avoided burden.

4.2.2 Waste treatment processes modelling

This section describes the modelling of the inventory for the following waste treatment processes: MRFs, composting, AD, existing and new incineration, sanitary landfill, recycling, and the application of compost on land. The processes and data presented herein were particularized as much as possible for the case study of Madrid. Primary data was obtained from the official reports covering the period 2015-2019 (Madrid City Council, 2019, 2018, 2017b, 2016, 2015). However, these reports provide rather limited technology performance data. Thus, most of the data used in the LCIs come from the literature. To maintain a certain degree of technological, geographical, and temporal representativeness, studies performed within the European context and published over the last ten years were prioritized. Furthermore, average data from several studies was used whenever possible to avoid biased inventories.

Material recovery facilities (MRFs)

The mass balance of MRFs was modelled with material-specific transfer coefficients, as described in Chapter 3. Transfer coefficients for the existing MRFs dedicated to sorting residual and packaging waste were presented in Table 3-5. MRFs for sorting paper/cardboard and glass were also implemented herein. The MRF for paper/cardboard recovers 98.6% of paper and 100% of cardboard, while the MRF for glass achieves a recovery efficiency of 95.5% (Haupt et al., 2018b). MRFs require electricity and diesel for daily activities and steel wire for packaging the recovered materials (Table 4-1).

Table 4-1. Raw materials and energy consumption at material recovery facilities (MRFs).

Input	Unit	MRF for residual waste	MRF for packaging waste	MRF for paper/ cardboard	MRF for glass
Electricity	kWh/tonne waste	11.56 (8,9,10,12)	33.0 (1,2,3,6,7,8,11,12)	10.85 (4,12,13)	11.1 (4,7,12,14)
Diesel	kg/tonne waste	0.53 (8,12)	1.56 (2,6,7,8,11)	1.45 (12,13)	0.45 (12)
Steel wire	kg/tonne waste	1.50 (15)	3.00 (2)	–	–

Source: [1] Civancik-Uslu et al. (2021); [2] Andreasi Bassi et al. (2020); [3] Faraca et al. (2019); [4] Haupt et al. (2018b); [5] Pinasseau et al. (2018); [6] Cimpan et al. (2016); [7] Cimpan et al. (2015b); [8] Pressley et al. (2015); [9] Montejo et al. (2013); [10] Carre et al. (2015); [11] Fitzgerald et al. (2012); [12] Bovea et al. (2010); [13] Merrild et al. (2009); [14] Larsen et al. (2009); [15] assumed 50% of the steel wire required by MRFs for packaging waste.

Composting

Composting is the aerobic degradation of organic matter by microorganisms under controlled conditions. Composting technologies are usually classified in open, if the process takes place outdoors (e.g., windrow composting and static piles), and enclosed, if the process takes place in enclosed buildings (e.g., tunnels composting and in-vessel) (Krogmann et al., 2011). The main difference between open and enclosed technologies is that in the latter case the exhaust gases can be collected and treated. The most popular technology for the treatment of exhaust gases is the biofilter. Following current practices in Madrid, the composting process modelled herein takes place in enclosed tunnels equipped with biofilters.

During composting, microorganisms partially degrade the biodegradable matter resulting in a 50-70% reduction of the initial waste mass (Smith et al., 2001). The rate of biodegradation varies between waste materials. Some materials decay rapidly (e.g., food waste), others have a relatively slow rate of biodegradation (e.g., cellulose and paper), and others are not biodegradable at all (e.g., plastics). To consider this variety, the biodegradation process was modelled with the material-specific rates summarized in Table 4-2. The degradation of the carbon and nitrogen contained in the waste deserves special attention due to the associated gaseous emissions (Andersen et al., 2011, 2010). The rate of degradation of carbon ranges from 23% to 66%, depending on the waste material. Furthermore, between 10% and 70% of the nitrogen can be lost due to ammonia volatilization and denitrification.

Table 4-2. Rate of dry matter, biogenic carbon, and nitrogen content of each waste material that is biodegraded during composting. The table also shows the post-treatment reject rate for each waste material (% mass).

Material	Dry matter degradation	Biogenic carbon degradation	Nitrogen degradation	Post-treatment reject (5)
Food waste	59.2 (1)	65.7 (1)	70.0 (4)	10
Green waste	46.7 (1)	63.1 (1)	10.0 (4)	10
Cellulose	19.7 (2)	23.4 (2)	25.0 (4)	20
Paper	24.0 (1)	28.8 (1)	25.0 (4)	20
Cardboard	35.1 (1)	37.9 (1)	25.0 (4)	20
PET	–	–	–	90
HDPE	–	–	–	90
LDPE	–	–	–	90
PP	–	–	–	90
Other plastic packaging	–	–	–	90
Non-packaging plastic	–	–	–	90
Cartons and alike	26.3 (3)	28.43 (3)	18.75 (4)	90
Glass	–	–	–	90
Ferrous metal	–	–	–	90
Non-ferrous metal	–	–	–	90
Textile	4.50	5.00 (5)	–	90
Wood	15.4 (1)	18.0 (1)	25.0 (4)	80
Others	–	–	–	90

Source: [1] US EPA (2003); [2] Average between paper and wood; [3] 75% of cardboard value; [4] Turner et al. (2016); [5] Cobo et al. (2018b); [5] Levis and Barlaz (2011)

Most of the biogenic carbon degraded during composting (97% on mass) is converted to biogenic CO_2 (Table 4-3). A relatively small fraction (2.7%) can be converted to (biogenic) CH_4 due to the occurrence of anaerobic conditions inside the waste pile (Boldrin et al., 2009). Furthermore, small amounts of CO can be produced due to factors that may include increased temperatures, moisture content, or the availability of oxygen (Stegenta-Dąbrowska et al., 2020). Herein, it was assumed that 0.3% of the degraded carbon is converted to CO after Andersen et al. (2010). The nitrogen degraded during composting contributes primarily to the formation and volatilization of NH_3 (89.5% of the degraded nitrogen) and, to a lesser extent, to the formation of N_2O as a by-product of nitrification and denitrification (1.4%).

The composting facility is equipped with biofilters to control odours. Therefore, the exhaust gases are collected (through a ventilation system based on exhaust fans) and passed through the biofilters which adsorp and then biologicaly oxidize the odorous compounds. The most common biofilter materials are pine-based wood chips, activated carbon, peat, soil, and compost (UK Environment Agency, 2013).

CH_4 oxidation efficiencies for biofilters have been reported in the range from 33% to 100% (Boldrin et al., 2009). Here, an average oxidation efficiency of 75% was retrieved from Turner et al. (2016). Note that the oxidation of CH_4 in the biofilters results in additional emissions of biogenic CO_2. NH_3 is easily removed through biofilters by adsorption/absorption mechanisms (UK Environment Agency, 2013). Thus, a 99% removal efficiency was assumed after Boldrin et al. (2011).

Table 4-3. Parameters to model emissions from composting.

Parameter	Unit	Value	Source
Carbon conversion to CO_2	% biogenic carbon degraded	97	Boldrin et al. (2009)
Carbon conversion to CH_4	% biogenic carbon degraded	2.7	Boldrin et al. (2009)
Carbon conversion to CO	% biogenic carbon degraded	0.3	Andersen et al. (2010)
Nitrogen conversion to NH_3	% nitrogen degraded	89.5	Boldrin et al. (2011)
Nitrogen conversion to N_2O	% nitrogen degraded	1.4	Boldrin et al. (2011)
Removal efficiency of CH_4	% vol. produced	75	Turner et al. (2016)
Removal efficiency of NH_3	% vol. produced	99	Boldrin et al. (2011)

The composition of the solid product of composting was estimated through the mass balance between the composition of the input waste and the mass of biodegradable matter degraded during composting (Pace et al., 2018). No leaching of metals is usually assumed to occur due to leachate recirculation (Boldrin et al., 2011; ten Hoeve et al., 2019). Therefore, all the heavy metals and other chemical elements contained in the input waste are transferred into the solid product.

The solid product of composting undergoes a mechanical post-treatment based on trommel screens that separate unwanted materials (e.g., plastics) and large particles. The post-treatment was modelled with material-specific transfer coefficients (Table 4-2) that describe the partitioning of each material among the two outputs: the rejected stabilized material, which is landfilled, and the compost, which can be spread on agricultural land in substitution of mineral fertilizers. The moisture content of the rejected stabilized material and compost was assumed equal to 30% (Cobo et al., 2018b).

Composting consumes electricity (for odor control, aeration, and post-treatment), diesel (to operate grinders and front-end loaders), and water. Tunnels composting requires about 69 kWh electricity/tonne waste (Pinasseau et al., 2018), 1.21 kg diesel/tonne waste (Boldrin et al., 2009; Cobo et al., 2018b; Smith et al., 2001; Turner et al., 2016; van Haaren et al., 2010), and 150 kg water/tonne waste (Pinasseau et al., 2018).

Anaerobic digestion (AD)

AD is a biological conversion process in which microorganisms break down organic matter in the absence of free oxygen (see also Section 1.2.1 in Chapter 1 for an overview of the AD process and technology). The AD process used in Madrid consists of a dry mesophilic, one stage, and one phase process operating at 37 °C with 21 days retention time.

The input organic waste undergoes a pre-treatment based on trommel screens to remove impurities. This pre-treatment generates a stream of rejects that was modelled with the transfer coefficients presented in Table 3-5 in Chapter 3. The remaining waste after pre-treatment is fed to the digesters.

The biogas produced in AD was assumed to consist of a mixture of CH_4 and CO_2. The share of each gas in the biogas is determined by the composition of the waste. Fat-rich materials increase the share of CH_4, while carbohydrate-rich materials produce a biogas with similar shares of CH_4 and CO_2 (Angelidaki and Batstone, 2011). The composition of the biogas produced from each waste material was calculated with the Buswell equation (data in Appendix A). Thus, following the approach described in Chapter 3, the production of CH_4 was quantified using the material-specific CH_4 potentials and the efficiency of AD. The efficiency of AD is represented by the rate of VS that are effectively degraded. This efficiency was established in Chapter 3 at 75% for the AD of organic waste collected separately and at 70% for the AD of residual organic waste.

Fugitive emissions of biogas are common due to leaks from gas engines, pipes, valves, biogas storage, biogas upgrading, etc. (Bakkaloglu et al., 2021; Kvist and Aryal, 2019; Scheutz and Fredenslund, 2019). In this work, overall fugitive emissions of biogas were assumed equal to 3.9% (Table 4-4). Note that this rate is higher than the 2.5% assumed in Chapter 3 for an AD facility where biogas is burned for electricity generation. This difference is because AD facilities with biomethane upgrading (as the one available in Madrid) exhibit higher rates of leakage due to the additional fugitive emissions from the upgrading section (Scheutz and Fredenslund, 2019).

The available biogas after subtracting fugitive emissions is primarily upgraded to biomethane through water scrubbing (46.5% of biogas volume) and burned in engines for electricity generation (41.8%). A small fraction of the biogas (1.7%) is combusted in boilers to satisfy the internal demand of heat, while the remaining 10% is flared[††] (on-site data provided by Madrid City Council (2019)).

Biomethane production was estimated from the volume of biogas dedicated to upgrading and the purity of the biomethane, which achieves 98.5% after water scrubbing (Bauer et al., 2013; Madrid City Council, 2019). The biomethane is fed into the natural gas network after which it is combusted in a central or small-scale heating system in substitution of natural gas. Electricity generation from biogas combustion in engines was calculated with the gross electrical efficiency and the oxidation efficiency

[††] 15% biogas flaring was assumed in Chapter 3. AD facilities with biomethane upgrading exhibit lower share of biogas flaring due to the additional compressed storage available at the upgrading section.

of CH_4 (estimated in Chapter 3 at 37.5% and 99%, respectively). It was assumed that all the electricity generated by AD is fed to the grid.

Table 4-4. Parameters to model emissions from anaerobic digestion and biogas management.

Parameter	Unit	Value	Source
VS degradation efficiency for organic waste collected separately	% VS	75	Chapter 3
VS degradation efficiency for residual organic waste	% VS	70	Chapter 3
Fugitive emissions	% biogas	3.9	
Biogas to boiler	% biogas*	1.7	Madrid City Council (2021)
Biogas to flare	% biogas*	10	Madrid City Council (2021); Smith et al. (2021)
Biogas to upgrading	% biogas*	46.5	Madrid City Council (2021)
Biogas to engines	% biogas*	41.8	Madrid City Council (2021)
Biomethane purity	% CH_4	98.5	Bauer et al. (2013); Madrid City Council (2021)
Methane oxidation efficiency	% CH_4	99	Chapter 3
Gross electrical efficiency of engines	% LHV of biogas	37.5	Chapter 3

* Available biogas after subtracting fugitive emissions

Emissions of biogenic CH_4 and CO_2 from combustion processes (i.e., flaring, boiler, engines, and biomethane combustion in a heating system) were computed from the CH_4 and CO_2 content of the biogas. The CO_2 is directly released to the atmosphere, while 99% of the CH_4 is oxidized to CO_2 and the remaining is released. Other compounds that cannot be directly related to the composition of biogas (since they depend heavily on operating conditions), such as incomplete combustion products (i.e., CO) and N_2O, were modelled with fixed emission factors that describe the amount of compounds emitted per m^3 of CH_4 combusted (Table 4-5). Thus, these emissions are linked to the CH_4 content of the biogas or biomethane.

Table 4-5. Emission factors for the combustion of biogas and biomethane (kg compound/m^3 CH_4).

Compound	Biogas	Source	Biomethane	Source
Carbon monoxide, biogenic	0.011	*1*	6.55E-03	*3*
Dinitrogen monoxide	5.73E-05	*1*	4.20E-04	*3*
Nitrogen oxides	7.23E-03	*1*	1.22E-02	*3*
Sulfur dioxide	6.87E-04	*2*	4.62E-04	*3*
NMVOC	3.58E-04	*1*	–	–
Benzene	–	–	3.36E-04	*3*
Benzo(a)pyrene	3.58E-11	*2*	8.40E-09	*3*
Toluene	–	–	1.68E-04	*3*
Dioxin, 2,3,7,8 tetrachlorodibenzo	3-44E-14	*1*	–	–
Particulates, < 2.5 um	1.61E-05	*2*	8.40E-05	*3*
Particulates, 2.5-10 um	7.37E-04	*2*	–	–

Source: [1] Calculated from Nielsen et al. (2010); [2] Møller et al. (2011); [3] Calculated from Ecoinvent v3.7.1

The digestate undergoes dewatering in a decanter centrifuge to reduce the moisture content. The dry mass of dewatered digestate and its chemical composition was modeled through the mass balance between the dry mass of the input waste, the mass of biogas, and the mass of chemicals lost during dewatering (Pace et al., 2018). Dewatering may result in a reduction in the range of 20%-95% of the nutrient content of digestate (Di Maria and Micale, 2016). Average losses of carbon and nutrients associated with physical dewatering were obtained from Drosg et al. (2015), as summarized in Table 4-6. Due to the lack of information regarding the fate of other chemicals (e.g., heavy metals), it was assumed that they are all transferred into the dewatered digestate. The moisture content of the dewatered digestate is approximately 45% (Sardarmehni et al., 2021). The liquid stream from dewatering is partially recirculated into the digesters as process water and the rest is treated at a wastewater treatment plant. The volume of wastewater produced by AD was modeled exogenously as described below.

The dewatered digestate is either disposed of in a landfill or stabilized through composting in tunnels to further reduce the biodegradable matter. The second option results in the production of compost that could be used on land in substitution of mineral fertilizers. In 2019, about 64% of the digestate produced in Madrid was stabilized and 36% was landfilled (Madrid City Council, 2019). Composting of digestate takes place in tunnels equipped with biofilters. It was assumed that about 35% of the carbon and 12% of the nitrogen contained in the digestate is degraded during composting (Jensen et al., 2017). The rest of the process was modeled following the same approach and using the same data as for the composting of organic waste described above. The moisture content of the solid produced was assumed to be 30% (Cobo et al., 2018b).

Table 4-6. Parameters to model the dewatering and composting of digestate.

Parameter	**Unit**	**Value**	**Source**
	Digestate dewatering		
Moisture content dewatered digestate	% mass	45	Sardarmehni et al. (2021)
Carbon lost dewatering	% mass	35	Drosg et al. (2015)
Nitrogen lost dewatering	% mass	70	Drosg et al. (2015)
Phosphorus lost dewatering	% mass	40	Drosg et al. (2015)
Potassium lost dewatering	% mass	75	Drosg et al. (2015)
Other chemicals lost dewatering	% mass	0	–
	Digestate composting		
Biogenic carbon degradation	% C biogenic	35	Jensen et al. (2017)
Nitrogen carbon degradation	% N	12	Jensen et al. (2017)
Moisture content compost	% mass	30	Cobo et al. (2018b)

AD requires electricity, diesel (machineries operating on site to move waste), and water. The heat demand is satisfied internally through the combustion of part of the biogas in a boiler. For an AD facility equipped with upgrading and digestate composting, raw materials and energy consumptions were estimated at 101 kWh electricity/tonne waste, 1.65 kg diesel/tonne waste, and 1,088 kg water/tonne waste. Furthermore, the process generates over 945 kg wastewater/tonne waste, which is treated at a wastewater treatment plant. The impacts of wastewater treatment were obtained from the Ecoinvent v3.7.1 database.

Existing incineration facility

The incineration facility of Madrid dates from 1993 and has an average capacity of 900 tonnes waste/day. The facility consists of three lines in parallel based on the circulating fluidized bed technology. The fluidized beds operate at 950 ºC with a residence time of the flue gases of approximately 4 seconds. The input waste (with a typical LHV of 14 MJ/kg) is mixed with hot sand by air supplied through the bottom of the reactor. The facility is equipped with a boiler, a steam turbine, and a power generator. Each line generates 41 tonnes of steam per hour at 425 ºC and 47 bars. The net power generation capacity achieves 23 MW, while the average net electrical efficiency achieves 12% with respect to the waste LHV (as indicated in Chapter 3 and in Table B-1 in Appendix B). Heat is not recovered due to the lack of heat demand and infrastructure for its distribution in the vicinity of the facility. The air pollution control (APC) system is composed of cyclones and bag filters for removing particles, semi-dry absorption with calcium hydroxide for removing acid gases, selective catalytic reduction (SCR) for removing NO_x, and activated carbon injection for removing dioxins and mercury.

When modelling the LCI of incineration, there is an agreement in the literature in distinguishing between waste- and process-specific emissions (Doka, 2013; Hellweg et al., 2001; Koehler et al., 2011; Riber et al., 2008). Waste-specific emissions are those for which a clear dependency on waste composition can be established. For example, CO_2, SO_2, and heavy metals emissions depend on the carbon, sulfur, and heavy metals content of the waste. Conversely, process-specific emissions are those that depend much more on operation conditions (i.e., temperature, fuel/oxygen mixing ratio, fuel residence time, etc.) and the APC system. Incomplete combustion products, such as CH_4, CO, and dioxins, are examples of process-specific emissions. There are exceptions, such as nitrogen compounds, that are both waste- and process-specific emissions as will be explained later.

Waste-specific emissions were modeled with transfer coefficients that describe how much of each chemical element is transferred to the flue gas (Table 4-7). It is important to note that the transfer coefficients are measured at the stack, i.e., they include the efficiency of the APC system. On-site measured emissions of HCl, SO_2, HF, Cd, Cu, Cr, Hg, Ni, Pb, and Zn were used to calculate transfer coefficients for Cl, S, F, Cd, Cu, Cr, Hg, Ni, Pb, and Zn, as described in Table B-2 in Appendix B. The remaining transfer coefficients were obtained from the literature. It should be noted that only the chemical elements contained in burnable materials can be transferred to the flue gas (see Table 3-1 for the burnability factors of waste materials). The chemical elements contained in inert materials (e.g., glass and metal) are completely transferred to the bottom ash.

Table 4-7. Transfer coefficients of chemical elements to the flue gas during incineration (% mass of chemicals contained in the waste).

Chemical element	Transfer coefficient	Source	Chemical element	Transfer coefficient	Source
Carbon, fossil	98.6	*1*	Cobalt	0.20	*1*
Carbon, biogenic	98.6	*1*	Copper	0.04	*2*
Nitrogen	99.0	*1*	Chromium	0.52	*2*
Sulphur	0.26	*2*	Lead	3.70	*2*
Chlorine	0.99	*2*	Mercury	58.04	*2*
Fluorine	5.56	*2*	Molybdenum	0.53	*3*
Bromine	0.00	*3*	Nickel	0.06	*2*
Phosphorus	0.08	*1*	Selenium	1.17	*3*
Antimony	0.94	*1*	Silver	0.06	*3*
Arsenic	0.00	*4*	Thallium	0.00	*1*
Barium	0.35	*3*	Tin	0.36	*1*
Beryllium	0.10	*3*	Vanadium	0.05	*1*
Cadmium	1.64	*2*	Zinc	0.21	*2*

Source: [1] Boesch et al. (2014); [2] Calculated from on-site measured emissions; [3] Doka (2013); [4] Chen and Christensen (2010)

Process-specific emissions of incomplete combustion products were modelled with fixed emission factors that represent the mass of compound emitted per tonne of waste incinerated (Table 4-8). This means that these emissions are a linear function of the mass of waste incinerated regardless of its composition. Emission factors for CO, dioxin, and particles were obtained from on-site measured emissions facilitated by the City Council on the annual reports. Other process-specific emission factors were retrieved from Boesch et al. (2014).

Table 4-8. Process-specific emission factors of incomplete combustion products in incineration.

Compound	Emission factor	Unit
Carbon monoxide, fossil	0.096	kg/tonne waste
Methane, fossil	0.055	kg/tonne waste
NMVOC	0.267	kg/tonne waste
Benzene	3.64E-03	kg/tonne waste
Benzene, hexachloro-	6.71E-06	kg/tonne waste
Benzene, pentachloro-	1.72E-05	kg/tonne waste
Benzo(a)pyrene	7.77E-08	kg/tonne waste
Phenol, pentachloro-	5.75E-07	kg/tonne waste
Toluene	7.38E-03	kg/tonne waste
Dioxin, 2,3,7,8 tetrachlorodibenzo	7.48E-11	kg/tonne waste
Particulates, < 2.5 um	6.68E-03	kg/tonne waste
Particulates, 2.5-10 um	3.36E-05	kg/tonne waste

Nitrogen compounds (i.e., NO_x, NH_3, and N_2O) were modeled as both waste- and process-specific emissions (Table 4-9). NO_x is originated from both the nitrogen content of the waste (waste-derived NO_x) and the combustion air (thermal-derived NO_x) (Doka, 2013). The waste-derived NO_x is produced from the combustion of the nitrogen content of the waste. Thermal-derived NO_x, on the other hand, is originated due to the oxidation of the elemental nitrogen content of the combustion air at elevated temperatures. Furthermore, its formation is largely associated with the heterogeneous nature of the waste and irregular operation conditions (Doka, 2013). Doka (2013) proposes a conservative assumption according to which 50% of NO_x is allocated to the nitrogen content of the waste (waste-derived NO_x) and 50% to the nitrogen content of the combustion air (thermal-derived NO_x). NO_x emissions from the incineration facility of Madrid were reported equal to 88.2 mg/Nm^3 flue gas or 0.846 kg/tonne waste (Pérez et al., 2018). Therefore, thermal-derived NO_x was modelled with an emission factor of 0.423 kg NO_x/tonne waste. The waste-derived NO_x was modeled with an emission factor that describes the mass of NO_x produced per mass of nitrogen transferred to the flue gas. Thus, an emission factor of 0.062 kg NO_x/kg nitrogen to flue gas was calculated for Madrid.

The process- and waste-specific emission factors of NH_3 were obtained from Doka (2013) and equal 0.009 kg NH_3/tonne and 0.001 kg NH_3/kg nitrogen to flue gas, respectively. Similarly, a waste-specific emission factor of 0.004 kg N_2O/kg nitrogen to flue gas was assumed after Doka (2013). It is worh noting that the treatment of NO_x by SCR increases the emission of NH_3, due to the addition of NH_3 to the flue gas, as well as N_2O, due to the reducing conditions. These additional emissions are included in the emission factors mentioned above.

Table 4-9. Waste- and process-specific emission factors of nitrogen-based compounds in incineration.

Compound	Emission type	Emission factor	Unit
Nitrogen oxides	Process-specific	0.423	kg/tonne waste
Nitrogen oxides	Waste-specific	0.062	kg/kg nitrogen to flue gas
Ammonia	Process-specific	0.009	kg/tonne waste
Ammonia	Waste-specific	0.001	kg/kg nitrogen to flue gas
Dinitrogen oxide	Waste-specific	0.004	kg/kg nitrogen to flue gas

The incineration facility of Madrid produces on average 37.5 kg bottom ash/tonne waste and 73.8 kg fly ash/tonne waste (Madrid City Council, 2019). Ferrous metal (scrap steel) is recovered from the bottom ash through ferromagnetic separation with an efficiency of 80% (Neuwahl et al., 2019). This efficiency represents the mass of ferrous metal recovered per unit mass of material entering the incineration (i.e., 0.8 kg recovered per input kg). After the separation of metals, the bottom ash is disposed of in the sanitary landfill, while the fly ash is collected in bags and disposed of in a security landfill. The LCI associated with the disposal of incineration ashes was obtained from the Ecoinvent v3.7.1 database.

Incineration consumes energy, water, and reagents for the operation of the APC system (Neuwahl et al., 2019). Electricity is needed to run the process (e.g., to run pumps and fans and for metals recovery from the bottom ash). However, this demand is satisfied internally. Light fuel oil is used as auxiliary burner to pre-heat the combustion air and to increase the temperature during start-up. Water is required for cooling and flue-gas cleaning. Calcium hydroxide is used to neutralize acid gases. Ammonia water is the most typical reagent to remove NO_x from the flue-gas. Furthermore, the SCR generally uses titanium dioxide as catalyst material. Activated carbon is typically injected into the flue-gas for removing dioxin and mercury. Table 4-10 summarizes the raw materials and energy consumed by incineration.

Table 4-10. Raw materials and energy input to incineration.

Input	Value	Unit	Source
Light fuel oil	2.12	kg/tonne waste	Neuwahl et al. (2019)
Water	150	kg/tonne waste	Bisinella et al. (2021)
Calcium hydroxide	12.0	kg/tonne waste	Neuwahl et al. (2019)
Ammonia water	2.25	kg/tonne waste	Neuwahl et al. (2019)
Titanium dioxide	0.04	kg/tonne waste	Ecoinvent v3.7.1
Activated carbon	0.50	kg/tonne waste	Beylot et al. (2017); Neuwahl et al. (2019)

New incineration facility

The new incineration facility was assumed to consist of a moving grate furnace as this technology is used in about 90% of the existing facilities in Europe (Neuwahl et al., 2019) and achieves higher efficiency than the fluidized bed technology (Chen and Christensen, 2010). The new facility was assumed to be similar to the existing one concerning treatment and power generation capacities and APC system. The emission levels modelled for the existing facility were found well below the BAT threshold levels (as will be discussed below). Therefore, the new facility was modelled considering the same transfer coefficients (*cf.* Table 4-7), process-specific emission factors (*cf.* Table 4-8 and Table 4-9), and raw materials and energy consumption (*cf.* Table 4-10). The major differences between the existing and new facility are the net electrical efficiency, the production rate of ashes, and the recovery of metals from the bottom ash (Table 4-11). In particular, the new facility was assumed to recover both ferrous metals (scrap steel) and non-ferrous metals (aluminium) from the bottom ash, following BAT practices (Neuwahl et al., 2019). The recovery of aluminium is performed through eddy current separation with an efficiency of 50%.

Table 4-11. Parameters difference between the existing incineration facility of Madrid and a new facility.

Parameter	Unit	Existing facility	New facility	Comments
Net electrical efficiency	% waste LHV	12	26	Efficiencies discussed in Section 3.4.5 in Chapter 3
Bottom ash	kg/tonne waste	37.5	250	Fluidized bed produces less bottom ash and more fly ash than grate furnace due to different operation conditions (Chen and Christensen, 2010; Neuwahl et al., 2019)
Fly ash	kg/tonne waste	73.8	65	
Ferrous metal recovery	% input mass	80	80	The same recovery efficiency was assumed for both facilities
Aluminium recovery	% input mass	0	50	The new facility recovers non-ferrous metals (aluminium) from the bottom ash (Neuwahl et al., 2019)

Sanitary landfill

The current landfill of Madrid dates from the year 2000 and occupies an area of 825,000 m^2 with a volumetric capacity of 22.7 million m^3 distributed between seven cells. The landfill is equipped with landfill gas (LFG) and leachate collection systems.

The decomposition of biodegradable materials in landfills is a slow process. Consequently, the generation of LFG and leachate occur over a long period of time following waste disposal. As mentioned above, a common choice in the LCA literature is to take a time horizon of 100 years (Lee et al., 2017; Manfredi and Christensen, 2009; Sauve and van Acker, 2020; Wang et al., 2020). Therefore, the following paragraphs describe the modelling of LFG and leachate generation, collection, and emissions over 100 years following waste disposal (i.e., the waste is disposed of in the landfill in year zero).

The LFG was assumed to consist of a mixture of 50% CH_4 and 50% CO_2 (IPCC, 2006a). The generation of CH_4 in the landfill over 100 years was estimated with a first-order decay model (US EPA, 2011). As shown in Eq. 4-1, this model calculates the generation of CH_4 from the mass of waste materials landfilled and their CH_4 potentials and decay rates.

$$Q_{CH_4} = \sum_{y\in\{1\ldots100\}} \sum_{k\in K} FINM_k \cdot (1 - moist_k) \cdot vs_k \cdot bmp_k \cdot \left(e^{-dr_k\cdot(y-1)} - e^{-dr_k\cdot y}\right) \qquad \textit{Eq. 4-1}$$

where Q_{CH_4} is the volumen of CH_4 generated over 100 years following waste disposal (m^3), $FINM_k$ is the mass of each waste material k landfilled in year zero (tonnes), $moist_k$ is the moisture content of waste materials (% mass), vs_k is the VS content of waste materials (% dry mass), bmp_k is the BMP of waste materials (m^3 CH_4/tonne VS), dr_k is the decay rate of waste materials ($year^{-1}$), and y refers to for the analysed year that ranges from one to 100.

The BMP represents the maximum CH_4 potential that could be achieved under laboratory conditions, and it is an inherent property of each waste material (Table 3-1). The decay rate, on the other hand, represents the rate at which materials are biodegraded in the landfill and depends on climate and feedstock conditions (e.g., temperature, precipitation, and pH). The decay rate determines the percentage of the maximum CH_4 potential that is achieved over 100 years. In this work, the default decay rates recommended in the IPCC Guidelines (IPCC, 2006a) for a wet template climate zone were used (Table 4-12). Overall, it is worth noting that by using material-specific CH_4 potentials and decay rates, the model can capture the consequences of changes in waste composition on the generation of LFG.

Table 4-12. Decay rate of waste materials in a landfill located in a wet template climate zone.

Material	**Decay rate ($year^{-1}$)**
Food waste	0.185
Green waste	0.100
Cellulose	0.100
Paper	0.060
Cardboard	0.060
Cartons and alike	0.045
Textile	0.060
Wood	0.030
Rest of materials	–

The biogenic carbon that is not degraded within the 100 years timeframe was assumed to remain stored in the landfill. The equivalent mass of biogenic CO_2 that remains stored in the landfill was calculated from the mass of biogenic carbon contained in the waste materials and the mass of biogenic carbon converted to LFG.

Once generated, the LFG is collected and burned in engines for electricity generation with an oxidation efficiency of CH_4 of 99%. LFG engines have a gross electrical efficiency of about 32.7% (Anshassi and Townsend, 2021; Kirkeby et al., 2007; Manfredi et al., 2010, 2009; Slorach et al., 2019; Wang et al., 2020), while the net efficiency achieves about 30.7% after subtracting the electricity used internally. Emissions were modelled as described above for biogas combustion.

Since not all the LFG can be collected, some of it passes through the landfill top cover and is released to the atmosphere (the so-called dispersive emissions). The uncollected CO_2 is directly released to the atmosphere, while the uncollected CH_4 is partially oxidized to CO_2 in the landfill top cover. The relevant parameters to model LFG collection and dispersive emissions are the time-dependent collection efficiency and top cover oxidation efficiency (Table 4-13). It is important to note that the LFG collection system does not operate over the 100 years time horizon due to technical constraints. For example, Starostina et al. (2014) assumed that LFG is no longer used for electricity generation after 30 years because the low flux of CH_4. Similarly, Sauve and Van Acker (2020) considered that LFG is not collected for the last 50 years. This assumption was also adopted in this work so that the LFG collection efficiency was set to 0% after year 50 (Table 4-13).

Table 4-13. Landfill gas collection efficiency and top cover oxidation efficiency of CH_4.

Period of time	LFG collection efficiency (1)	Top cover oxidation efficiency (2)
Years 1–2	25%	5%
Year 3	50%	10%
Year 4	70%	20%
Years 5-10	75%	20%
Years 11-50	95%	20%
Years 51-100	0%	20%

Source: [1] Barlaz et al. (2009); [2] US EPA (2011)

Leachate generation in the landfill over 100 years was computed from the annual precipitation and the infiltration rate (i.e., percentage of precipitation that becomes leachate). Leachate generation decreases over time as the waste landfilled in year zero is covered with more waste and the intermediate and final covers (US EPA, 2011). Reported data shows that the infiltration rate could achieve up to 50% of precipitation for the first five years and up to 30% for the remaining years (Obersteiner et al., 2007). Based on these figures and the annual precipitation in Madrid (421 L/m^2), the annual generation of leachate was estimated at 210-126 L/m^2 (Table 4-14). These estimations fall within the range of annual leachate generation in Europe, estimated at 50-400 L/m^2 (Camba et al., 2014). Note that these figures refer to the volume of leachate generated per surface area of landfill. To allocate leachate generation to the mass of waste landfilled, the previous values must be divided by the mass of waste per unit of landfill area, which was estimated at 23 tonnes/m^2[‡‡]. Thus, the annual generation of leachate ranges from 0.009 to 0.005 m^3/tonne waste, while the accumulated generation over 100 years is 0.56 m^3/tonne waste. For example, Damgaard et al. (2011) estimated the accumulated leachate generation over 100 years for a landfill in Denmark at 1.12 m^3/tonne waste.

Leachate composition (e.g., heavy metals content) largely depends on the composition of the waste. In this regard, the composition of the leachate for different waste materials reported by Manfredi et al. (2010) were adopted in this work.

[‡‡] Based on the landfill surface area (825,000 m^2), volumetric capacity (22.7 million m^3), and the average density of waste bulk in the landfill (838 kg/m^3).

The leachate is collected and treated to remove nutrients and heavy metals before being discharged to surface water bodies. The collection efficiency of leachate over 100 years was obtained from Damgaard et al. (2011) and ranges from 95% during the first 20 years to 80% for the years 21-40 and 60% for the years 41-50 (Table 4-14). Just as for the LFG collection system, it was assumed that the leachate collection system stops operating after year 50 (Sauve and van Acker, 2020). The treatment removes 4.1% of nitrogen, 58.6% of phosphorus, 48.2% of potassium, and 85% of heavy metals content of the leachate (Sauve and van Acker, 2020; US EPA, 2011). After treatment, a concentrated stream containing the nutrients and heavy metals is recirculated into the landfill, while a clean stream is discharged to surface water bodies. Furthermore, the uncollected leachate migrates through the bottom of the landfill contaminating ground water bodies.

Table 4-14. Parameters to model landfill leachate generation and collection over 100 years following waste disposal.

Period of time	**Annual precipitation [L/m^2]** A	**Infiltration rate [%]** B	**Annual leachate generation [L/m^2]** A*B	**Leachate collection [% leachate]**
Years 1–5	420.9	50	210	95
Year 6–20	420.9	30	126	95
Year 21–40	420.9	30	126	80
Years 41–50	420.9	30	126	60
Years 51–100	420.9	30	126	0

Finally, the daily operation of a landfill requires energy. Electricity demand was assumed to be satisfied internally with on-site power generation. Furthermore, 1.42 kg diesel/tonne waste is used for compactors and daily cover (Damgaard et al., 2011; Kirkeby et al., 2007; Manfredi et al., 2010, 2009; US EPA, 2011).

Recycling

The recyclable materials recovered by MRFs (including ferrous metal from incineration) are baled and sold to recyclers who process these into secondary materials that can substitute primary materials. The net environmental impacts of recycling are the difference between the impacts generated due to transportation and recycling activities and the impacts avoided due to the substitution of primary materials (Andreasi Bassi et al., 2017; Haupt et al., 2018a; Rigamonti et al., 2014). The amount of primary materials substituted depends on the amount of recyclable materials recovered, the efficiency of recycling, and the substitution ratio. The recycling efficiency represents the ratio between the mass of input material to recycling and the mass of secondary material obtained (Table 4-15). This efficiency reflects material losses that occur during recycling. The substitution ratio represents how much primary material is substituted by a secondary material considering quality losses during recycling.

Table 4-15 summarizes the data used to model recycling. The distance to recycling facilities is highly uncertain since recycling takes place in an international market. A conservative assumption of 400 km by truck was made for all the recyclable materials. Average recycling efficiencies and substitution ratios were retrieved from the literature, while the impacts of recycling activities and the production of primary materials were obtained from the Ecoinvent v3.7.1 database. Note that the substitution ratio of glass is superior to 100% (120%). This is because up to 20% of the raw materials used for primary glass production are lost as gases during melting. Therefore, 1 kg of secondary glass can avoid the production of 1.2 kg of primary glass (Larsen et al., 2009).

Table 4-15. Parameters to model recycling.

Recyclable material	Secondary material	Substituted primary material	Distance to recycling [km]	Recycling efficiency [%]	Substitution ratio [%]
Paper, sorted and baled	Paper, newsprint, recycled	Paper, newsprint, virgin	400	81.0 (9)	90.0 (14)
Cardboard, sorted and baled	Containerboard, linerboard, recycled	Containerboard, linerboard, virgin	400	81.0 (9)	90.0 (14)
PET, sorted and baled	PET granulate, amorphous, recycled	PET granulate, amorphous, virgin	400	82.2 (3,4,6,7,8,10,12)	92.8 (4,6,7,8,10,12)
HDPE, sorted and baled	HDPE granulate, amorphous, recycled	HDPE granulate, amorphous, virgin	400	87.5 (3,4,6,7,8,10,12)	81.7 (4,6,7,8,10,11,12)
LDPE, sorted and baled	LDPE granulate, amorphous, recycled	LDPE granulate, amorphous, virgin	400	73.2 (2,3,4,7,8,10)	77.8 (4,7,8,10)
PP, sorted and baled	PP granulate, amorphous, recycled	PP granulate, amorphous, virgin	400	84.1 (1,2,3,4,6,7,10)	81.2 (1,4,6,7,10)
Mix plastic, sorted and baled	Mix plastic granulate, amorphous, recycled	PET granulate, amorphous, virgin	400	69.6 (2,3,7,10,12)	70.0 (8)
Cartons & alike, sorted and baled	Containerboard, linerboard, recycled	Containerboard, linerboard, virgin	400	68.5 (8)	90.0 (15)
Glass, sorted and baled	Packaging glass, green, recycled	Packaging glass, green, virgin	400	99.1 (10)	120 (13)
Ferrous metal, sorted and baled	Steel, low-alloyed, recycled	Steel, low-alloyed, virgin	400	88.0 (8)	100 (8)
Non-ferrous metal, sorted and baled	Aluminium, cast alloy, recycled	Aluminium, cast alloy, virgin	400	94.0 (5)	100 (8)

Source: [1] Cardamone et al. (2021); [2] Civancik-Uslu et al. (2021); [3] Andreasi Bassi et al. (2020); [4] Meys et al. (2020); [5] European Aluminum (2020); [6] Faraca et al. (2019); [7] Van Eygen et al. (2018); [8] Haupt et al. (2018b); [9] Van Ewijk et al. (2018); [10] Turner et al. (2016); [11] Gala et al. (2015); [12] Rigamonti et al. (2014); [13] Larsen et al. (2009); [14] Merrild et al. (2008); [(15)] Same substitution ratio as cardboard

Application of compost on land

The compost obtained by composting or AD is spread on agricultural land in substitution of mineral fertilizers. The net environmental impacts of using compost are the difference between the impacts generated due to transportation and emissions from soil and the impacts avoided due to the substitution of mineral fertilizers.

An average transportation distance of 50 km by truck was assumed for both the compost and the substituted mineral fertilizers. The use on land of compost results in a slow release of biogenic CO_2 (due to organic matter mineralization), N_2O and NO_x (produced as intermediate products in the denitrification process and as by-products in the nitrification process), NH_3 (due to ammonia volatilization), and leaching of nutrients to surface and ground water (Brockmann et al., 2018; Bruun et al., 2006; Nemecek and Kagi, 2007). Furthermore, the heavy metals contained in the compost are transferred to the soil. Emissions to air, water, and soil occur over a long period of time following land application. Just as for the LFG, a common choice in the literature is to account for the emissions produced over 100 years following land application (Breunig et al., 2019; Sardarmehni et al., 2021; ten Hoeve et al., 2019; Yoshida et al., 2018). Therefore, emissions from soil were modelled with emission factors that represent the mass percentage of a chemical element applied to soil that is emitted to the air, water, or soil compartment over 100 years (Table 4-16). Note that the biogenic carbon that is not mineralized within 100 years is assumed to remain stored in soil.

Table 4-16. Emission factors to model the use on land of compost.

Compound	Compartment	Unit	Value	Source
Carbon dioxide, biogenic	Air	% applied biogenic C	87	Bruun et al. (2006)
Carbon dioxide, storage	Soil	% applied biogenic C	13	Bruun et al. (2006)
Ammonia	Air	% applied N	1.3	Bruun et al. (2006); Tonini et al. (2019)
Dinitrogen monoxide	Air	% applied N	1	IPCC (2006b)
Nitrogen oxides	Air	% of N_2O	21	Nemecek and Kagi (2007)
Nitrate	Surface water	% applied N	20	Bruun et al. (2006)
Nitrate	Ground water	% applied N	20	Bruun et al. (2006)
Phosphorus	Surface water	% applied P	3 [(a)]	Tonini et al. (2019); van Dijk et al. (2016)
Phosphorus	Ground water	% applied P	3 [(a)]	Tonini et al. (2019); van Dijk et al. (2016)
Heavy metals	Soil	% applied	100	–

Note: [a] Total phosphorus losses from crop production in the European Union were estimated at 6% (van Dijk et al., 2016), that is distributed equally between surface water (3%) and ground water (3%) based on Tonini et al. (2019)

The use of compost on agricultural land avoids the production and use of mineral fertilizers and the associated impacts. The nitrogen, phosphorus, and potassium content of the compost is uptake by plants, thus avoiding to some extent the use nitrogen (as N), phosphate (as P_2O_5), and potassium (as K_2O) mineral fertilizers. The impacts of producing these mineral fertilizers were calculated from the amount of fertilizers substituted and their life cycle impacts provided in the Ecoinvent v3.7.1 database. It is important to note the fertilizing value of one kg of nitrogen from compost is lower than the fertilizing value of one kg of nitrogen mineral fertilizer, since the nitrogen in compost is present in both mineral and organic forms (Brockmann et al., 2018; Hansen et al., 2006). In this context, the amount of nitrogen mineral fertilizer that could be substituted is calculated through the mineral fertilizer equivalence, which was estimated at 40% based on the average from (Boldrin et al., 2009; Hansen et al., 2006; Levis and Barlaz, 2011; Sardarmehni et al., 2021; van Haaren et al., 2010). The fertilizing value of phosphorus and potassium from compost was assumed the same as for mineral fertilizers, in line with current practice in the literature (Brockmann et al., 2018; Hansen et al., 2006).

Besides production, the substitution of mineral fertilizers also avoids the emissions that would be produced from spreading these on land. These emissions involve N_2O and NO_x, for which the same emission factors as for compost were assumed (Table 4-16), leaching of nitrate to surface and ground water, which were assumed equal to 10% and 4% of the nitrogen content of the mineral fertilizers (Yoshida et al., 2018), and transfer of heavy metals to soil. The heavy metals content of mineral fertilizers was obtained from (McBride and Spiers, 2001).

4.2.3 Waste management indicators

The performance of waste management indicators, such as recycling and landfill rates, are evaluated along with the environmental performance. The recycling rate is calculated as established in Directive 2018/851. Therefore, the recycling rate includes the mass of materials entering recycling plus the mass of organic waste collected separately that enters AD and/or composting (after subtracting the amount of pre-treatment rejects). The landfill rate is calculated following the rules established by *Article 5a* in the Directive 2018/850. Accordingly, the calculation of the landfill rate must include: (1) the mass of waste streams directed to landfilling without previous treatment, (2) the mass of intermediate streams, such as rejects and residual organic waste, directed to landfilling, and (3) the mass of stabilized material produced by the composting of the residual organic waste. The intermediates and residues produced during recycling and recovery operations (e.g., AD and incineration) are not included in this calculation.

4.2.4 Capacity restrictions

The assessment of real MSW management systems requires considering the available waste treatment capacities. This feature is particularly important when addressing prospective scenarios where the waste flows could change drastically (e.g., due to the improvement of separate collection) but no new treatment capacity is planned (such as in the case of Madrid). In this regard, logical restrictions were implemented into the model to ensure that the existing capacities were not exceeded. If the capacity of a process was exceeded, the surplus waste was diverted towards other processes in a sequential manner until reaching the landfill. Table 4-17 summarizes the waste treatment capacities available in Madrid and the diversion sequence of surplus waste.

Table 4-17. Waste treatment capacities available in Madrid and the assumed diversion sequence for surplus waste if the existing capacity is exceeded.

Waste treatment facility	**Capacity** [(a)] (tonnes/year)	**Surplus diversion sequence**
Existing MRF for residual waste	1,054,000	Incineration → Landfilling
Existing MRF for packaging waste	126,500	Incineration → Landfilling
Existing MRF for paper/cardboard	Unconstrained	–
Existing MRF for glass	Unconstrained	–
Existing AD of organic waste collected separately	161,000 [(b)]	AD for residual waste → Landfilling
Existing AD of residual organic waste	108,175 [(b)]	Composting → Landfilling
Existing composting of residual organic waste	331,290	Landfilling
Existing incineration	328,500	Landfilling
Existing sanitary landfill	3,134,300 [(c)]	–

Note: [a] The operative capacity was calculated assuming a maximum capacity utilization rate of 91.3%; [b] Capacity of digesters; [c] Disposal capacity based on a volumetric capacity of 22.7 million m^3, an average waste bulk density of 838 kg/m^3, and the amount of waste accumulated in the landfill since it was opened in 2000 (15.9 million tonnes). MRF: material recovery facility, AD: anaerobic digestion.

It should be noted that there are two AD facilities, one dedicated to the organic waste collected separately and another one dedicated to the residual organic waste. This is because these two streams should not be mixed to prevent cross-contamination of compost. As shown in Chapter 3, the residual organic waste contains a high share of non-biodegradable materials such as plastic, glass, metal, and others.

4.3 Environmental assessment of phasing-out incineration in Madrid

This section presents the application of the integrated MFA and LCA framework to evaluate the potential environmental consequences of phasing-out MSW incineration in Madrid. First, Section 4.3.1

defines the scenarios for analysis. Subsequently, Section 4.3.2 presents the MFA of the scenarios, while the life cycle inventory analysis and life cycle impact assessment are presented in Sections 4.3.3 and 4.3.4, respectively.

4.3.1 Scenarios definition

The reference scenario is defined by the existing MSW management system of Madrid in 2019. This scenario is characterized by a relatively low rate of separate collection, a moderate use of incineration, and a high rate of landfilling. The mass and composition of the MSW streams collected in Madrid in 2019 were presented in Chapter 3, while the partitioning of the waste streams among treatment processes is summarized in Table 4-18.

Table 4-18. Partitioning of waste streams among waste treatment processes in each scenario.

Waste stream	Process	Scenario			
		2019	**2025**	**2030**	**2040**
Waste streams collected separately (feedstocks)					
Household residual waste	Material recovery facility	97.4	100	100	100
	Landfill	2.6	0	0	0
Household packaging waste	Material recovery facility	100	100	100	100
	Landfill	0	0	0	0
Household paper/cardboard	Material recovery facility	100	100	100	100
Household glass	Material recovery facility	100	100	100	100
Household organic waste	Anaerobic digestion	100	100	100	100
Commercial residual waste	Material recovery facility	4.4	100	100	100
	Landfill	95.6	0	0	0
Commercial organic waste	Anaerobic digestion	100	100	100	100
Street cleaning waste	Landfill	100	100	100	100
Intermediate streams and residues					
Residual organic waste	Composting	30	43	43	43
	Anaerobic digestion	39	57	57	57
	Landfill	31	0	0	0
Rejects from material recovery facility	Incineration	73	0 [(a)]	0 [(a)]	0 [(a)]
	Landfill	27	100	100	100
Rejects from anaerobic digestion	Incineration	0	0	0	0
	Landfill	100	100	100	100
Digestate from anaerobic digestion	Composting	64	100	100	100
	Landfill	36	0	0	0

Note: [a] For the future scenarios with incineration, rejects are received at the incineration facility according to the available capacity, the rest being diverted to landfilling.

In the reference scenario, the packaging waste, paper/cardboard, and glass streams are delivered to MRFs for additional sorting before recycling. The organic waste collected separately is treated through AD. The household residual waste is primarily delivered to MRFs (97.4%), with only a negligible share being landfilled (2.6%). Conversely, the commercial residual waste and street cleaning waste are primarily disposed of in the landfill without previous treatment. The residual organic waste separated at MRFs is distributed between composting (30%), AD (39%), and landfilling (31%). The rejects from sorting packaging and residual waste at MRFs are 73% incinerated and 27% disposed of in the landfill. The rejects from AD pre-treatment are landfilled, while 64% of the digestate is stabilized through composting before landfilling (only a negligible amount of compost is applied on land). Finally, 34% of the compost produced by the composting of residual organic waste is sold as organic fertilizer, the rest being landfilled.

The reference scenario was compared against a range of plausible scenarios describing the system in the years 2025, 2030, and 2040 with and without incineration. In the scenarios without incineration, the rejects from MRFs are fully diverted to landfilling. Conversely, the future scenarios with incineration assume that rejects are directed to the existing or a new incineration facility according to the current capacity, the rest being diverted to landfilling. Other changes that apply to the future scenarios are:

- The rate of separate collection increases from 25% in 2019 up to 33% by 2025, 39% by 2030, and 46% by 2040, based on Chapter 3.
- No waste is landfilled without previous treatment. This means that 100% of the household and commercial residual waste is delivered to MRFs under the future scenarios. The street cleaning waste is diverted to composting due to its high share of green waste (34%).
- The residual organic waste is distributed between composting (43%) and AD (57%) with the aim of minimizing the landfilling of biodegradable waste in the future scenarios.
- All the digestate produced from the organic waste collected separately undergoes composting with the aim of producing a compost that can be applied on land.
- In line with the current legislation, a usable compost is only produced from the organic waste collected separately. The composting or AD of the residual organic waste has as unique output a stabilized material or digestate that must be landfilled.
- The environmental impacts of the Spanish electricity mix were modelled time-dependent. The generation mix has been projected up to 2040 based on the Spanish energy and climate integrate plan (Ministerio para la Transición Ecológica, 2020). The major changes are the elimination of coal by 2025, a decrease in the share of natural gas from 34% in 2019 to 4% by 2040, and an increase in the share of wind from 21% in 2019 to 43% by 2040 and in the share of photovoltaics from 4% in 2019 to 28% by 2040 (Table B-3 in Appendix B).

4.3.2 Material flow analysis

Table 4-19 summarizes the MFA of the reference scenario 2019 and the future scenarios with and without incineration. The total amount of waste disposed of in landfill in the reference scenario 2019 was estimated at 791,297 tonnes. This amount is disaggregated into 136,777 tonnes of street cleaning waste, 131,388 tonnes of MRFs rejects, 102,976 tonnes of residual organic waste, 96,271 tonnes of commercial residual waste, 85,227 tonnes of AD pre-treatment rejects, 74,337 tonnes of AD digestate, 69,595 tonnes of stabilized material from composting of residual organic waste, 21,153 tonnes of household residual waste, and 11,254 tonnes of incineration bottom ash. To calculate the landfill rate, the residues produced by recovery operations (i.e., AD digestate and incineration ashes) must be excluded (*cf.* Section 4.2.3). This results in 705,666 tonnes of waste disposed of in the landfill and a landfill rate of 51%. In this scenario, about 300,000 tonnes of MRFs rejects are incinerated (22% of MSW). Furthermore, about 186,917 tonnes of materials enter recycling and 92,576 tonnes of separate collected organic waste enter AD, resulting in a recycling rate of 20%.

Table 4-19. Material flow analysis (MFA) of the municipal solid waste management system of Madrid in 2019 (reference scenario), 2025, 2030, and 2040. The future scenarios are evaluated with and without incineration.

Waste treatment facility		**Without incineration**			**With incineration (existing or new)**		
	2019	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**
Waste flows (tonnes/year)							
MRF for residual waste	785,575	797,809	707,688	602,437	797,809	707,688	602,437
MRF for packaging waste	96,231	101,661	106,313	109,546	101,661	106,313	109,546
MRF for paper/cardboard	73,475	115,036	150,802	188,685	115,036	150,802	188,685
MRF for glass	58,809	70,609	80,755	93,612	70,609	80,755	93,612
AD of organic waste collected separately [(a)]	92,576	123,929	154,770	223,365	123,929	123,929	154,770
AD of residual organic waste [(a)]	76,156	98,790	94,480	60,624	98,790	94,480	60,624
Composting	96,520	288,911	267,474	276,977	288,911	267,474	276,977
Incineration	300,000	0	0	0	300,000	300,000	300,000
Landfill [(b)]	705,666	903,125	843,365	796,295	603,125	543,365	496,295
Recycling	186,917	242,635	282,640	327,333	243,989	284,110	329,041
Compost used on land	13,290	39,853	48,835	60,386	39,853	48,835	60,386
Waste management indicators (% MSW)							
Separate collection rate	25	33	39	46	33	39	46
Recycling rate [(c)]	20	26	31	39	26	31	40
Incineration rate	22	0	0	0	22	22	22
Landfill rate [(b)]	51	65	61	57	43	39	36

Notes: [a] Amount of organic waste after subtracting pre-treatment rejects; [b] Calculated as established in the Landfill Directive 2018/850 (see Section 4.2.3); [c] Calculated as established in the Directive 2018/851 (see Section 4.2.3). MRF: material recovery facility. AD: anaerobic digestion.

The MFA of the future scenarios suggest that the phasing-out of incineration would increase the landfill rate despite the improvement in separate collection and recycling and the enhancement of biological treatments (i.e., composting and AD). The landfill rate reaches 57% in the 2040 scenario without incineration, which corresponds to 796,295 tonnes of waste disposed of in landfill. This relatively high landfill rate is primarily attributed to the disposal of MRFs rejects, which totals 394,251 tonnes in this scenario. Furthermore, the landfilling of stabilized material from composting and rejects from AD pre-treatment increases up to 266,209 and 135,835 tonnes, respectively. The amount of materials that enters recycling reaches 327,333 tonnes, while the amount of separate collected organic waste treated through AD increases up to 223,365 tonnes. Therefore, the recycling rate reaches approximately 39% in the 2040 scenario.

All in all, the MFA results suggest that eliminating incineration while improving separate collection and recycling and enhancing biological treatments is not sufficient to reduce landfilling. A strong reduction of the landfill rate could be achieved if incineration continues in operation. The availability of an incineration facility would allow decreasing the landfill rate to 36% by 2040.

4.3.3 Life cycle inventory analysis

Table 4-20 summarizes the main inventory results for the studied scenarios. On a mass basis, CO_2 is by far the largest direct emission produced by the MSW management system. CO_2 emissions in the reference scenario 2019 were estimated at 620,046 tonnes, divided into 201,506 tonnes of fossil CO_2 and 418,540 tonnes of biogenic CO_2. Focusing on the year 2030, CO_2 emissions range from 279,523 tonnes in the scenario without incineration to 603,348 tonnes in the scenario with incineration (note that there is no difference in terms of direct CO_2 emissions between the existing and new incineration facilities). Thus, phasing-out incineration by 2030 could reduce CO_2 emissions a 116%. More significantly, fossil CO_2 emissions become zero in the scenarios without incineration since this is the only source of fossil CO_2 associated with MSW treatment (note that the collection and transportation stage was excluded in this study).

Table 4-20. Main inventory results for the municipal solid waste management scenarios for Madrid.

		Without incineration			**With existing incineration facility**			**With new incineration facility**		
	2019	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**
Direct emissions to air (kg/year)										
CO_2 fossil	201,505,538	–	–	–	226,669,214	243,073,395	268,702,460	226,669,214	243,073,395	268,702,460
CO_2 biogenic	418,540,479	303,010,793	279,523,018	248,519,290	387,179,591	360,274,871	322,709,267	387,179,591	360,274,871	322,709,267
CO_2 to soil >100 yr	1,268,747	1,891,727	2,306,240	2,834,504	1,891,727	2,306,240	2,834,504	1,891,727	2,306,240	2,834,504
CH_4 fossil	7,504	0	0	0	7,794	8,037	8,426	7,794	8,037	8,426
CH_4 biogenic	9,204,012	8,754,342	7,623,317	6,032,051	4,876,291	4,083,089	3,142,459	4,876,291	4,083,089	3,142,459
CO fossil	13,029	0	0	0	13,531	13,953	14,629	13,531	13,953	14,629
CO biogenic	644,154	744,322	675,813	579,601	534,344	486,413	430,201	534,344	486,413	430,201
NH_3	11,521	12,223	12,278	12,620	17,100	17,134	17,320	17,100	17,134	17,320
N_2O	24,348	26,244	25,425	24,606	33,793	32,994	31,763	33,793	32,994	31,763
NO_x	695,266	439,231	405,294	343,813	567,847	546,084	500,496	567,847	546,084	500,496
SO_2	33,890	31,697	28,717	23,769	23,218	21,237	18,030	23,218	21,237	18,030
NMVOC	97,519	17,918	15,907	12,796	90,622	89,288	87,489	90,622	89,288	87,489
PM <2.5 um	2,716	839	850	805	2,718	2,740	2,718	2,718	2,740	2,718
PM 2.5-10 um	672	676	600	483	412	362	294	412	362	294
Dioxin	2.41E-05	1.72E-06	1.53E-06	1.23E-06	2.35E-05	2.33E-05	2.32E-05	2.35E-05	2.33E-05	2.32E-05
Barium	207	–	–	–	213	224	239	213	224	239
Chromium	147	–	–	–	129	120	92	129	120	92
Lead	451	–	–	–	438	449	450	438	449	450
Zinc	391	–	–	–	339	317	240	339	317	240
Direct emissions to ground water (kg/year)										
Nitrogen	124,808	89,236	84,123	75,963	50,695	45,971	39,566	50,695	45,971	39,566
Nitrate	88,554	125,235	151,607	184,773	125,235	151,607	184,773	125,235	151,607	184,773
Phosphorus	1,850	1,258	1,199	1,105	727	668	587	727	668	587
Phosphate	557	466	423	375	255	219	178	255	219	178
Phenol	233	193	176	156	106	91	74	106	91	74
Zinc	243	259	249	233	136	122	102	136	122	102

Table 4-20. *(continued)*

		Without incineration			With existing incineration facility			With new incineration facility		
	2019	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**
Direct emissions to surface water (kg/year)										
Nitrogen	89,347	63,882	60,221	54,380	36,291	32,910	28,324	36,291	32,910	28,324
Nitrate	101,837	144,020	174,348	212,488	144,020	174,348	212,488	144,020	174,348	212,488
Phosphorus	572	389	370	342	225	206	181	225	206	181
Phosphate	215	180	164	145	99	85	69	99	85	69
Phenol	174	144	131	116	79	68	55	79	68	55
Zinc	27	29	28	26	15	14	11	15	14	11
Direct emissions to soil (kg/year)										
Barium	1,174	8,292	10,247	12,795	8,292	10,247	12,795	8,292	10,247	12,795
Copper	304	1,087	1,337	1,660	1,087	1,337	1,660	1,087	1,337	1,660
Chromium	140	871	1,071	1,330	871	1,071	1,330	871	1,071	1,330
Lead	188	1,476	1,827	2,284	1,476	1,827	2,284	1,476	1,827	2,284
Selenium	413	2,209	2,697	3,321	2,209	2,697	3,321	2,209	2,697	3,321
Silver	346	3,140	3,823	4,691	3,140	3,823	4,691	3,140	3,823	4,691
Zinc	1,128	5,767	7,101	8,831	5,767	7,101	8,831	5,767	7,101	8,831
Commodities and services (units/year)										
Electricity (kWh)	45,925,211	69,596,792	69,967,582	67,348,752	69,596,792	69,967,582	67,348,752	69,596,792	69,967,582	67,348,752
Diesel (kg)	2,346,797	3,074,484	3,016,865	2,900,693	2,647,955	2,590,336	2,474,164	2,647,955	2,590,336	2,474,164
Heat (MJ)	30,602,824	–	–	–	30,602,824	30,602,824	30,602,824	30,602,824	30,602,824	30,602,824
Water (m^3)	336,522	423,279	444,307	416,262	468,279	489,307	461,262	468,279	489,307	461,262
Chemicals (kg)	4,425,000	–	–	–	4,425,000	4,425,000	4,425,000	4,425,000	4,425,000	4,425,000
Transport (tkm)	75,431,265	99,046,581	115,497,575	133,952,646	99,588,406	116,085,937	134,635,795	100,140,376	116,692,360	135,345,794
Wastewater (m^3)	240,804	330,242	351,314	325,699	330,242	351,314	325,699	330,242	351,314	325,699
Bottom ash disposal (kg)	11,254,387	0	0	0	11,254,387	11,254,387	11,254,387	75,000,000	75,000,000	75,000,000
Fly ash disposal (kg)	22,128,064	0	0	0	22,128,064	22,128,064	22,128,064	19,500,000	19,500,000	19,500,000

Table 4-20. *(continued)*

		Without incineration			**With existing incineration facility**			**With new incineration facility**		
	2019	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**	**2025**	**2030**	**2040**
Materials and energy substitution (units/year)										
Primary materials (kg)	161,774,878	207,604,316	241,064,812	279,152,345	208,796,333	242,359,208	280,655,272	210,093,461	243,784,303	282,323,771
Mineral fertilizers (kg)	207,396	579,398	705,914	866,964	579,398	705,914	866,964	579,398	705,914	866,964
Electricity (kWh)	282,583,535	159,225,261	147,824,537	126,384,925	248,262,990	244,661,979	236,122,415	405,965,870	405,751,356	401,290,842
Biomethane (m^3)	4,876,813	6,314,307	6,857,818	6,961,974	6,314,307	6,857,818	6,961,974	6,314,307	6,857,818	6,961,974

The second largest emitted compound on a mass basis is CH_4, predominantely biogenic CH_4 from the decomposition of biodegradable materials in landfill. CH_4 emissions decrease from 9,211 tonnes in the reference scenario 2019 to 6,032–8,754 tonnes in the future scenarios without incineration. This reduction may appear rather surprising since the MFA showed that phasing-out incineration increases the amount of waste disposed of in landfill (Table 4-19). However, the reduction in CH_4 emissions is largely explained by the higher separate collection of organic waste, paper, and cardboard in the future scenarios, which prevent the disposal of highly biodegradable materials in the landfill. Nonetheless, the lowest CH_4 emissions are achieved in the future scenarios with incineration.

For most of the other relevant compounds released to air, the future scenarios without incineration exhibit a lower level of emissions compared with the reference scenario. For example, phasing-out incineration by 2030 could reduce the emissions of NH_3, N_2O, NO_x, and SO_2 between 23% and 35%, NMVOC an 82%, dioxins a 93%, and particulates (PM) between 66% and 69%. Furthermore, heavy metals emissions to air (e.g., Cr, Pb, and Zn) become zero since incineration is the only source of these elements.

Direct emissions to ground and surface water are indirectly affected by the phasing-out of incineration. The elimination of incineration and the subsequent intensification of landfilling could increase the production of landfill leachate and, in consequence, the emissions of nutrients (e.g., nitrogen, and phosphorus) and heavy metals (e.g., Zn) to water bodies. It should be noted that Table 4-20 does not include the emissions associated with the disposal of incineration ashes, which were modelled based on exogenous inventories obtained from the Ecoinvent database. These emissions and their environmental implications are discussed later in the life cycle impact assessment section. Finally, direct emissions to soil are exclusively associated with the application of compost on land and, therefore, they are not affected by the phasing-out of incineration.

The recycling of the materials recovered by the MSW management system avoids the production of 161,775 tonnes of primary materials in the reference scenario 2019 and between 241,064 and 243,784 tonnes by 2030. The application of compost on land allows avoiding the production and application of 207 tonnes of mineral fertilizers in the reference scenario and up to 706 tonnes in the 2030 scenarios. The amount of electricity delivered to the grid by the MSW management system in the reference scenario reaches 283 GWh. The phasing-out of incineration has a big impact on electricity production and substitution, which could drop to 148 GWh by 2030. The extended use of the existing incineration facility would allow maintaining a stable level of electricity production (between 248 and 236 GWh), while the investment in a new more efficient facility would boost electricity production up to 401-406

GWh. The biomethane injected into the grid (and the substituted volumen of natural gas) increases from 4.88 million m^3 in the reference scenario up to 6.31-6.96 million m^3 in the future scenarios due to the more intensive use of AD.

4.3.4 Life cycle impact assessment

The life cycle impact assessment of the studied scenarios and the potential environmental consequences of phasing-out incineration are presented below. Non-toxic (i.e., climate change, acidification, eutrophication, etc.) and toxicity (i.e., human health and ecotoxicity) impact categories are presented separately in Figure 4-2 and Figure 4-3, respectively, while the categories related to resources depletion are presented in Figure 4-4.

Climate change impacts

The net impact on climate change generated by the MSW management system of Madrid in the reference scenario 2019 was estimated at 391,151 tonnes CO_2 eq (Figure 4-2a). This corresponds to a carbon footprint of 343 kg CO_2 eq per tonne of MSW. Landfilling and incineration appear as the top contributors to the impact with 277,570 and 188,306 tonnes CO_2 eq, respectively. Recycling results in a net avoided impact of -86,340 tonnes CO_2 eq, while the other processes (i.e., MRFs, composting, AD, and the use of compost) have negligible contributions.

The future scenarios that consider the extended use of the existing incineration facility show that the climate change impacts could decrease to 282,848 tonnes CO_2 eq by 2025, 264,024 tonnes CO_2 eq by 2030, and 256,942 tonnes CO_2 eq by 2040. This decrease is primarily driven by an increase in recycling. The future scenarios without incineration suggest that the phasing-out of incineration could decrease impacts to 183,839 tonnes CO_2 eq by 2025, 137,618 tonnes CO_2 eq by 2030, and 78,456 tonnes CO_2 eq by 2040. When comparing these two sets of scenarios, it is observed that the phasing-out of the existing incineration facility could reduce the climate change impacts a 35% by 2025, 48% by 2030, and 69% by 2040.

The impact in the future scenarios that consider the substitution of the existing incineration facility with a new (more efficient) facility ranges from 237,041 tonnes CO_2 eq by 2025 to 216,434 and 213,749 tonnes CO_2 eq by 2030 and 2040, respectively. This means a reduction between 17% and 18% with respect to the extended use of the existing facility. Therefore, the investment in a new incineration facility would bring reduced climate benefits compared with the elimination of incineration.

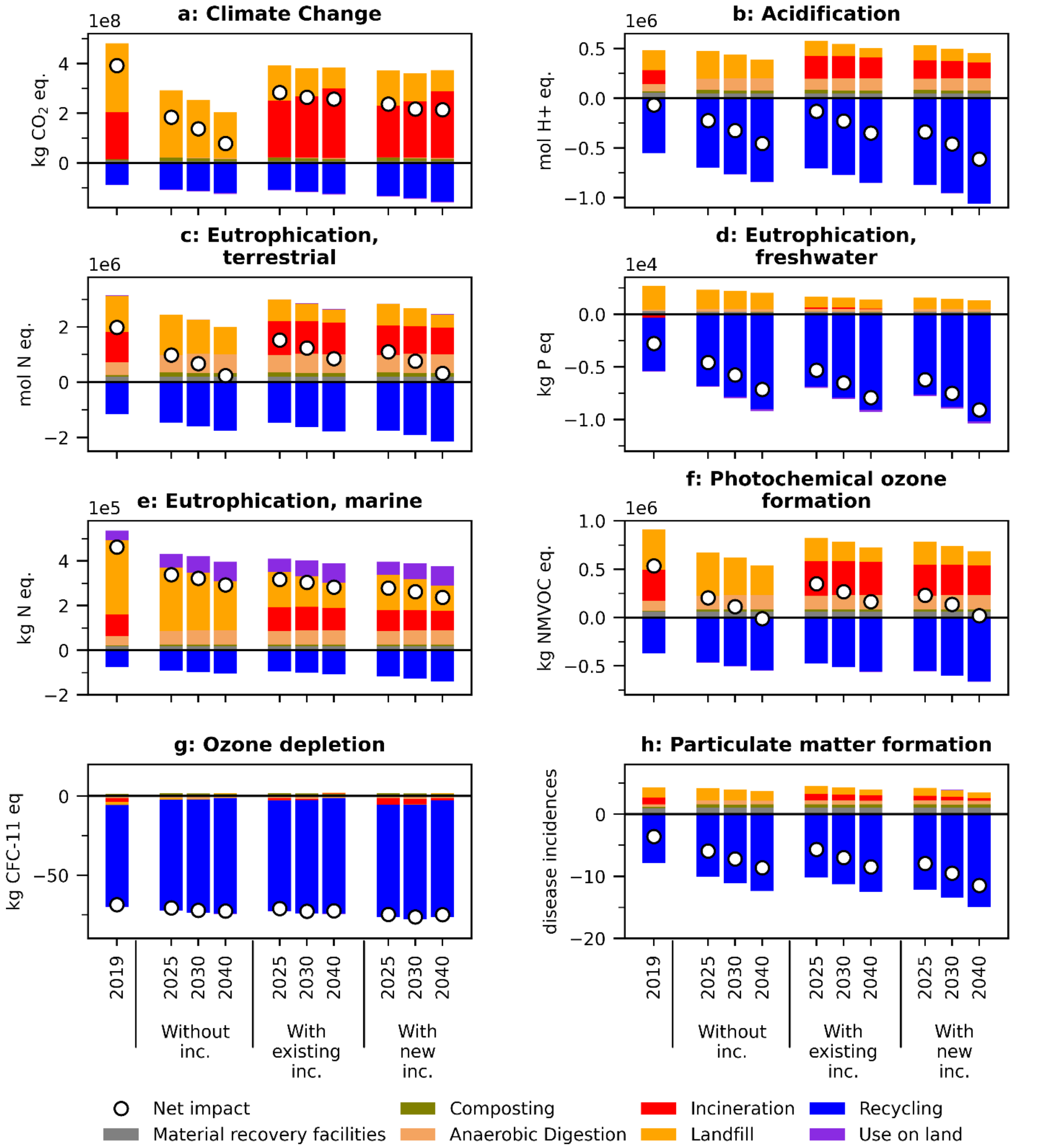

Figure 4-2. Life cycle impact assessment (non-toxic categories) of the municipal solid waste (MSW) management system of Madrid in 2019 (reference scenario), 2025, 2030, and 2040. The future scenarios are evaluated with and without incineration (labeld as “inc.” in the figure). Positive values represent impacts generated by the system, while negative values represent avoided impacts. The functional unit is the treatment of the amount of MSW collected in the studied year.

It is worth noting that incineration exhibits a net positive impact on climate change across all the scenarios. This means that the emissions avoided due to the substitution of the electricity mix do not counterbalance the emissions produced from burning fossil-based waste materials, such as plastic. This trend is partially explained by the reduced carbon footprint of the Spanish electricity mix due to the high share of renewables. Since the major difference between the existing and a new incineration facility is that the new one exhibits a higher electrical efficiency, the reduced impact of the electricity mix explains the limited climate benefits of a new incineration facility for Madrid.

Furthermore, the carbon footprint of incineration increases over time, from 628 kg CO_2 eq/tonne waste in the reference scenario 2019 to 930 kg CO_2 eq/tonne waste in the 2040 scenario with the existing facility and 892 kg CO_2 eq/tonne waste in the 2040 scenario with the new facility. This increasing trend can be explained by two factors: (1) the changing composition of the waste fed to incineration and (2) the transition of the Spanish electricity mix towards a higher share of renewables. While the carbon footprint of incineration increases over time, that of landfilling decreases from 436 kg CO_2 eq/tonne waste in the reference scenario 2019 to 374 kg CO_2 eq/tonne waste by 2040. This reduction is due to the higher separate collection of organic waste, paper, and cardboard, which reduces the CH_4 emissions from landfills as explained in the LCI analysis. All in all, these trends explain why the future scenarios without incineration exhibit a lower carbon footprint despite an increased use of landfilling.

<u>Other non-toxic impacts</u>

Other non-toxic impact categories, such as acidification, terrestrial and freshwater eutrophication, photochemical ozone formation, ozone depletion, and particulate matter formation are largely dominated by the impacts avoided through recycling. Thus, the net impact decreases in the future scenarios due to the increase of recycling. Incineration and landfilling play a minor role compared with recycling. Notably, incineration exhibits a net positive impact on acidification (Figure 4-2b), primarily due to the emission of NO_x and, to a lesser extent, NH_3 and SO_2. In this context, the phasing-out of the existing incineration facility could reduce the impact on acidification between 42% in 2025 and 23% in 2040. Similar reductions are observed for terrestrial eutrophication and photochemical ozone formation, whereas freshwater and marine eutrophication, ozone depletion, and particulate matter formation are only marginally affected.

The substitution of the existing incineration facility with a new one shows a slightly different behaviour, especially in terms of acidification. Focusing on the year 2030, a new incineration facility reduces the acidification impact a 41% compared with the scenario without incineration and up to 101% compared with the scenario with the existing incinerator. This reduction is largely attributable to the fact that the new incineration facility recovers aluminium from the bottom ash. The recovered aluminium can substitute primary aluminium, thus avoiding the relatively high acidification impact caused by the high amount of energy consumed by primary production. In terms of the other non-toxic categories, the new incineration facility outperforms the existing facility, but do not achieve a net reduction compared with the scenarios without incineration. Therefore, the phasing-out of incineration scenario remains as the best pathway in terms of impact reduction.

Human toxicity and ecotoxicity impacts

The impact on human toxicity–cancer effects generated by the MSW management system is almost exclusively associated with incineration and recycling (Figure 4-3a). Furthermore, incineration is the main contributor to ecotoxicity (Figure 4-3c). The high contribution of incineration to these categories is due to the disposal of fly ashes (i.e., APC residues), which cause the migration of heavy metals towards ground and surface water bodies.

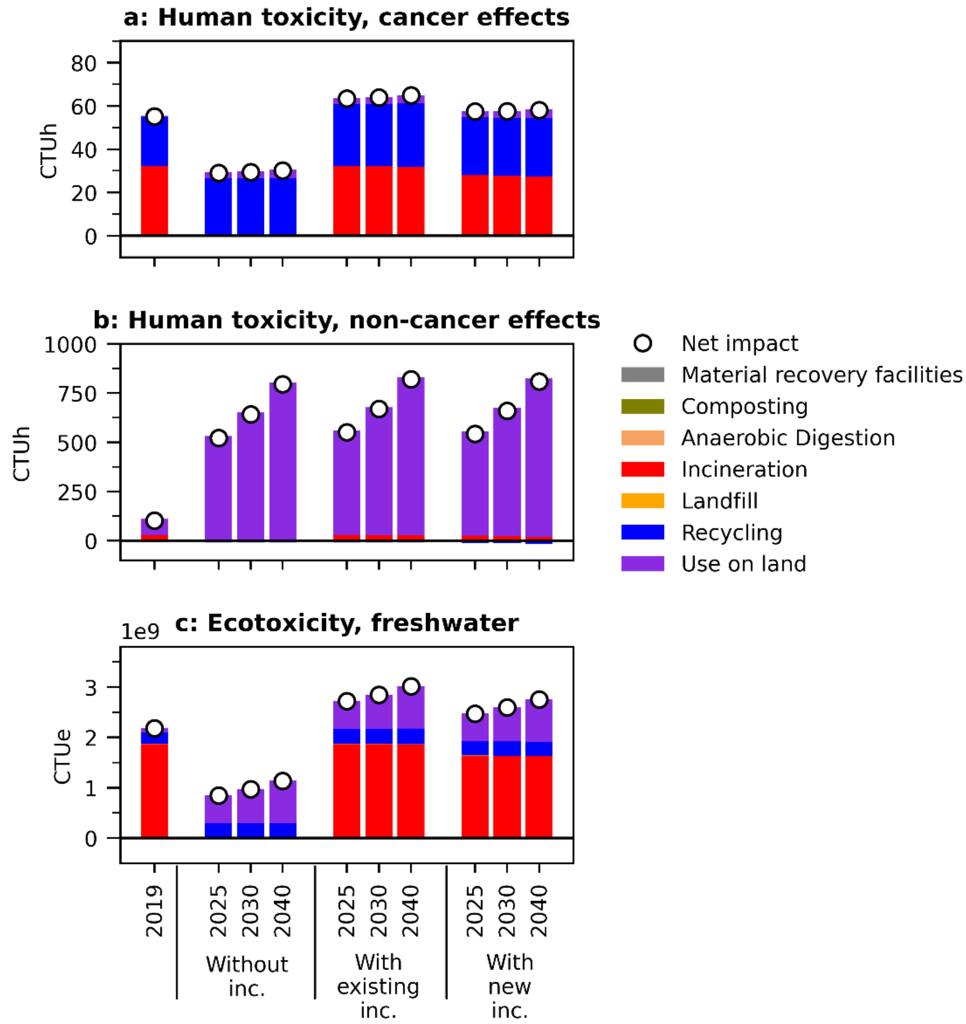

Figure 4-3. Life cycle impact assessment (<u>human toxicity and ecotoxicity categories</u>) of the municipal solid waste (MSW) management system of Madrid in 2019 (reference scenario), 2025, 2030, and 2040. The future scenarios are evaluated with and without incineration (labeld as "inc." in the figure). Positive values represent impacts generated by the system, while negative values represent avoided impacts. The functional unit is the treatment of the amount of MSW collected in the studied year.

The impact on human toxicity–cancer effects is associated (99% of the impact) with the emission of hexavalent chromium, Cr VI, to surface water bodies, while the impact on ecotoxicity is primarily due to the emission of antimony (Sb) to surface water (82%) and, to a lesser extent, the emission of Cr VI (17%). The major sources of Cr and Sb in fly ashes are inert materials such as glass and metals that do not combust during incineration (Götze et al., 2016b; Viczek et al., 2020). Cr is used widely in the metal industry in alloys and surface coating, while antimony compounds are used in the glass industry as opacifiers or refining agents (Viczek et al., 2020). Furthermore, Sb is extensively used in plastics as a flame retardant. However, only a small fraction (6%) of the Sb content of combustible materials is transferred into the ashes during incineration (Table 4-7).

All in all, the phasing-out of incineration could reduce the impact 1.2 times for human toxicity–cancer effects and 1.7-2.2 times for ecotoxicity. Incineration has a negligible contribution to the impact on human toxicity–non-cancer effects, which is dominated by the compost due to the addition of heavy metals to soil (Figure 4-3c).

Resources depletion

The depletion of resources is negative across all the scenarios, meaning that the MSW management system allows avoiding the depletion of non-renewable energy (Figure 4-4a) and minerals and metals (Figure 4-4b). The avoided impact comes primarily from the substitution of primary materials through recycling. Incineration could have notable contribution to prevent the consumption of fossil energy sources. However, this potential benefit is seriously compromised by the transition towards a more renewable electricity mix, approaching zero by 2040. For example, the phasing-out of the existing incineration facility in 2025 increases the depletion of non-renewable energy a 15%, while the phasing-out in 2040 increases the impact only 0.3%. Therefore, the relevance of incineration to the depletion of non-renewable energy is progressively reduced as the share of fossil fuels in the electricity mix becomes marginal.

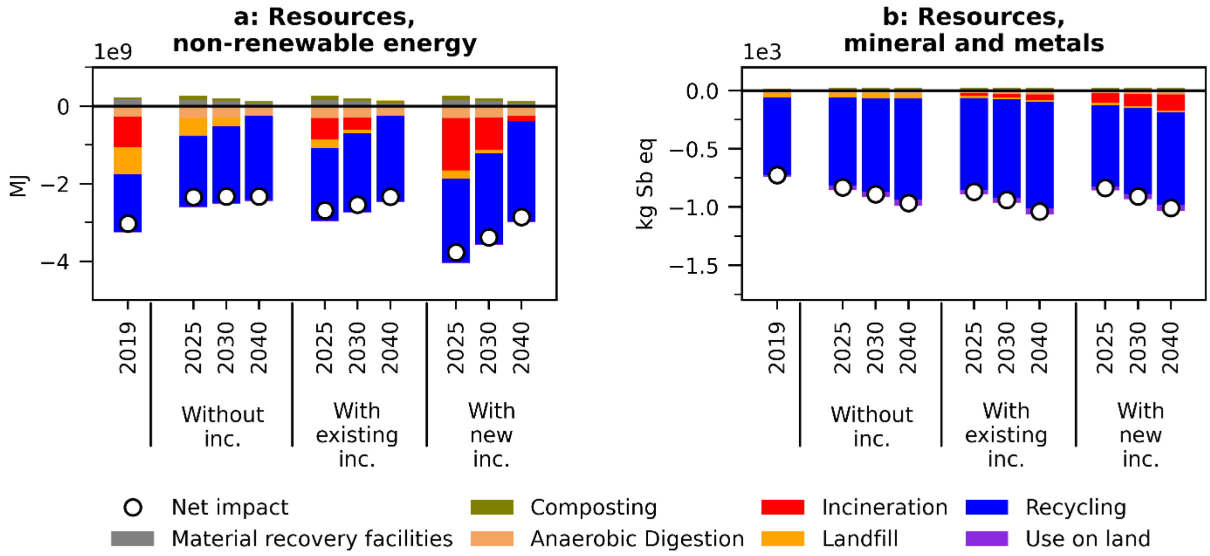

Figure 4-4. Life cycle impact assessment (resources depletion categories) of the municipal solid waste (MSW) management system of Madrid in 2019 (reference scenario), 2025, 2030, and 2040. The future scenarios are evaluated with and without incineration (labeld as "inc." in the figure). Positive values represent impacts generated by the system, while negative values represent avoided impacts. The functional unit is the treatment of the amount of MSW collected in the studied year.

4.4 Final remarks and future work

Phasing-out the existing incineration facility of Madrid represents an opportunity to reduce the impacts on climate change, acidification, terrestrial eutrophication, freshwater eutrophication, photochemical ozone formation, human toxicity–cancer effects, and ecotoxicity. However, the elimination of incineration could increase the landfill rate up to 65% by 2025, 61% by 2030, and 57% by 2040, which is far from the 10% landfill goal by 2035 established in the Landfill Directive. Based on these results, it is concluded that incineration phase-out could reduce substantially some of the environmental impacts associated with MSW management, but it will jeopardize the realization of ambitious landfill targets (**RQ2**).

Three impact categories deserve special attention, namely climate change, human toxicity–cancer effects, and ecotoxicity. The analysis performed in this work suggests potential climate benefits associated with incineration phase-out despite that this decision will intensify the use of landfilling. This conclusion may seem counter-intuitive for some readers since landfills are usually ranked as the last option from a climate perspective (e.g., in the waste hierarchy). However, it is largely explained by the dynamics of the MSW management system and the background conditions. On the one hand, the climate change impacts of incineration are likely to increase in the future driven by changes in waste composition (higher share of plastic fed to incineration) and the reduced climate benefits from substituting an electricity mix increasingly dominated by renewable energies (these two issues are further addressed in Chapter 5). On the other hand, enhancing the separate collection of highly biodegradable materials, such as food waste, paper, and cardboard, reduces the carbon footprint of landfilling due to the reduction in the CH_4 potential of the residual waste and rejects. These changes in the MSW management system and the electricity market seriously compromise the climate performance of incineration relative to landfilling. In this context, the substitution of the existing incineration facility with a new more efficient facility was proved to not introduce notable differences.

The human toxicity–cancer effects and ecotoxicity categories should be interpreted with caution due to the large uncertainties. The LCA case studies reviewed in Chapter 1 identified the emission of dioxins, furans, and heavy metals as the source of toxicity in incineration, whereas this work points to the disposal of fly ashes. It is worth noting that the impacts associated with the disposal of incineration ashes were obtained from the Ecoinvent v3.7.1 database due to the lack of consistent data to explicitly model these impacts. Consequently, the results are directly influenced by the modelling approach and assumptions adopted in this database. The major assumptions are leachate concentrations and the accounting of long-term leachate generation and emissions (Doka, 2009). While the high toxicity potential of fly ashes disposal has received attention in other works (Beylot et al., 2018; Lausselet et al., 2016), the uncertainties surrounding this issue remain large.

Although this work focused on the case study of Madrid, the findings are of potential interest to other European municipalities. However, the extrapolability of the outcomes to other contexts deserves attention. The waste treatment processes models developed in this work tries to represent as much as possible the situation of Madrid. Primary data for the period 2015-2019 was combined with data from recent literature covering the European context. Thus, this study addressed the phasing-out of an existing incineration facility (or its substitution with a new facility) in a modern integrated MSW management system. For example, the emission levels calculated by the model for the existing incineration facility of Madrid are well below the threshold levels of a BAT incinerator (Neuwahl et al., 2019) (Table 4-21).

Table 4-21. Incineration emission levels calculated by the model compared with treshold levels established in the best available techniques (BAT) document (kg compound/tonne waste)

Substance	This work	BAT threshold
Carbon monoxide	0.096	0.478
Ammonia	0.017	0.096
Nitrogen oxides	0.877 – 0.937	1.44
Sulphur dioxide	0.009 – 0.011	0.384
Hydrochloric acid	0.031	0.077
Hydrofluoric acid	0.002	0.010
Dioxin, 2,3,7,8 tetrachlorodibenzo	7.48E-11	5.75E-10
Cadmium	7.52E-06 – 8.44E-06	1.92E-04
Mercury	5,78E-06 – 6.94E-06	9.59E-02

Nevertheless, there are aspects that may differ between the case study of Madrid and other contexts and that could affect the results. First and foremost, this work assessed the specific MSW management system of Madrid, which can differ significantly from other systems. For example, it has been already highlighted in Chapter 3 the importance of the collection scheme and how this varies across regions and countries. Secondly, the incineration facility modelled herein produces only electricity. This is the general situation in Spain due to the lack of district heating networks. The combined heat and power (CHP) generation, as performed in Northern Europe, would reduce the environmental impacts of incineration due to the substitution of conventional heat sources, such as natural gas (Eriksson and Finnveden, 2017). Thirdly, this work considered a sanitary landfill equipped with a highly efficient LFG collection system. This largely contributes to the relatively reduced climate change impacts of landfilling. While LFG collection is a standard practice in Europe (Sauve and van Acker, 2020), the collection efficiency may deviate significantly from the values used in this work.

Other environmental challenges associated with landfilling, such as the loss of plastics and the subsequent pollution of rivers and oceans with micro-plastics, have not been investigated in this work since the underlying impact assessment methods are still not well developed (Maga et al., 2021; Woods et al., 2021; Yadav et al., 2020). However, landfills have been identified as an important source of marine plastic pollution (UNEP, 2018). In this context, the objective of reducing the landfill rate to 10% by 2035 should not be disregarded. Achieving 10% landfill by 2035 may require structural changes in the MSW management system beyond increasing separate collection or the extended use of incineration, which were proved insufficient. This issue and its implications for the deployment and performance of WTE capacity is addressed in Chapter 5.

Chapter 5

Developing a framework for the optimal planning of waste-to-energy pathways

This chapter presents a multi-period optimization framework for the systematic assessment and prioritization of WTE pathways in the context of an increasing circular MSW management system (**RQ3**)[§§]. The following Section 5.1 introduces the chapter. The multi-period optimization framework developed in this thesis is presented in Section 5.2, while its capabilities are demonstrated through its application to the case study of Madrid in Section 5.3. Finally, Section 5.4 contextualizes the results and outlines future research possibilities.

5.1 Introduction

The previous chapters provide some insights on the role that WTE might play in the context of an increasing circular economy. It was found that the separate collection and recycling targets do not necessarily compromise the energy recovery potential (Chapter 3) and that phasing-out incineration could reduce the impacts on climate change and human health at the expense of jeopardizing the realization of landfill targets (Chapter 4). Furthermore, the future scenarios assessed in the different chapters highlighted that the context in which the WTE technologies will operate can change substantially. With this background in mind, efforts should be focused on (1) identifying the long-term optimal WTE pathways from an economic and environmental perspective and (2) assessing the prospective economic and environmental performance of these pathways.

Addressing these objectives requires a tool capable of simultaneously assessing several hundreds of possible waste treatment pathways to identify the best solution. Such a modelling feature is offered by mathematical optimization, which has been introduced in Chapter 1. When focusing on the development of a comprehensive optimization framework for MSW management planning, three modelling gaps have been identified in the existing literature (*cf.* Section 1.4.3): (1) the extended use and/or retrofitting of existing facilities is generally neglected, (2) economies of scale are not considered, and (3) the non-linear response of waste treatment processes to waste composition is not captured. These modelling gaps have large implications for the assessment and prioritization of WTE pathways, as discussed in Section 1.4.3. Consequently, the optimization framework developed in this thesis aims at covering these three gaps.

The third modelling gap deserves special attention. The importance of waste composition has been continuously emphasised throughout this thesis. The MFA framework described in Chapter 3 and the

[§§] Based on: Istrate, I.R., E., Galvez-Martos, J.L., Vázquez, D., Guillén-Gosálbez, G., Dufour, J. Optimal planning, economics, and carbon footprint of waste-to-energy incineration considering the impact of separate collection and recycling (*Submitted to Renewable & Sustainable Energy Reviews*) life cycle inventories (LCIs) from Chapter 4 were thoroughly formulated to capture the response of waste treatment processes to changes in waste composition. This modelling feature, however, has been largely discarded in the existing waste optimization models. As highlighted in Section 1.4.3, the use of fix yields (e.g., kWh electricity generated per tonne waste) and emission factors (e.g., kg CO_2 per tonne waste) is the common practice in the literature (Chang et al., 2012; Ooi et al., 2021; Yousefloo and Babazadeh, 2020). This trend was primarily motivated by the fact that the introduction of waste composition as a model variable results in a nonlinear nonconvex model (as will be shown in Section 5.2). A nonconvex model exhibits multiple local optimum solutions, which substantially increase the difficulty of the problem (see also Section 1.4.3 in Chapter 1).

Commercial global optimization solvers can be used to solve such nonconvex problems, but they may require a prohibitive computational effort (Misener and Floudas, 2009). Alternatively, local optimization solvers can be used to find a feasible solution in a reduced period of time, but this local solution may be far from the global optima. Within this context, Levis et al. (2013) proposed a modelling framework that overcomes the issue of nonconvexities by forcing the model to choose at most one treatment pathway for each waste stream. Their approach, however, oversimplifies the problem by discarding possible technology combinations. Furthermore, Roberts et al. (2018) used a genetic algorithm to solve a nonlinear nonconvex waste treatment optimization problem, yet such metaheuristics approaches do not provide information on the quality of the final solution.

Other heuristic procedures have been proposed to produce good solutions for nonline nonconvex problems in a reduced period of time. Previous research has shown that the solution generated by a local optimization solver improves significantly if a good initial guess is provided (Castro et al., 2007; Deng et al., 2021; Galan and Grossmann, 1998). However, the determination of an initial guess is not a trivial task since the model might involve several hundreds of possible solutions. Thus, a more systematic approach is required. In this regard, a heuristic algorithm typically consists of the successive solution of a relaxed (linear) version of the original nonconvex model and the original model itself (Galan and Grossmann, 1998). In summary, the solution of the relaxed model (first part) provides the good initial guess for the solution of the original nonconvex model (second part). Heuristic procedures have been extensively applied to the optimal design of wastewater treatment networks (Castro et al., 2007; Teles et al., 2008) but they have never been applied to MSW treatment networks.

This chapter presents a multi-period optimization framework that determines time-dependent waste treatment capacities and flows according to economic or climate objectives and subject to waste availability and composition, system constraints and restriction, policy targets, and background conditions (e.g., the decarbonization of the electricity mix). The framework was built on the top of a thorough MSW treatment network that includes existing and new waste treatment facilities as well as the option of decommissioning or retrofitting the existing ones. Furthermore, the framework considers economies of scale and addresses the nonlinear response of waste treatment processes to changes in waste composition. An effective heuristic procedure was implemented to solve the resulting nonconvex nonlinear model. The capabilities of this framework are demonstrated through the assessment of the optimal WTE pathways (AD and incineration) for Madrid over 2020-2040.

5.2 Multi-period waste treatment optimization framework

This section describes the multi-period waste treatment optimization framework. First, Section 5.2.1 provides a formal description of the MSW treatment network. Subsequently, Section 5.2.2 defines the goal and scope of the framework, while Section 5.2.3 provides the mathematical formulation. Finally, Section 5.2.4 describes the solution procedure.

5.2.1 Municipal solid waste treatment network

The MSW treatment network studied throughout this thesis has been already introduced in Chapter 4 (Figure 4-1). However, a formal description within the set theory is necessary to be consistent with the mathematical formulation provided below. Therefore, the MSW treatment network consists of the following sets: (1) the set of waste streams collected from generation sources (hereinafter referred to as the set of <u>feedstocks</u>), (2) the set of <u>waste materials</u> contained in the MSW streams, (3) the set of <u>waste treatment facilities</u>, (4) the sets of <u>intermediates</u>, <u>residues</u>, <u>products</u>, and <u>emissions</u> released by the facilities, (5) the set of <u>commodities and services</u> consumed by facilities, and (6) the set of <u>counterfactual market products</u>.

The set of feedstocks includes the packaging, paper/cardboard, glass, organic, and residual waste collected from households, the organic and residual waste collected from commercial activities, and the street cleaning waste. The set of waste materials covers the 18 materials included in the MFA framework (*cf.* Table 3-1). The set of waste treatment facilities was divided into the subset of existing facilities, the subset of retrofitted facilities, and the subset of new facilities. The existing facilities represent the situation of the MSW treatment network before the initial time period of optimization. Retrofitted facilities are existing facilities that will undergo technology modifications[***]. For example, MRFs can be retrofitted to replace the existing sorting equipment with a new equipment, or an existing

[***] In this work, *retrofitting* implies technology modification but not capacity expansion.

AD facility can be retrofitted by replacing biomethane upgrading with a combined heat and power (CHP) unit.

It is worth noting that not every feedstock, intermediate, residue, and product is connected to every facility in the network. Some facilities are designed to process specific waste streams. For example, AD works for organic waste, but not for rejects or RDF. Moreover, not all the facilities produce every product. AD, for example, can produce biomethane or electricity, but composting does not. To improve the computational performance, the non-viable connections in the network were eliminated (Restrepo-Flórez and Maravelias, 2021). This was achieved through the construction of multidimensional sets that cover all the viable connections in the network. For example, the set FJ covers what feestocks are accepted by each facility, while the set FI covers what intermediates are produced by what facilities. Finally, Table 5-1 summarizes the sets and subsets used.

Table 5-1. Sets and subsets used in the optimization framework.

Set	
TP	$\{t$: Time periods of five years$\}$
J	$\{j$: Feedstocks$\}$
K	$\{k$: Waste materials$\}$
U	$\{u$: Facilities outputs, including intermediates, residues, and products$\}$
$I \subset U$	$\{i$: Intermediates$\}$
$R \subset U$	$\{r$: Residues$\}$
$P \subset U$	$\{p$: Products$\}$
$P^{REC} \subset P$	$\{p$: Recyclable materials$\}$
$P^{FERT} \subset P$	$\{p$: Compost$\}$
$P^{ENERGY} \subset P$	$\{p$: Energy products$\}$
S	$\{s$: Substituted market products$\}$
$S^{PRIM} \subset \mathrm{S}$	$\{s$: Substituted primary materials$\}$
$S^{FERT} \subset \mathrm{S}$	$\{s$: Substituted mineral fertilizers$\}$
$S^{ENERGY} \subset \mathrm{S}$	$\{s$: Substituted energy$\}$
E	$\{e$: Emissions$\}$
C	$\{c$: Commodities and services$\}$
F	$\{f$: Waste treatment facilities$\}$
$F^{EXIST} \subset F$	$\{f$: Existing waste treatment facilities$\}$
$F^{RETROFIT} \subset F$	$\{f$: Retrofitted waste treatment facilities$\}$
$F^{NEW} \subset F$	$\{f$: New waste treatment facilities$\}$
$F^{MRF} \subset F$	$\{f$: Material recovery facilities$\}$
$F^{COMP} \subset F$	$\{f$: Composting facilities$\}$
$F^{AD} \subset F$	$\{f$: Anaerobic digestion facilities$\}$
$F^{INC} \subset F$	$\{f$: Incineration facilities$\}$
$F^{LF} \subset F$	$\{f$: Landfill facilities$\}$
FI	$\{(f, i)$: Intermediate i is an output of facility $f\}$
FR	$\{(f, r)$: Residue r is an output of facility $f\}$
FP	$\{(f, p)$: Product p is an output of facility $f\}$
JF	$\{(j, f)$: Feedstock j is connected to facility $f\}$
FIF	$\{(f', i, f)$: Intermediate i produced by facility f' is connected to facility $f\}$
FRF	$\{(f', r, f)$: Residue r produced by facility f' is connected to facility $f\}$

5.2.2 Goal and scope

The optimization framework aims at determining the design of the MSW treatment network that maximizes the net present value (NPV) or minimizes the climate change impacts while considering the availability and composition of feedstocks, mass and energy balances, capacity constraints, and policy targets. To achieve this goal, the optimization model takes the following decisions:

- Whether and when to use, retrofit, or decommission existing waste treatment facilities.
- Whether and when to invest in new waste treatment facilities and with what capacities.
- Flow of feedstocks, intermediates, and residues to and between waste treatment facilities.

These decisions are made in each period over the decision horizon. By default, the studied decision horizon covers from 2020 to 2040 divided in 5-year time intervals[†††]. The 5-year time period is a common assumption in energy systems modelling and integrated assessment models (DeCarolis et al., 2017; Lopion et al., 2018). In addition to the decision horizon, the optimization model requires the following information (i.e., model parameters or the knowns of the problem):

- Mass and composition of the feedstocks collected in each period.
- Techno-economic and technology parameters of waste treatment facilities.
- Economic and environmental information on the background processes.

The NPV of the MSW treatment network was assessed through the net cash flow, which includes four types of costs: capital costs, fixed and variable operating costs, and revenues (Martinez-Sanchez et al., 2015; Taelman et al., 2020). The climate change impacts were assessed from a life cycle perspective, following the approach described in Chapter 4. The functional unit was reformulated to capture the long-term scope of the optimization framework. Thus, it was defined as “treatment of the MSW collected in the studied region over 2020-2040”. The system boundaries of the LCA were chosen those established in Chapter 4, the only exception being the inclusion of the life cycle impacts of the infrastructure required for the new and retrofitted facilities. Furthermore, multifunctionality issues, LCI modelling, and the impact assessment follows the approach described in Chapter 4.

5.2.3 Mathematical formulation

This section describes the algebraic formulation of the multi-period waste treatment optimization framework. The model was formulated as a mixed-integer non-linear program (MINLP), where continuous variables denote mass and energy flows, treatment capacities, costs, emissions, and climate change impacts, while binary variables represent capacity use and expansion decisions. The framework

[†††] Note that the model accounts for the accumulated mass of feedstocks collected over each 5-year period.

was structured into seven blocks of equations: (1) objective function(s), (2) network mass balance, (3) capacity adequacy, (4) waste treatment, (5) economic assessment, (6) climate change impacts assessment, and (7) policy targets. The following paragraphs describe each block. Uppercase letters are used to represent model variables, while lowercase letters and Greek characters are used to represent parameters.

Objective functions

The optimization framework aims at maximizing the net present value (NPV) or minimizing the climate change impacts of the MSW treatment network over the decision horizon.

Economic objective function

The economic objective function (Eq. 5-1) quantifies the NPV from the summation of the discounted net cash flow over all time periods. As defined in the goal and scope, the net cash flow in each period t was calculated as the difference between the revenues and the capital and operating costs associated with the waste treatment facilities.

$$\max OBJ^{NPV} = \sum_{t \in TP} \frac{\sum_{f \in F} REV_{f,t} - COST_{f,t}^{ACC} - COST_{f,t}^{FO} - COST_{f,t}^{VO}}{(1+d)^{t \cdot l - 2}} \quad \textit{Eq. 5-1}$$

.

where $REV_{f,t}$ denotes the revenues generated by facility f in period t [EUR/period], $COST_{f,t}^{ACC}$ is the annualized capital cost related to facility f incurred in period t [EUR/period], $COST_{f,t}^{FO}$ and $COST_{f,t}^{VO}$ are the fixed and variable operating costs of facility f in period t [EUR/period], d is the discount rate to calculate the present value of future costs [%], and l is the length of the time periods [years/period]. Note that the cash flow was centered in the middle year of each 5-year time period (i.e., the third year of the period). The middle year was calculated as the period multiplied by the length of the period (5 years) and subtracting 2 years to reach the middle.

Climate change objective function

The climate objective function (Eq. 5-2) quantifies the life cycle climate change impacts as the difference between the generated impacts (i.e., the impacts of direct emissions, new infrastructure, commodities and services, and downstream processing of products) and the avoided impacts due to the substitution of primary materials, mineral fertilizers, and energy.

$$\min OBJ^{CCI} = \sum_{t\in TP}\sum_{f\in F} \underbrace{CCI_{f,t}^{WASTE} + CCI_{f,t}^{AINFR} + CCI_{f,t}^{COM} + CCI_{f,t}^{RECYC} + CCI_{f,t}^{FERT} + CCI_{f,t}^{ENERGY}}_{Generated\ Impacts} - \underbrace{aCCI_{p,t}^{RECYC} - aCCI_{f,t}^{FERT} - aCCI_{f,t}^{ENERGY}}_{Avoided\ Impacts} \quad \textit{Eq. 5-2}$$

where generated impacts are (all having units of kg CO_2 eq/period):

- $CCI_{f,t}^{AINFR}$: annualized life cycle climate change impacts of the infrastructure related to facility f incurred in period t,
- $CCI_{f,t}^{WASTE}$: climate change impacts of direct GHG emissions generated by facility f in period t,
- $CCI_{f,t}^{COM}$: life cycle climate change impacts of commodities and services consumed by facility f in period t,
- $CCI_{f,t}^{RECYC}$: life cycle climate change impacts of recycling the recyclable materials recovered by facility f in period t,
- $CCI_{f,t}^{ENERGY}$: climate change impacts of using the energy recovered by facility f in period t, and
- $CCI_{f,t}^{FERT}$: climate change impacts of spreading on agricultural land the compost produced by facility f in period t.

while avoided impacts are (all having units of kg CO_2 eq/period):

- $aCCI_{f,t}^{RECYC}$: avoided life cycle climate change impacts due to the substitution of primary materials with the secondary materials sourced from facility f in period t,
- $aCCI_{f,t}^{FERT}$: avoided life cycle climate change impacts due to the substitution of mineral fertilizers with the the compost produced by facility f in period t, and
- $aCCI_{f,t}^{ENERGY}$: avoided life cycle climate change impacts due to the substitution of market energy mix with the energy recovered by facility f in period t.

<u>Network mass balance</u>

This block of equations models the flow of waste through the MSW treatment network. First, Eq. 5-3 imposes that the amount of each feedstock j collected in period t is fully processed within that period (i.e., no stocks are assumed).

$$\omega_{j,t} = \sum_{f \in JF} FFLOW_{j,f,t}\,, \qquad \forall\, j \in J, t \in TP \qquad \textit{Eq. 5-3}$$

where $\omega_{j,t}$ is a parameter for the mass of waste feedstock j collected in period t [tonnes/period] and $FFLOW_{j,f,t}$ is a continuous variable denoting the flow of feedstock j towards facility f in period t [tonnes/period].

Similarly, Eq. 5-4 imposes that the amount of intermediates and residues produced by each facility in each period is fully processed within that period.

$$FOUT_{f',u,t} = \sum_{f \in FIF \cup FRF} OFLOW_{f',u,f,t}\,, \qquad \forall\, (f',u) \in FI \cup FR, t \in TP \qquad \textit{Eq. 5-4}$$

where $FOUT_{f',u,t}$ represents the mass of output u (here intermediate or residue) produced by facility f' in period t [tonnes/period] and $OFLOW_{f',u,f,t}$ represents the flow of output u from facility f' to facility f in period t [tonnes/period].

The total mass of waste delivered to each facility in each period equals the summation of the input mass of feedstocks ($FFLOW_{j,f,t}$) and intermediates and residues ($OFLOW_{f',u,f,t}$), as shown in Eq. 5-5.

$$FIN_{f,t} = \sum_{j \in JF} FFLOW_{j,f,t} + \sum_{f' \in F} \sum_{u \in FIF \cup FRF} OFLOW_{f',u,f,t}\,, \qquad \forall\, f \in F, t \in TP \qquad \textit{Eq. 5-5}$$

where $FIN_{f,t}$ denotes the mass of waste delivered to facility f in period t [tonnes/period].

The mass of each waste material k delivered to facility f in period t is computed in Eq. 5-6 from the input mass of feedstocks and intermediates and their composition.

$$FINM_{f,k,t} = \sum_{j \in JF} FFLOW_{j,f,t} \cdot \sigma_{j,k,t} + \sum_{f' \in F} \sum_{i \in FIF} OFLOW_{f',i,f,t} \cdot ICOMP_{f',i,k,t}\,, \qquad \textit{Eq. 5-6}$$
$$\forall\, f \in F, k \in K, t \in TP$$

where $FINM_{f,k,t}$ is a continuous variable denoting the mass of waste material k delivered to facility f in period t [tonnes/period], $\sigma_{j,k,t}$ is a parameter for the share of material k in feedstock j collected in period t [% mass], and $ICOMP_{f',i,k,t}$ is a continuous variable denoting the share of material k in intermediate i produced by facility f' in period t [% mass].

It is worth noting that the composition of feedstocks in each period is a known parameter. Meanwhile, the composition of intermediate is an unknown variable that must be endogenously determined by the optimization model. In this regard, Eq. 5-7 computes the composition of the intermediate streams as the ratio of the mass of material k transferred into the intermediate i to the total mass of the intermediate.

$$ICOMP_{f,i,k,t} = \frac{FINM_{f,k,t} \cdot tc_{f,i,k}}{FOUT_{f,i,t}}, \qquad \forall\ (f \in F, i) \in FI, k \in K, t \in TP \qquad \textit{Eq. 5-7}$$

where $FOUT_{f,i,t}$ is the mass of intermediate i produced by facility f in period t [tonnes/period] and $\alpha_{f,i,k}$ is the transfer coefficient for waste material k into intermediate i at facility f [% input mass]. The division can be avoided by moving the variable $FOUT_{f,i,t}$ to the left side.

Note that both Eq. 5-6 and Eq. 5-7 feature bilinear terms resulting from the product of two variables (e.g., the multiplication of the input mass of intermediates ($OFLOW_{f',i,f,t}$) times the composition of intermediates ($ICOMP_{f',i,k,t}$)). Bilinear terms are the source of nonlinearity and nonconvexity.

Capacity adequacy

This block of equations describes the waste treatment capacity use and expansion mechanisms.

Available waste treatment capacity

Each waste treatment facility that populates the MSW treatment network has a treatment capacity. The total mass of waste delivered to each facility in each period ($FIN_{f,t}$) is bounded according to this capacity and the capacity utilization range, as shown in Eq. 5-8‡‡‡:

$$CAP_{f,t}^{TOTAL} \cdot cur_f^{min} \cdot l \leq FIN_{f,t} \leq CAP_{f,t}^{TOTAL} \cdot cur_f^{max} \cdot l, \qquad \forall\ f \in F - F^{LF}, t \in TP \qquad \textit{Eq. 5-8}$$

where $CAP_{f,t}^{TOTAL}$ denotes the treatment capacity of facility f in period t [tonnes/period] and cur_f^{min} and cur_f^{max} are the minimum and maximum capacity utilization rate for facility f [% capacity installed].

The maximum capacity utilization rate considers periods in which the facility is out of service due to feedstock fluctuation or maintenance and other technical constraints, while the minimum capacity utilization rate represents the minimum level of use required (e.g., due to economic or technical constraints).

‡‡‡ It should be noted that the capacity of AD facilities refers to the capacity of the digesters and not to the capacity of the pretreatment. Therefore, this constraint was formulated based on the mass of waste entering the digesters, i.e. after pretreatment.

The above equation applies to all the facilities but landfills. Since landfills are fundamentally continuing construction projects (Levis et al., 2013), they were separately modelled through the so-called leftover capacity. The leftover capacity is the available capacity in any period after subtracting the mass of waste accumulated in the landfill in the previous periods. Therefore, the mass of waste disposed of in a landfill in each period ($FIN_{f,t}$) must not exceed the leftover capacity in that period (Eq. 5-9).

$$FIN_{f,t} \leq CAP_{f,t}^{LF}, \qquad \forall f \in F^{LF}, t \in TP \qquad \textit{Eq. 5-9}$$

where $CAP_{f,t}^{LF}$ is the leftover capacity of landfill f in period t [tonnes].

Capacity of existing facilities

Existing facilities were considered available before the initial time period of optimization and their capacities were assumed to be given (i.e., model parameter). The optimization model decides whether an existing facility should be used or decommissioned. If an existing facility is decommissioned, its capacity turns to zero. This was mathematically modelled through the binary variable $Y_{f,t}$, which takes value of 1 if existing facility f is used in period t and 0 otherwise (Eq. 5-10).

$$CAP_{f,t}^{TOTAL} = cap_f^{exist} \cdot Y_{f,t}, \qquad \forall f \in F^{EXIST} - F^{LF}, t \in TP \qquad \textit{Eq. 5-10}$$

where cap_f^{exist} is the known capacity of existing facility f [tonnes/period].

Note that once decommissioned, existing facilities are no longer available. This restriction is imposed by Eq. 5-11 explained below.

Capacity of retrofitted facilities

The optimization could instead decide to retrofit an existing facility. The optimization model decides whether an existing facility f should be replaced with its retrofitted version f' through the binary variable $Y_{f',t}$, which takes value of 1 if the retrofitted facility f' is used in period t and 0 otherwise. Since a retrofitted facility virtually replaces an existing facility, the latter should be no longer available. Furthermore, an existing facility can have several retrofitting options but only one can be implemented. Eq. 5-11 imposes these two restrictions by forcing the optimization to choose between operating the existing facility, decommissioning it, or performing at most one of the possible retrofittings.

$$Y_{f,t} + \sum_{f' \in F^{RETROFIT}} Y_{f',t} \cdot retrofit_{f,f'} \leq 1, \qquad \forall f \in F^{EXIST} - F^{LF}, t \in TP \qquad \textit{Eq. 5-11}$$

where $retrofit_{f,f'}$ is a binary parameter equals to 1 if facility f' is a retrofitted version of existing facility f and 0 otherwise.

Retrofitting is assumed to occur at the beginning of each period. In consequence, the retrofitted facilities are readily available to be used within that period. If retrofitting is performed in period t, the capacity of a retrofitted facility f' in that period (and until the end of the decision horizon) equals the capacity of the replaced existing facility f, as shown in Eq. 5-12.

$$CAP_{f',t}^{TOTAL} = \sum_{f \in F^{EXIST} - F^{LF}} \left(cap_f^{exist} \cdot retrofit_{f,f'}\right) \cdot Y_{f',t}, \qquad \forall f' \in F^{RETROFIT}, t \in TP \qquad \textit{Eq. 5-12}$$

An existing facility cannot undergo retrofitting if it has been already decommissioned. In this regard, Eq. 5-13 ensures that retrofitting is only performed if the existing facility has been used in the previous period.

$$Y_{f,t} + \sum_{f' \in F^{RETROFIT}} Y_{f',t} \cdot retrofit_{f,f'} \leq Y_{f,t-1} + \sum_{f' \in F^{RETROFIT}} Y_{f',t-1} \cdot retrofit_{f,f'}, \quad \forall f \in F^{EXIST} - F^{LF}, t > 1 \qquad \textit{Eq. 5-13}$$

Finally, Eq. 5-14 ensures that retrofitting is only performed if the retrofitted facility is to be used until the end of the decision horizon since, once retrofitted, the lifetime of the facility is extended (minimum by 20 years):

$$Y_{f',t} \geq Y_{f',t-1}, \qquad \forall f' \in F^{RETROFIT}, t > 1 \qquad \textit{Eq. 5-14}$$

Capacity of new facilities

New facilities, by definition, are not available before the initial period. The optimization model decides to invest in new facilities through the binary variable $N_{f,t}$, which takes value of 1 if new facility f is installed in period t and 0 otherwise. Furthermore, the capacity of each new facility installed in each period is constrained within the lower and upper design capacity, as shown in Eq. 5-15.

$$N_{f,t} \cdot cap_f^{min} \leq CAP_{f,t}^{NEW} \leq N_{f,t} \cdot cap_f^{max}, \qquad \forall f \in F^{NEW}, t \in TP \qquad \textit{Eq. 5-15}$$

where $CAP_{f,t}^{NEW}$ denotes the capacity of new facility f installed in period t [tonnes/period] and cap_f^{min} and cap_f^{max} are the lower and upper design capacity of new facility f [tonnes/period].

It was assumed that the installation of new facilities occurs at the beginning of each period and therefore they are readily available to be used within that period (i.e., over-night investments). In the first period, the capacity of new facilities equals whatever capacity is installed during that period (Eq. 5-16). In the following periods, the capacity of new facilities equals the available capacity at the end of the previous period plus the new capacity installed during the actual period (Eq. 5-17).

$$CAP_{f,t}^{TOTAL} = CAP_{f,t}^{NEW}, \qquad \forall\, f \in F^{NEW} - F^{LF}, t = 1 \qquad \textit{Eq. 5-16}$$

$$CAP_{f,t}^{TOTAL} = CAP_{f,t-1}^{TOTAL} + CAP_{f,t}^{NEW}, \qquad \forall\, f \in F^{NEW} - F^{LF}, t > 1 \qquad \textit{Eq. 5-17}$$

A set of constraints were implemented to guarantee that no more than a certain number of facilities of the same type is built over the decision horizon. For example, Eq. 5-18 and Eq. 5-19 ensure that at most one new incinerator and one new landfill are opened.

$$\sum_{f \in F^{INC} \cup F^{NEW}} \sum_{t \in TP} N_{f,t} \leq 1 \qquad \textit{Eq. 5-18}$$

$$\sum_{f \in F^{LF} \cup F^{NEW}} \sum_{t \in TP} N_{f,t} \leq 1 \qquad \textit{Eq. 5-19}$$

Furthermore, Eq. 5-20 guarantees that a new landfill is installed only if the leftover capacity of the existing landfill is zero (i.e., it cannot receive more waste):

$$1 - N_{LF\_N,t} \geq \frac{CAP_{LF\_E,t}^{LF} - FIN_{LF\_E,t}}{cap_{LF\_E}^{exist}}, \qquad \forall\, t \in TP \qquad \textit{Eq. 5-20}$$

Leftover capacity of landfills

The leftover capacity of existing landfills in the first period was assumed to be an input parameter (Eq. 5-21). In the following periods, the leftover capacity is the difference between the capacity in the previous period and the mass of waste accumulated in the landfill in the previous periods (Eq. 5-22).

$$CAP_{f,t}^{LF} = cap_f^{exist}, \qquad \forall\, f \in F^{LF} \cup F^{EXIST}, t = 1 \qquad \textit{Eq. 5-21}$$

$$CAP_{f,t}^{LF} = CAP_{f,t-1}^{LF} - FIN_{f,t-1}, \qquad \forall\, f \in F^{LF} \cup F^{EXIST}, t > 1 \qquad \textit{Eq. 5-22}$$

The leftover capacity of new landfills in the first period equals whatever capacity is installed during that period multiplied with the expected lifetime of the landfill (n_f) (Eq. 5-23). In the following time periods, the leftover capacity accounts for the mass of waste accumulated in the landfill and the new capacity installed (Eq. 5-24).

$$CAP_{f,t}^{LF} = CAP_{f,t}^{NEW} \cdot n_f, \qquad \forall\, f \in F^{LF} \cup F^{NEW}, t = 1 \qquad \textit{Eq. 5-23}$$

$$CAP_{f,t}^{LF} = CAP_{f,t-1}^{LF} - FIN_{f,t-1} + CAP_{f,t}^{NEW} \cdot n_f, \qquad \forall\, f \in F^{LF} \cup F^{NEW}, t > 1 \qquad \text{Eq. 5-24}$$

Waste treatment

This block of equations quantifies the amount of intermediates, residues, products, and emissions produced by each waste treatment facility in each period. To this end, the LCIs developed in Chapter 4 were implemented into the optimization framework to compute the following variables:

- $FOUT_{f,u,t}$: Amount of output u (either intermediate, residue, or product) produced by facility f in period t [units/period],
- $COMIN_{f,c,t}$: Amount of commodity or service c consumed by facility f in period t [units/period], and
- $GHG_{f,e,t}$: Amount of GHG e emitted by facility f in period t [kg/period].

The underlaying mathematical relationships calculate these variables based on the mass and physicochemical properties of the 18 input waste materials, following the same principles described in Chapter 4. For example, the mechanical sorting carried out at MRFs was modeled with the input mass of waste materials (i.e., variable $FINM_{f,k,t}$) and the material-specific transfer coefficients. The amount of electricity generated by incineration was calculated from the input mass of waste materials, their burnability, LHV, and moisture content, and the net electrical efficiency of the facility. Fossil and biogenic CO_2 emissions from incineration were calculated from the input mass of waste materials and their fossil and biogenic carbon content. Overall, by adopting the predictive LCIs from Chapter 4, the optimization framework can capture the response of waste treatment facilities to changes in waste composition.

Economic assessment

This block of equations computes the capital costs, fixed and variable operating costs, and revenues associated with each waste treatment facility.

Capital costs

The capital costs of existing facilities were assumed amortized. Therefore, they do not affect future decisions and they were set to zero. The capital costs of decommissioning existing facilities were also ignored. On the other hand, the capital costs of retrofitting an existing facility or installing a new facility are endogenously calculated by the optimization model though the variable $COST_{f,t}^{CAPEX}$ that denotes the capital inversion (i.e., EUR) in facility f in period t.

The capital costs of retrofitting existing facilities equal the amount of money required to perform the retrofitting multiplied by the binary variable $Y_{f,t}$ (Eq. 5-25). Thus, the capital costs of retrofitting incur exclusively when the binary variable turns to one, which would mean that the retrofitted facility f is used in period t. To ensure that the capital costs incur only in the period when the retrofitting is performed, the value of the binary variable in the previous period is subtracted from the current value, as shown in Eq. 5-26.

$$COST_{f,t}^{CAPEX} = capex_f^{retrofit} \cdot Y_{f,t}, \qquad \forall\, f \in F^{RETROFIT}, t = 1 \qquad \textit{Eq. 5-25}$$

$$COST_{f,t}^{CAPEX} = capex_f^{retrofit} \cdot (Y_{f,t} - Y_{f,t-1}), \qquad \forall\, f \in F^{RETROFIT}, t > 1 \qquad \textit{Eq. 5-26}$$

where $capex_f^{retrofit}$ represents the capital costs of retrofitted facility f [EUR].

The capital costs of new facilities were modelled through linear functions that capture the relationship between the capital costs and the capacity of the new facility (i.e., economies of scale), as shown in Eq. 5-27.

$$COST_{f,t}^{CAPEX} = a \cdot CAP_{f,t}^{NEW} + b \cdot N_{f,t}, \qquad \forall\, f \in F^{NEW}, t \in TP \qquad \textit{Eq. 5-27}$$

where a and b are the slope and intercept of the capital costs function. Note that although the relationship between capital costs and capacity is linear, the unit capital cost decreases with capacity due to the value of the intercept (see Section 5.3.2).

Capital costs are annualized to incorporate them into the net cash flow. This is achieved through the capital recovery factor (CRF), which is defined as $\frac{i \cdot (1+i)^{n_f}}{(1+i)^{n_f} - 1}$, where i is the interest of borrowing capital and n_f is the lifetime of the facility (Cimpan et al., 2016). The CRF converts capital costs to annual payment over n years at a specified interest rate i.

The annualized capital costs are paid until the end of the decision horizon (note that the lifetime of the facilities, between 20 and 30 years, is higher than the decision horizon analyzed, usually 20 years). In consequence, the annualized capital costs incurred in each period is calculated from the capital costs incurred during the actual period multiplied by the CRF multiplied by the length in years of the period plus the annualized capital costs related to the new facilities installed in previous periods (Eq. 5-28).

$$COST_{f,t}^{ACC} = COST_{f,t}^{CAPEX} \cdot \frac{i \cdot (1+i)^{n_f}}{(1+i)^{n_f} - 1} \cdot l + COST_{f,t-1}^{ACC}, \qquad \textit{Eq. 5-28}$$
$$\forall\, f \in F^{RETROFIT} \cup F^{NEW}, t \in TP$$

where $COST_{f,t}^{ACC}$ denotes the annualized capital costs related to facility f incurred in period t [EUR/period].

Fixed operating costs

The fixed operating costs were assumed to consist of maintenance, insurance, and labor costs. These costs do not depend on the input mass of waste but on the treatment capacity. Consequently, they incur exclusively if the facility is operative. To consider economies of scale, fixed operating costs were modelled as a function of treatment capacity using linear regression models. Eq. 5-29 and Eq. 5-30 perform the calculation for existing and new facilities, respectively.

$$COST_{f,t}^{FO} = \left(a \cdot CAP_{f,t}^{TOTAL} + b \cdot Y_{f,t}\right) \cdot l, \qquad \forall\, f \in F^{EXT} - F^{LF}, t \in TP \qquad \textit{Eq. 5-29}$$

$$COST_{f,t}^{FO} = \left(a \cdot CAP_{f,t}^{TOTAL} + b \cdot \sum_{\substack{t \in TP \\ tt \leq t}} N_{f,tt}\right) \cdot l, \qquad \forall\, f \in F^{NEW} - F^{LF}, t \in TP \qquad \textit{Eq. 5-30}$$

where $COST_{f,t}^{FO}$ denotes the fixed operating costs of facility f in period t [EUR/period] and a and b are the slope and intercept of the fixed operating costs function.

The fixed operating costs of landfills were assumed proportional to the mass of waste disposed of in landfill, as shown in Eq. 5-31.

$$COST_{f,t}^{FO} = a \cdot FIN_{f,t}, \qquad \forall\, f \in F^{LF}, t \in TP \qquad \textit{Eq. 5-31}$$

Variable operating costs

Variable operating costs incorporate the costs of commodities and services consumed by waste treatment facilities. Eq. 5-32 calculates these costs as the summation of the amounts of commodities and services consumed (as calculated in the LCI block of equations) multiplied by the corresponding purchasing price.

$$COST_{f,t}^{VO} = \sum_{c \in C} COMIN_{f,c,t} \cdot price_{c,t}\,, \qquad \forall\, f \in F, t \in TP \qquad \textit{Eq. 5-32}$$

where $COST_{f,t}^{VO}$ denotes the variable operating costs of facility f in period t [EUR/period], $COMIN_{f,c,t}$ is the amount of commodity or service c consumed by facility f in period t [units/period], and $price_{c,t}$ is the purchasing price of commodity or service c in period t [EUR/unit].

Revenue

Waste treatment facilities generate revenue from the sale of recyclable materials, compost, and energy. The revenue generated by each facility in each period equals the sales of products (as calculated in the LCI block) multiplied by the product selling price.

$$REV_{f,t} = \sum_{p \in FP} FOUT_{f,p,t} \cdot rev_{p,t}, \qquad \forall\, f \in F, t \in TP \qquad \textit{Eq. 5-33}$$

where $REV_{f,t}$ denotes the revenues generated by facility f in period t [EUR/period] and $rev_{p,t}$ is a parameter for the selling price of product p in period t [EUR/unit].

Climate change impacts assessment

This block of equations calculates the generated and avoided life cycle climate change impacts associated with each waste treatment facility (i.e., the variables of Eq. 5-2) following the same procedure as in Chapter 4. Thus, the climate change impacts of direct GHG emissions were calculated from the amount of GHGs emitted (provided by the waste treatment block) and the corresponding characterization factors. The impacts of new infrastructure, the supply of commodities and services, processing activities, and substitution are calculated by multiplying their amount with the corresponding unit life cycle impacts.

Policy targets

This block of equations implements waste policy targets. For the case study of Madrid, scenarios that align with the 10% landfill goal by 2035 were assessed. Thus, Eq. 5-34 ensures that the landfill rate in each period is below a pre-defined target.

$$LFR_t \leq lft_t, \qquad \forall\, t \in TP \qquad \textit{Eq. 5-34}$$

where LFR_t is the landfill rate in period t [% MSW] and lft_t is the landfill target established for period t [% MSW]. The calculation of the landfill rate follows the rules established in Article 5a in the Landfill Directive (*cf.* Section 4.2.3).

5.2.4 Solution procedure

The optimization model presented above corresponds to a large-scale nonconvex MINLP. The nonconvexities arise due to the presence of bilinear terms (i.e., the product of two variables) in the equations that compute the mass flow of waste materials entering each facility (Eq. 5-6) and the composition of intermediate streams (Eq. 5-7). A heuristic algorithm was implemented to solve the nonconvex MINLP model. The algorithm consists of the successive solution of a relaxed (linear) version of the nonconvex MINLP and the original model itself (Figure 5-1). First, the nonconvex MINLP model (M1) was reformulated into a mixed-integer linear programming (MILP) model (M2) through the relaxation of the bilinear terms. This relaxation was built using the McCormick method, which consists of substituting the nonconvex equations with their corresponding convex envelopes (McCormick, 1976). The resulting linear model M2 can be solved to global optimality with standard solvers. Subsequently, the solution obtained for M2 was used as the initial guess for the solution of the original model M1. This heuristic procedure has the advantage that quickly generates a feasible optimal solution that is in the region near the global optimum solution. It cannot ensure that the global optimum is found, which is, however, a common limitation when dealing with large-scale nonconvex problems (Misener and Floudas, 2009).

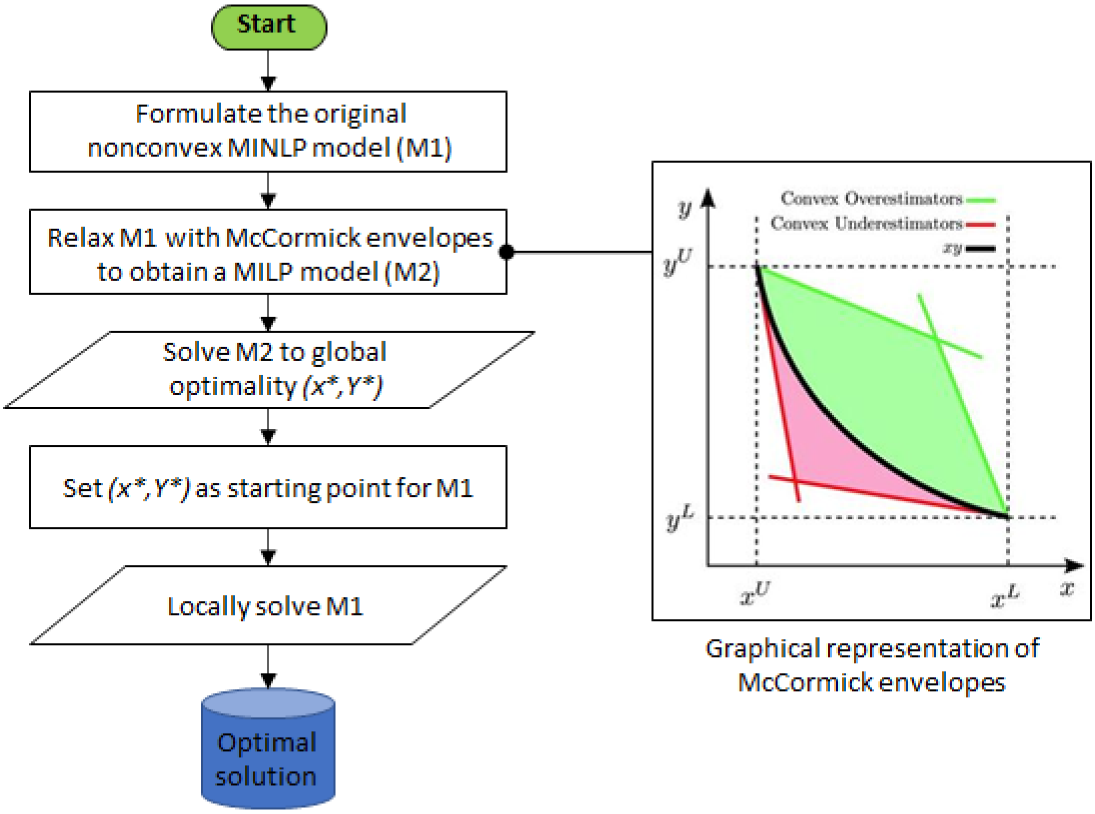

Figure 5-1. Heuristic algorithm implemented in this work to solve a large-scale nonconvex mixed-integer nonlinear program (MINLP).

The multi-period waste treatment optimization framework was implemented in Pyomo 6.1.2, a Python-based open-source software package for formulating and solving optimization problems (Hart et al., 2017, 2011). The framework uses CPLEX and SBB solvers to solve the MILP and MINLP models, respectively. The solvers are accessed through the interface that connects Python to the General Algebraic Modelling System (GAMS 35.2.0).

5.3 Assessing optimal waste-to-energy pathways for Madrid

This section presents the application of the optimization framework to assess the economic and climate optimal WTE pathways that could be prioritized in Madrid over 2020-2040. Section 5.3.1 defines the scenarios for analysis, while Section 5.3.2 summarizes the data used for the case study. Subsequently, Section 5.3.3 presents the optimal MSW treatment network according to each scenario, while Section 5.3.4 and Section 5.3.5 focus on the long-term analysis of AD and incineration, respectively.

5.3.1 Scenarios definition

The optimal WTE pathways were assessed under three scenarios that differ in the objective function and the major restrictions imposed (Table 5-2). The Business-as-Usual (BAU) scenario aims at determining the optimal MSW treatment network that maximizes the NPV while using the existing facilities with no landfilling restrictions. Investment in new facilities is allowed whenever the existing capacity is not enough to treat all the waste. However, the BAU does not consider the installation of new incineration capacity. The other two scenarios aim at determining the optimal MSW treatment network that maximizes the NPV (Max. Net Present Value scenario) or minimizes the climate change impacts (Min. Climate Change scenario) with total freedom of choice of waste treatment facilities. The number of new incineration facilities has been restricted to one, as it is unlikely that more than one new facility could be installed. Furthermore, these two scenarios were forced to achieve a 10% landfill target by 2035, in line with Directive 2018/850. The landfill target was assumed to decrease linearly from 40% in the first period to 10% in the last one.

Table 5-2. Optimization scenarios.

<table>
<tr><th>Scenario</th><th>Objective function</th><th>Main restrictions</th></tr>
<tr><td>Business-as-Usual (BAU)</td><td>Maximize net present value (NPV)</td><td>• Use of existing facilities.
• Limited investment in new facilities.
• No investments in new incineration capacity
• No landfill targets
• No landfilling of household residual waste without previous treatment</td></tr>
<tr><td>Max. Net Present Value</td><td>Maximize net present value (NPV)</td><td rowspan="2">• Freedom of choice of waste treatment facilities.
• Only one new incineration facility
• 10% landfill target by 2035
• No landfilling of household residual waste without previous treatment</td></tr>
<tr><td>Min. Climate Change</td><td>Minimize climate change impacts</td></tr>
</table>

5.3.2 Case study data

The following paragraphs summarize the data implemented in the optimization model to perform the case study of Madrid.

Mass and composition of feedstocks

The mass and composition of feedstocks collected in Madrid over 2020-2040 was projected following the methodology proposed in Chapter 3. Waste generation and collection was calculated for each year over the decision horizon and aggregated into 5-year periods as required by the optimization framework. Table 5-3 summarizes the average annual amounts of feedstocks collected in each period (note that the optimization model considers the accumulated amounts over each 5-year period). The composition of the packaging waste, paper/cardboard, glass, and organic waste collected from households and the organic waste collected from commercial activities were assumed to remain constat over the decision horizon (see Table 3-4 for current compositions). The composition of the residual waste streams was adjusted accordingly. Furthermore, the physicochemical properties of the waste materials are the same as described in the previous chapters.

Table 5-3. Average annual amounts of feedstocks collected in Madrid in each 5-year period (tonnes/year).

Feedstock	2020-2025	2025-2030	2030-2035	2035-2040
Residual waste, households	691,432	609,086	539,470	490,776
Packaging waste, households	107,636	112,647	116,390	118,120
Paper/cardboard, households	81,442	111,880	136,956	153,269
Glass, households	55,574	64,161	71,581	77,074
Organic waste, households	142,775	181,594	215,481	241,148
Residual waste, commercial	89,776	90,285	91,121	92,475
Organic waste, commercial	14,405	15,491	16,251	16,493
Street cleaning waste	118,042	119,850	121,658	123,466

Techno-economic parameters

The MSW treatment network was populated with 33 facilities, including 10 existing facilities, seven retrofitted facilities, and 15 new facilities. Each facility was characterized by the techno-economic parameters, including treatment capacity, capacity utilization rate, capital costs, fixed operating costs, and lifetime (Table 5-4).

The capacities of the existing waste treatment facilities in Madrid were defined in Chapter 4 (*cf.* Table 4-17). Furthermore, seven retrofitted facilities were implemented. First, the existing MRFs for residual and packaging waste can be retrofitted to improve the sorting efficiency of recyclable materials. This can be achieved by replacing the sorting equipment (e.g., trommels, air classifier, ballistic separator, magnetic separator, eddy current, or NIR sorters). Secondly, the existing AD facilities can be retrofitted to switch from biomethane upgrading (current situation) to CHP generation. Finally, as established in Chapter 4, the organic waste collected separately and the residual organic waste separated at MRFs cannot be treated together in order to prevent the cross-contamination of the compost. In this regard, it was assumed that the existing AD and composting facilities dedicated to treat residual organic waste can be retrofitted to accept organic waste collected separately. Note that the treatment capacities of the retrofitted facilities equal the capacities of the replaced existing facilities.

The new facilities were characterized by a lower and upper design capacity to avoid unrealistically sized investments (e.g., very small or very large incinerators). This range was established based on a review of current industrial practices and the literature (see Table C-1 in Appendix C). A continuous capacity range was defined for most of the facilities. For example, the design capacity of a new AD facility ranges from 25,000 to 300,000 tonnes/year. Conversely, three new incineration facilities were defined based on the design capacity, namely a small facility (50,000 to 100,000 tonnes/year), a medium facility (100,000 to 250,000 tonnes/year), and a large facility (250,000 to 450,000 tonnes/year). This discretization of incineration was performed to account for the increase in the electrical efficiency at higher capacities, as will be shown later in the technology parameters.

The maximum capacity utilization rate was assumed equal to 91% for all the facilities (but landfills, which are always available to receive waste), based on 8,000 hours of operation per year. The minimum capacity utilization rate, on the other hand, is specific to each facility and was established based on current practices in Madrid and the literature (see Table C-2 in Appendix C).

Table 5-4. Techno-economic parameters of the waste treatment facilities implemented in the optimization framework.

Facility	**Capacity** (tonnes/year)	**Capacity utilization** (% capacity installed)	**Capital costs**	**Fixed operating costs**	**Lifetime** (years)
Existing MRF for residual waste	1,054,000	50–91%	–	o = 13x + 3,439,445	–
Existing MRF for packaging waste	126,500	50–91%	–	o = 33x + 2,131,015	–
Existing MRF for paper/cardboard	Unconstrained	0–100%	–	–	–
Existing MRF for glass	Unconstrained	0–100%	–	–	–
Existing tunnels composting for residual organic waste	331,290	10–91%	–	o = 1x + 310,323	–
Existing AD for organic waste collected separately (w/ biomethane upgrading)	161,000	50–91%	–	o = 10x + 1,982,379	–
Existing AD for residual organic waste (w/ biomethane upgrading)	108,175	50–91%	–	o = 10x + 1,982,379	–
Existing incineration	328,500	70–91%	–	o = 18x + 6,525,117	–
Existing sanitary landfill	3,134,300	–	–	5 EUR/tonne waste	–
Existing security landfill for fly ash	Unconstrained	–	–	5 EUR/tonne waste	–
Retrofitted MRF for residual waste: improve sorting efficiency	1,054,000	50–91%	66,087,772 EUR	o = 10x + 2,277,840	20
Retrofitted MRF for packaging waste: improve sorting efficiency	126,500	50–91%	12,663,675 EUR	o = 23x + 1,389,781	20
Retrofitted AD for organic waste collected separately: biomethane upgrading is substituted by CHP generation	161,000	50–91%	6,860,993 EUR	o = 11x + 1,978,454	20
Retrofitted AD for residual organic waste: biomethane upgrading is substituted by CHP generation	108,175	50–91%	5,091,654 EUR	o = 10x + 1,920,095	20
Retrofitted AD for residual organic waste: accepts organic waste collected separately	108,175	50–91%	–	–	20
Retrofitted AD for residual organic: accept organic waste collected separately and biomethane upgrading is substituted by CHP generation	108,175	50–91%	5,091,654 EUR	o = 10x + 1,920,095	20
Retrofitted composting: accept organic waste collected separately	331,290	10–91%	–	–	20
New MRF for packaging waste	25,000–100,000	50–91%	c = 179x + 2,706,231	o = 33x + 2,131,015	20
New RDF facility	45,000–360,000	50–91%	c = 54x + 3,222,995	o = 10x + 3,303,932	20
New tunnels composting for residual organic waste	10,000–300,000	10–91%	c = 29x + 1,852,654	o = 1x + 310,323	25
New tunnels composting for organic waste collected separately	10,000–300,000	10–91%	c = 29x + 1,852,654	o = 1x + 310,323	25
New windrow composting for residual organic waste	10,000–300,000	10–91%	c = 28x + 1,782,914	o = 1x + 312,904	25
New windrow composting for organic waste collected separately	10,000–300,000	10–91%	c = 28x + 1,782,914	o = 1x + 312,904	25

Table 5-4. *(continued)*

Facility	**Capacity** (tonnes/year)	**Capacity utilization** (% capacity installed)	**Capital costs**	**Fixed operating costs**	**Lifetime** (years)
New AD for organic waste collected separately (w/ biomethane upgrading)	25,000–300,000	50–91%	c = 327x + 18,974,502	o = 10x + 1,982,379	25
New AD for organic waste collected separately (w/ CHP generation)	25,000–300,000	50–91%	c = 350x + 18,849,896	o = 11x + 1,978,454	25
New AD for residual organic waste (w/ biomethane upgrading)	25,000–300,000	50–91%	c = 327x + 18,974,502	o = 10x + 1,982,379	25
New AD for residual organic waste (w/ CHP generation)	25,000–300,000	50–91%	c = 350x + 18,849,896	o = 11x + 1,978,454	25
New small incineration	50,000–100,000	70–91%	c = 522x + 21,371,794	o = 31x + 2,585,538	30
New medium incineration	100,000–250,000	70–91%	c = 381x + 37,205,879	o = 24x + 4,075,156	30
New large incineration	250,000–450,000	70–91%	c = 296x + 58,523,300	o = 18x + 6,525,117	30
New large incineration for RDF	250,000–450,000	70–91%	c = 296x + 58,523,300	o = 18x + 6,525,117	30
New landfill	10,000–2,000,000	–	c = 27x + 559,996	5 EUR/tonne waste	25

Note: c: capital costs (EUR); x: treatment capacity (tonnes/year); o: fixed operating costs (EUR/year). AD: anaerobic digestion. CHP: combined heat and power. MRF: material recovery facility. RDF: refuse-derived fuel.

The capital costs of retrofitting the existing MRFs were calculated based on the cost of the new sorting equipment, which was assumed 50% of the total capital costs of a new MRF (Cimpan et al., 2016). The capital costs of retrofitting an existing AD facility to implement CHP generation were assumed equal to the cost of the CHP unit. The capital costs of the new facilities, on the other hand, were modelled as a function with the form $c = ax + b$, where c denotes the capital costs and x is the treatment capacity. These functions establish a linear relationship between the capital costs and capacity and allow capturing economies of scale. This implies a reduction in the capital costs per unit of capacity at larger capacities, as shown in Figure 5-2.

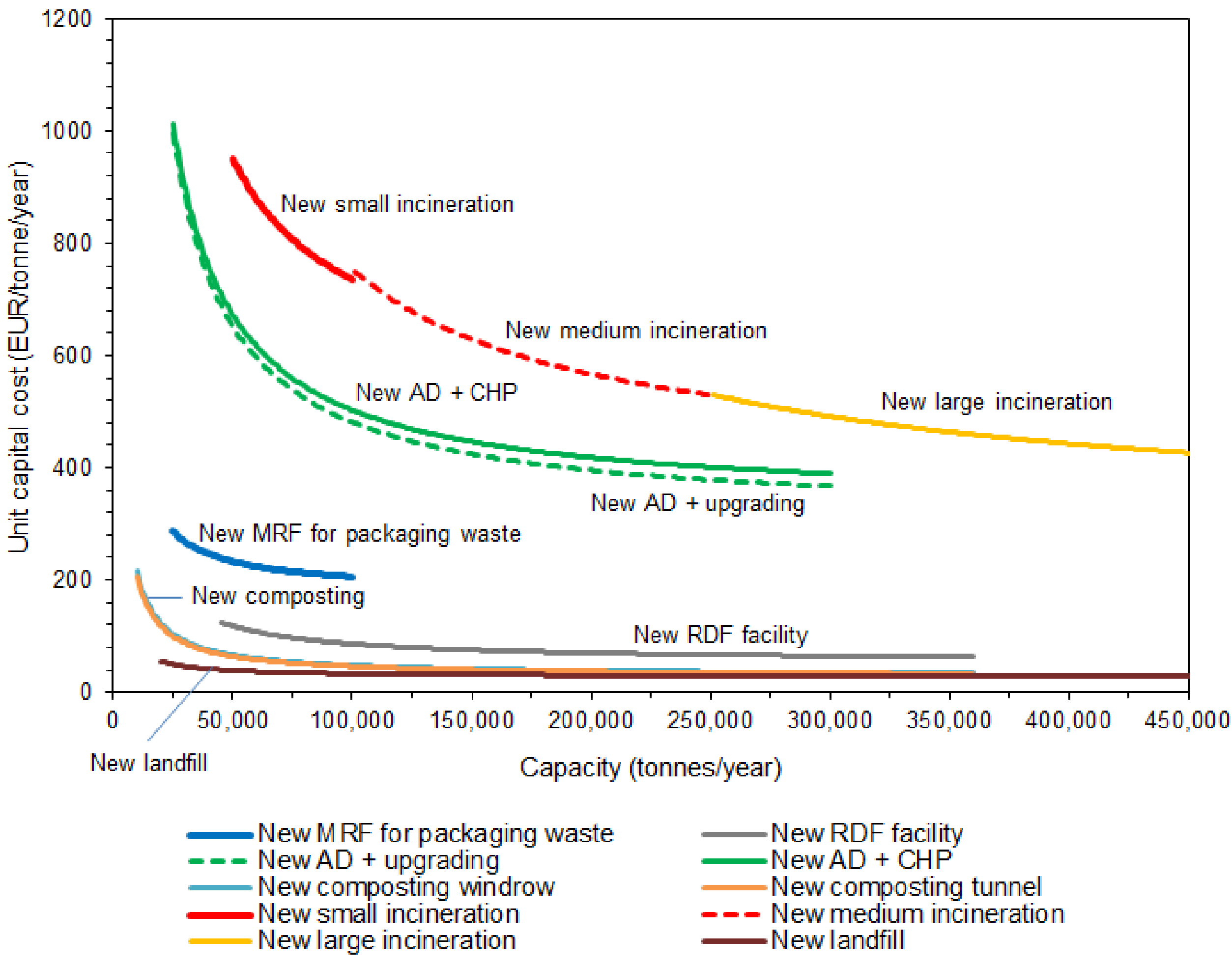

Figure 5-2. Unit capital costs of new waste treatment facilities as a function of capacity based on the linear functions developed in this thesis. AD: anaerobic digestion. CHP: combined heat and power. MRF: material recovery facility. RDF: refuse-derived fuel.

To perform the linear regression and obtain the parameters a and b, the capital costs of each new facility were computed at a range of capacity levels (between the lower and upper design capacity) following two approaches: (1) literature data if available or (2) using the scaling factor method. The scaling factor method establishes that the capital costs at any capacity can be approximated from the capital costs and the capacity of a reference facility (Ng and Phan, 2021; Sadhukhan et al., 2014).

Further details on the development of the linear capital costs functions are provided in Table C-3 in Appendix C.

The fixed operating costs were also modelled as a function with the form $o = ax + b$, where o denotes the fixed operating costs and x is the treatment capacity. To perform this linear regression, the fixed operating costs were computed at a range of capacity levels based on the annual maintenance, insurance, and labor costs. Annual maintenance and insurance costs were calculated as a percentage of capital costs, while the annual labor costs were calculated from the number of workers and the average gross annual salary, supervision costs, and salary overhead (see Table C-4 in Appendix C).

Finally, the variable operating costs and the revenues generated by each facility were calculated from the prices of the commodities and services (Table C-5 in Appendix C) and the selling price of recyclable materials, compost, and energy (Table C-6 in Appendix C), respectively.

Box 5.1 Harmonization of economic data

The economic data used for the case study was harmonized to EUR2019. Capital costs from years other than 2019 were adjusted by applying the Chemical Engineering Plant Cost Index (CEPCI) as follows:

$$Cost_{ref} = Cost_t \cdot \left(\frac{CEPCI_{ref}}{CEPCI_t}\right)$$

where $Cost_{ref}$ is the capital cost in the reference year (2019), $Cost_t$ is the capital cost in the original year t; and $CEPCI_{ref}$ and $CEPCI_t$ are the CEPCI value in the years 2019 and t, respectively.

Prices of commodities and services and revenues from the sale of materials and energy were adjusted by applying the Industrial Producer Price Index (PPI).

Technology parameters

Technology parameters include all the parameters required to model the mass and energy balances of waste treatment facilities, i.e., transfer coefficients, energy efficiencies, commodities and services consumption, transport distances, and other technology-specific parameters. The transfer coefficients of existing and new MRFs, RDF facility, and AD pre-treatment were presented in Table 3-5. The retrofitted MRFs were assumed to have the same transfer coefficients as the new MRFs. In general, the existing, retrofitted, and/or new MRFs, composting, AD, and landfilling facilities were modelled with the technology parameters presented in the LCIs developed in Chapter 4 (*cf.* Section 4.2.2).

Following the same assumptions as in Chapter 4, the new incineration facilities were assumed to consist of a moving grate furnace, whereas the existing facility is based on the circulating fluidized bed technology. All the incineration facilities are equipped with the same air pollution control (APC) system (see Chapter 4 for further details). The major differences between the existing and new facilities are the net electrical efficiency, the production rate of ashes, and the recovery of metals from the bottom ash (Table 5-5). The net electrical efficiency of the existing incineration facility was calculated at 12% (see Chapter 3 and Table B-1 in Appendix B). For the new facilities, the net electrical efficiency ranges from 18% for the small facility to 21% for the medium-scale facility, 26% for the large facility, and 30% for the large facility dedicated to RDF (Table C-7 in Appendix C). Thus, the efficiency is higher for large facilities due to improved steam parameters (*cf.* Section 1.2.1). Furthermore, the new facilities were assumed to recover both ferrous metals (scrap steel) and non-ferrous metals (aluminium) from the bottom ash.

Table 5-5. Technology parameters of existing and new incineration facilities implemented into the optimization model.

Parameter	Unit	Existing incineration	New small incineration	New medium incineration	New large incineration	New large incineration for RDF
Net electrical efficiency	% waste LHV	12	18	21	26	30
Bottom ash	kg/tonne waste	37.5	250	250	250	250
Fly ash	kg/tonne waste	73.8	65	65	65	65
Ferrous metal recovery	% input mass	80	80	80	80	80
Aluminium recovery	% input mass	0	50	50	50	50

LCA data

The LCA data includes the unit life cycle impacts of background processes and characterization factors as documented in Chapter 4.

5.3.3 Optimal municipal solid waste treatment network

Figure 5-3 shows the optimal mass of waste delivered to each waste treatment facility in each period, while Table 5-6 summarizes the optimal capacities of waste treatment facilities. The following paragraphs discuss these results according to each scenario.

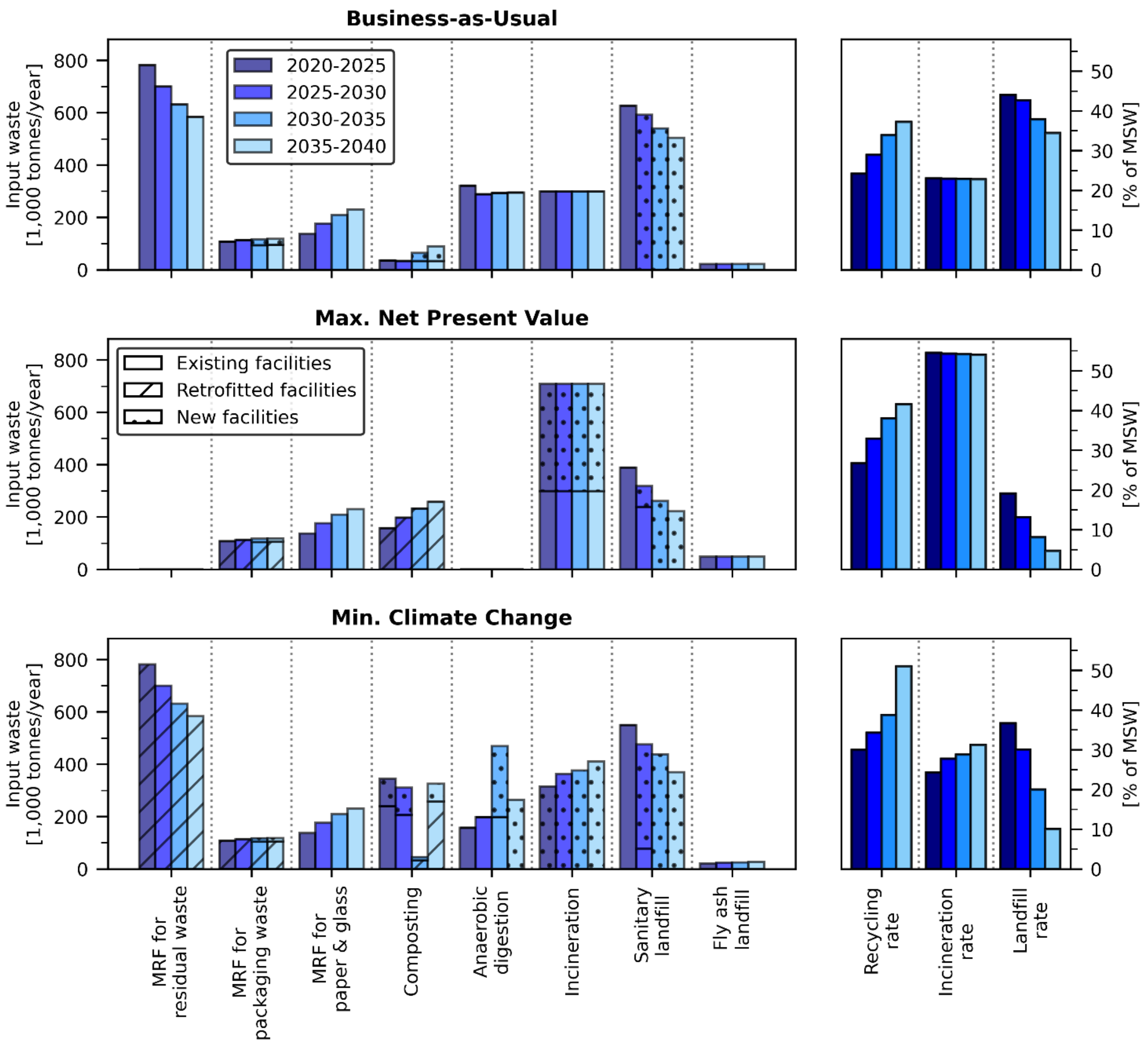

Figure 5-3. Input mass of waste to each treatment in each period in the Business-as-Usual, Max. Net Present Value, and Min. Climate Change scenarios. The recycling, incineration, and landfill rates in each period are displayed on the right. MRF: material recovery facility.

Table 5-6. Optimal waste treatment capacities in the Business-as-Usual, Max. Net Present Value, and Min. Climate Change scenarios. Units are tonnes waste/year for all the facilities but landfills, which refer to the leftover capacity at the beginning of each period.

Facility	Business-as-Usual				Max. Net Present Value				Min. Climate Change			
	2020-2025	2025-2030	2030-2035	2035-2040	2020-2025	2025-2030	2030-2035	2035-2040	2020-2025	2025-2030	2030-2035	2035-2040
Existing MRF for residual waste	1,054,000	1,054,000	1,054,000	1,054,000	D	D	D	D	1,054,000 R	1,054,000 R	1,054,000 R	1,054,000 R
Existing MRF for packaging waste	126,500	126,500	126,500	126,500	126,500 R	126,500 R	126,500 R	126,500 R	126,500 R	126,500 R	126,500 R	126,500 R
Existing MRF for paper/cardboard	U	U	U	U	U	U	U	U	U	U	U	U
Existing MRF for glass	U	U	U	U	U	U	U	U	U	U	U	U
Existing AD for organic waste collected separately	161,000	161,000	161,000	161,000	D	D	D	D	161,000	161,000	161,000	D
Existing AD for residual organic waste	108,175	108,175	108,175	108,175	D	D	D	D	D	D	D	D
Existing composting tunnels	331,290	331,290	331,290	331,290	331,290 R	331,290 R	331,290 R	331,290 R	331,290	331,290	331,290 R	331,290 R
Existing incineration	328,500	328,500	328,500	328,500	328,500	328,500	328,500	328,500	D	D	D	D
Existing sanitary landfill	3,134,300	0	0	0	3,134,300	1,191,114	0	0	3,134,300	385,610	0	0
Existing fly ash landfill	U	U	U	U	U	U	U	U	U	U	U	U
New MRF for packaging waste	–	–	25,000	25,000	–	–	25,000	25,000	–	–	25,000	25,000
New AD for residual organic waste	–	–	–	–	–	–	–	–	–	–	173,862	173,862
New composting windrow	–	–	62,001	62,001	–	–	–	–	–	–	–	–
New composting tunnel	–	–	–	–	–	–	–	–	114,974	114,974	114,974	114,974
New large incineration	–	–	–	–	450,000	450,000	450,000	450,000	450,000	450,000	450,000	450,000
New landfill	–	8,177,784	5,217,353	2,519,552	–	2,815,250	2,414,973	1,109,504	–	6,024,029	4,031,638	1,846,030

Note: U: unconstrained capacity. D: decommissioning. R: retrofitting.

Business-as-Usual scenario

The BAU scenario depicts a situation similar to the current one extended up to 2040. The household and commercial residual waste streams are delivered to the existing MRFs, whose capacity is enough to absorb the 781,207 and 583,251 tonnes/year collected in the first and last period, respectively. The packaging waste, paper/cardboard, and glass streams are also delivered to the corresponding MRFs. Due to the increasing separate collection of packaging waste (from 107,636 to 118,120 tonnes/year over the decision horizon), a new MRF with 25,000 tonnes/year capacity is installed in the period 2030-2035. The capacity of the MRFs for paper/cardboard and glass was assumed unconstrained (i.e., unlimited capacity).

The existing AD facilities receive 321,114 and 294,896 tonnes/year of organic waste in the first and last period, respectively. There are two AD facilities, one dedicated to the organic waste collected separately and another one dedicated to the residual organic waste because these two streams cannot be mixed. The amount of feedstock received by the first facility increases from 157,180 to 201,220 tonnes/year between the first and last period due to an increase in the separate collection of organic waste. Conversely, the second facility receives 163,963 tonnes/year of residual organic waste in the first period but only 93,676 tonnes/year in the last one. Note that, in this study, the existing AD facilities were assumed to dedicate all the available biogas to produce biomethane.

The existing composting facility is used to treat an average of 33,608 tonnes/year of residual organic waste in each period. Furthermore, the existing AD capacity proves insufficient to deal with the increasing amount of organic waste collected separately. Thus, a new windrow composting facility with 62,001 tonnes/year capacity is installed in the period 2030-2035. Windrow composting is preferred due to its significant lower costs.

The existing incineration facility is used over the entire decision horizon receiving an average of 298,935 tonnes/year of rejects from MRFs. These rejects come from the MRFs that sort residual waste (between 226,221 and 234,508 tonnes/year) and the MRFs that sort packaging waste (between 64,427 and 72,714 tonnes/year). Finally, the capacity of the existing landfill is exceeded in the first period, thus a new landfill with a capacity for 8.2 million tonnes of waste is opened. Overall, the recycling and landfill rates in this scenario reach 37% and 34% by 2035-2040, while the incineration rate remains constant at 23%.

Max. Net Present Value scenario

The Max. Net Present Value and Min. Climate Change scenarios depict two different pathways that would allow Madrid to achieve the goal of 10% landfill by 2035. In the Max. Net Present Value scenario, the landfill rate reaches 5% in the period 2035-2040. This is largely achieved through the intensive use of incineration. An average of 708,435 tonnes/year of waste are incinerated in each period in this scenario. A major difference over the BAU is that the household and commercial residual waste streams are directly incinerated, while the existing MRFs for sorting residual waste are decommissioned. Since the capacity of the existing incineration facility is not enough, a new large incineration facility with 450,000 tonnes/year capacity is installed from the start. The total incineration capacity would therefore reach 778,500 tonnes/year, while the incineration rate equals 54% in each period. This incineration rate is similar to that of EU countries that achieve less than 1% landfill, i.e., Finland (56%), Sweden (53%), and Denmark (48%).

In addition to the direct incineration of the residual waste, the Max. Net Present Value scenario also implements the following changes: (1) the existing MRFs for packaging waste are retrofitted in the first period, thus increasing materials recovery and revenues, (2) a new MRF for packaging waste with 25,000 tonnes/year capacity is installed in the period 2030-2035 (as in the BAU), (3) the existing AD facilities are decommissioned due to their high costs compared with composting, (4) the existing composting facility is retrofitted to receive the organic waste collected separately (currently it receives residual organic waste), and (5) a new landfill with a capacity for 2.8 million tonnes of waste is opened in the second period.

Min. Climate Change impacts scenario

The Min. Climate Change scenario achieves 10% landfill by 2035. This scenario involves technological choices that maximize recycling due to the large GHG emissions avoided through the substitution of primary materials. Thus, this scenario exhibits the highest recycling rate, with 51% by 2035. The existing MRFs for residual and packaging waste are retrofitted from the start to maximize materials recovery. The existing incineration facility is replaced in the first period with a new large incineration facility with 450,000 tonnes/year capacity. This substitution is motivated by the lower climate change impacts of the new facility due to its higher net electrical efficiency and the recovery of non-ferrous metal (aluminium) from the bottom ash.

The minimization of the climate change impacts reveals an interesting behavior of organic waste treatment. The organic waste collected separately is treated through the existing AD facility up to the period 2030-2035 when it is diverted to the existing composting facility. This switch from AD to composting is motivated by the ever more stringent landfilling restrictions. In fact, the organic waste collected separately is diverted away from AD due to the relatively high amount of rejects produced during the pretreatment (up to 27% of the incoming organic waste). This flow of rejects cannot be fully diverted to landfill because of the landfilling targets and thus put additional pressure on the incineration capacity. Conversely, composting does not generate pretreatment rejects and produces compost and a stable material that does not contribute to the landfill target, as established in Directive 2018/850. Overall, the organic waste collected separately is diverted from AD to composting when approaching 2035 to comply with the 10% landfill target.

Surprisingly, the residual organic waste shows the opposite trend, i.e., a switch from composting to AD. This stream is treated through composting until 2030-2035, when it is diverted to AD. Since the existing AD facility dedicated to residual organic waste is decommissioned from the start, a new facility with 173,862 tonnes/year capacity and equipped with biomethane upgrading is installed in the period 2030-2035. This change is motivated by the landfilling targets. It is important to note that the compost produced from the residual organic waste cannot be used on land. In consequence, composting of residual organic waste cannot be listed as a recovery option and all the output flows must be disposed of in landfill, thus contributing to the landfill target. Conversely, the AD of the residual organic is listed as a recovery option because of energy recovery. Therefore, the output flows from AD that end up in landfills does not contribute to the landfill target.

Cost and carbon footprint of the optimal MSW treatment network

Figure 5-4 displays the breakdown of the costs and carbon footprint of the optimal MSW treatment network in each scenario. In the BAU, the average cost and carbon footprint are 17 EUR/tonne MSW and 200 kg CO_2 eq/tonne MSW, respectively. Both indicators remain stable over the decision time horizon. The top contributors to the cost are MRFs (32% of net cost), AD (29%), and incineration (24%). The carbon footprint is dominated by incineration and landfilling, while MRFs (including recycling) exhibit a net avoided impact. AD and composting have a negligible contribution.

The Max. Net Present Value scenario saves 13 EUR/tonne MSW (78% reduction) and avoids 11 kg CO_2 eq/tonne MSW (6% decrease) over the BAU. This scenario relies heavily on incineration, which makes up for the biggest part of the cost incurred, chiefly due to the operating costs and the investment in a new facility. Incineration also dominates the carbon footprint, whereas the contribution of landfilling decreases substantially due to its low utilization (5% landfill rate by 2035).

Finally, the Min. Climate Change scenario avoids 193 kg CO_2 eq/tonne MSW (95% reduction) at the expense of increasing the cost by 6 EUR/tonne MSW (35% increase) over the BAU. In this scenario, MRFs dominate the net cost (32%) due to the investment required for retrofitting. Other important contributors are AD (24%) and incineration (23%). The carbon footprint approaches zero in this scenario, chiefly due to the increase in recycling. Thus, the climate change impacts are dominated by the avoided impacts from recycling.

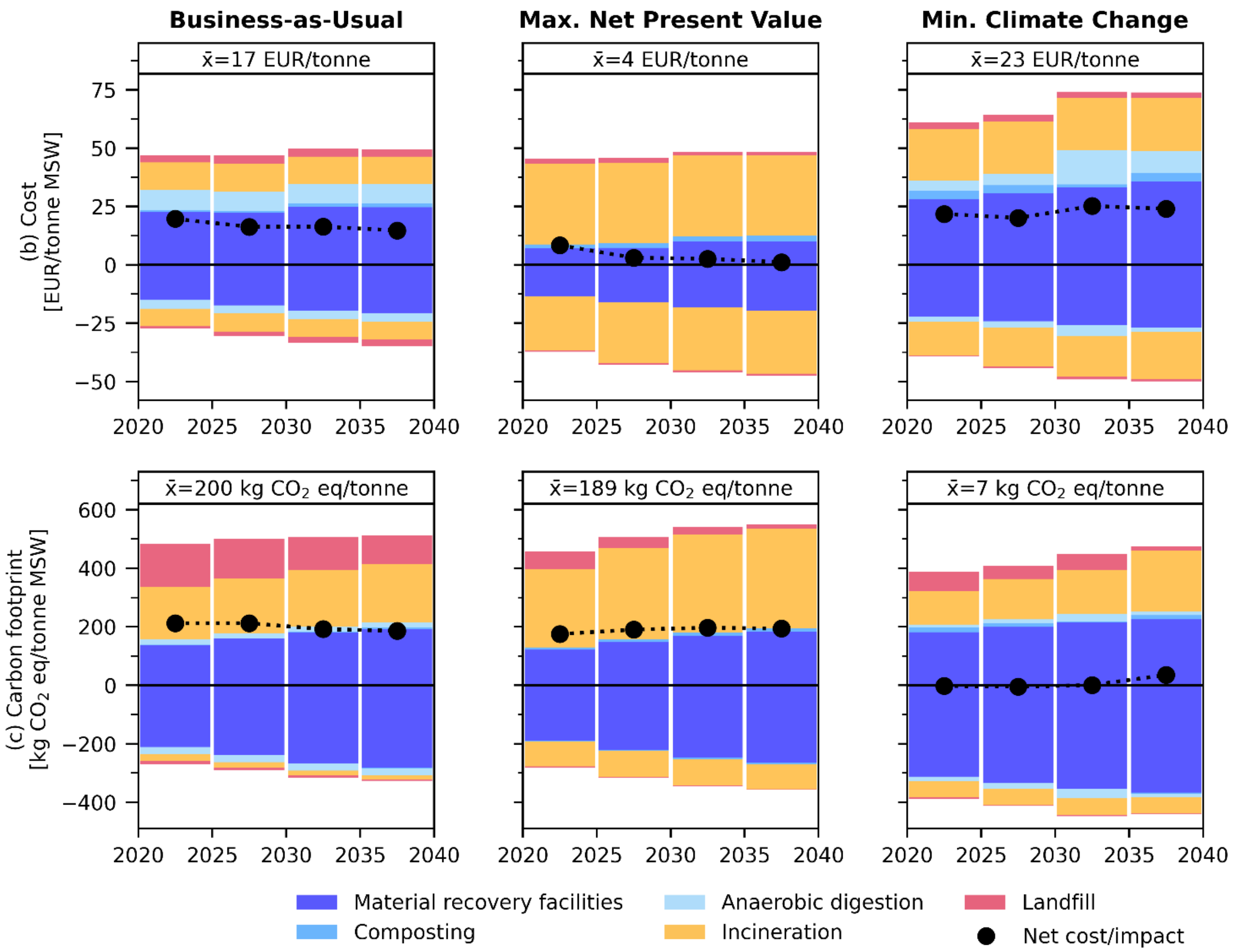

Figure 5-4. Breakdown of costs (a) and carbon footprint (b) of the optimal municipal solid waste treatment network in the Business-as-Usual, Max. Net Present Value, and Min. Climate Change scenarios. Positive values represent incurred costs/impacts, while negative values represent revenues/avoided impacts. The text boxs show the averages over 2020-2040. The economic data is harmonized to EUR2019. Material recovery facilities include the impacts of recycling activities as well as the avoided impacts from primary materials substitution.

5.3.4 Long-term analysis of anaerobic digestion

AD is utilized in the BAU and Min. Climate Change scenarios but not in the Max. Net Present Value scenario due to its higher cost compared with composting, which is preferred. Furthermore, biomethane upgrading is preferred over electricity generation in both scenarios. In this context, Figure 5-5 displays the breakdown of AD costs and carbon footprint in the BAU and Min. Climate Change scenarios. The performance of AD is assessed separately for the organic waste collected separately (panels 'A' in Figure 5-5) and the residual organic waste (panels 'B' in Figure 5-5) since these streams have different properties and they are treated by different facilities.

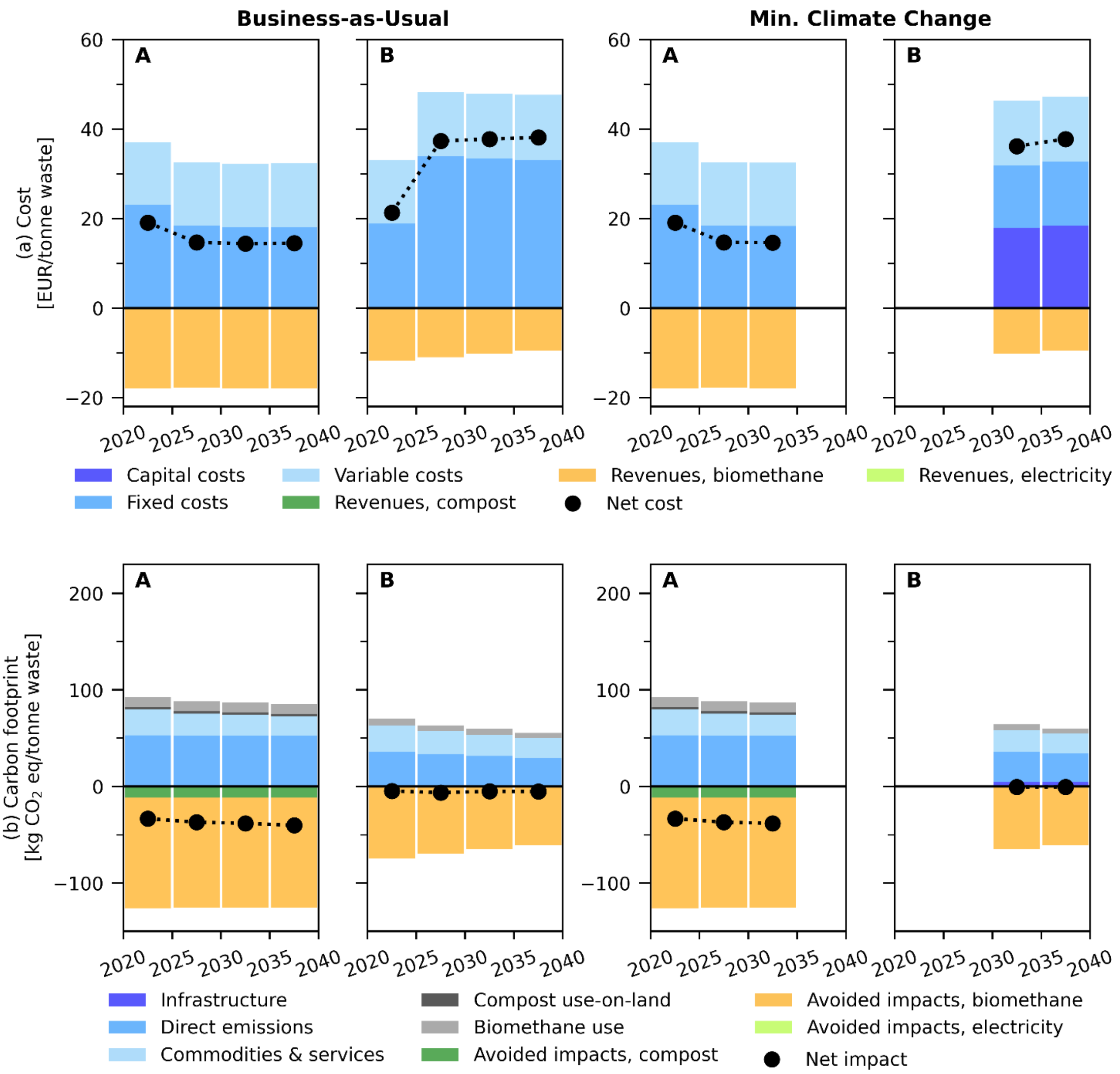

Figure 5-5. Breakdown of costs (a) and carbon footprint (b) of anaerobic digestion in the Business-as-Usual and Min. Climate Change scenarios. 'A' panels show results for anaerobic digestion of organic waste collected separately, while 'B' panels show results for anaerobic digestion of residual organic waste. The economic data is harmonized to EUR2019.

The net cost of AD of organic waste collected separately ranges from 19 EUR/tonne in the first period to 15 EUR/tonne in the last period. The net cost decreases over time due to the higher utilization of AD capacity as separate collection increases. Furthermore, the net cost is the same in both scenario as it was assumed that the composition of the organic waste collected separately remains constant. The fixed operating costs account for 62% of the incurred costs, while the variable operating costs account for the remaining 38%. Revenues are obtained from the selling of biomethane, as the compost was assumed to be given away to farmers for free following common practice (Cobo et al., 2019). The selling of biomethane compensate up to 48% of the incurred costs based on the assumption that the biomethane is sold at 0.42 EUR/m^3 or 45 EUR/MWh (Hogg et al., 2015). The net cost resulting from the AD of residual organic waste is comparably higher, ranging from 21 to 38 EUR/tonne in the BAU (first to last period). The net cost increases over time due to changes in the composition of the residual organic waste. As demonstrated in Chapter 3, the BMP of the residual organic waste decreases over time and so do the revenues from the selling of biomethane.

AD of organic waste collected separately results in a net negative carbon footprint ranging from -13 to -14 kg CO_2 eq/tonne in the first and last period, respectively. This means that the impacts avoided through the substitution of natural gas and mineral fertilizers counterbalance the impacts generated. The production of biomethane and the subsequent substitution of natural gas dominates the avoided impacts. The impacts generated by AD come primarily from direct emissions, i.e., fugitive emissions from digesters and upgrading. The AD of residual organic waste also results in a net negative carbon footprint, even though it is close to neutrality. This difference is mainly due to the lower methane potential associated with the residual organic waste.

5.3.5 Long-term analysis of incineration

Incineration arises as a fundamental part of the MSW treatment network across all the scenarios. However, the waste streams directed to incineration and their quantities differ among scenarios due to different designs of the MSW treatment network. For example, incineration receives rejects from MRFs in the BAU scenario, while the residual waste is directly incinerated in the Min. Cost scenario. These changes have large implications for the composition and energy content of the incinerated waste and, in consequence, for the economic and climate performance of incineration.

As it can be observed in Figure 5-6, the waste incinerated in the BAU contains a higher share of plastic and a lower share of organic waste compared with the other two scenarios. This is because the BAU scenario uses MRFs that separate the organic matter, thus reducing its presence in the rejects directed to incineration. In the Max. Net Present Value scenario, the residual waste is directly fed to incineration without separating the organic matter. On the other hand, the Min. Climate Change scenario uses retrofitted MRFs that separate both organic matter and plastic with a higher efficiency.

The composition of the incinerated waste changes significantly over the time across all the scenarios. Over 2020-2040, the plastic content increases from 24% to 29% in the BAU, from 14% to 20% in the Max. Net Present Value scenario, and from 12% to 20% in the Min. Climate Change scenario. Meanwhile, there is a decrease over time in the share of biogenic-based combustible materials, especially paper, cardboard, and textile. These trends are largely explained by the influence of separate collection. Here, it was assumed that the rate of separate collection achieves 90% for paper/cardboard, 85% for (plastic and metal) packaging waste, and 60% for textile by 2035-2040. However, while the separate collection of biogenic-based combustible materials (paper, cardboard, and textile) is highly efficient, a large share of the plastic packaging waste continues to be disposed in the residual waste stream.

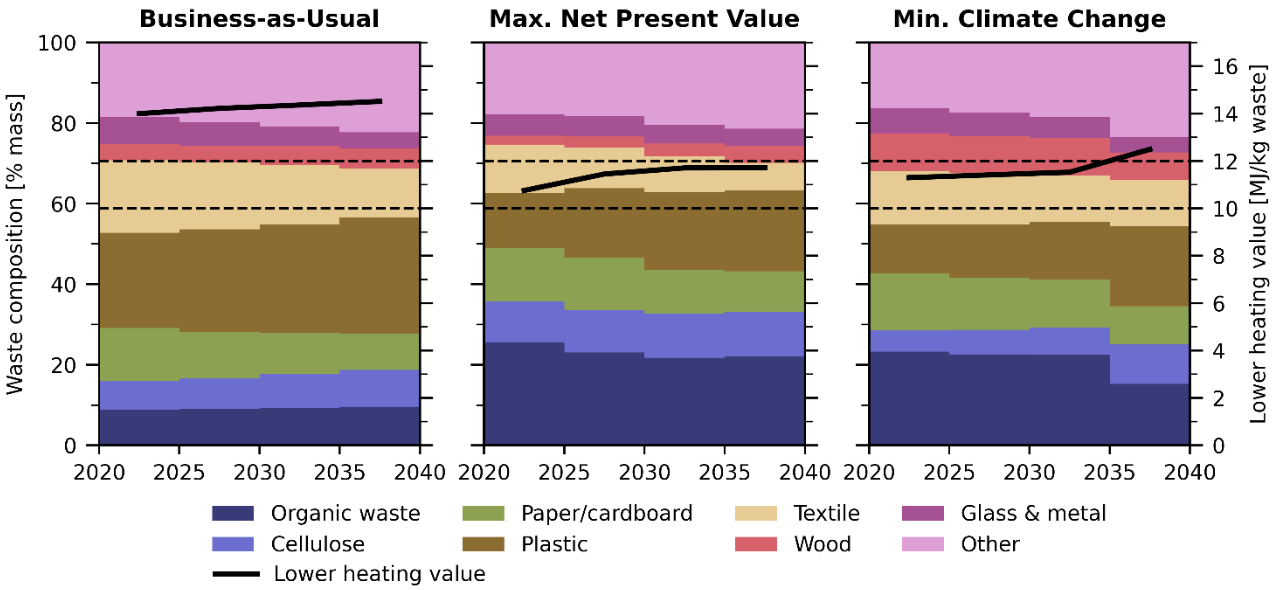

Figure 5-6. Composition (stacked area and left y-axis) and lower heating value (solid line and right y-axis) of the input waste to incineration in each period in the Business-as-Usual, Max. Net Present Value, and Min. Climate Change scenarios. Dotted horizontal lines indicate the typical lower heating value range of waste received at incineration facilities in developed countries (Reimann, 2012). The 18 waste materials considered in the model were aggregated into more generic categories to facilitate interpretation.

The LHV of the waste fed to incineration is not particularly affected by the increase in separate collection and recycling. The LHV slightly increases over time from 14.0 to 14.5 MJ/kg waste in the BAU, from 10.8 to 11.7 MJ/kg in the Max. Net Present Value scenario, and from 11.3 to 12.5 MJ/kg in the Min. Climate Change scenario. This increase is explained by the increased share of plastic in the waste fed to incineration, as mentioned above.

The average gross cost of incineration over the decision horizon (i.e., the cost without accounting for the revenues from the sale of electricity and metals) ranges from 53 EUR/tonne in the BAU to 67 and 86 EUR/tonne in the Max. Net Present Value and Min. Climate Change scenarios, respectively (Figure 5-7a). The Max. Net Present Value and Min. Climate Change scenarios show a higher gross cost due to the additional capital costs of the new incineration facility (192 million EUR). The average net cost (i.e., the cost after considering the revenues) ranges from 20 EUR/tonne in the BAU and Max. Net Present Value scenarios to 25 EUR/tonne in the Min. Climate Change scenario. As can be observed, the three scenarios show a similar net cost despite the large investment in the Max. Net Present Value and Min. Climate Change scenarios. The rationale behind this is that these two scenarios generate on average 31% (Max. Net Present Value) and 71% (Min. Climate Change) more electricity per tonne of waste than the BAU due to the higher efficiency of the new facility. Thus, the sale of electricity compensates up to 64% of the gross cost in the Max. Net Present Value scenario and 67% in the Min. Climate Change scenario.

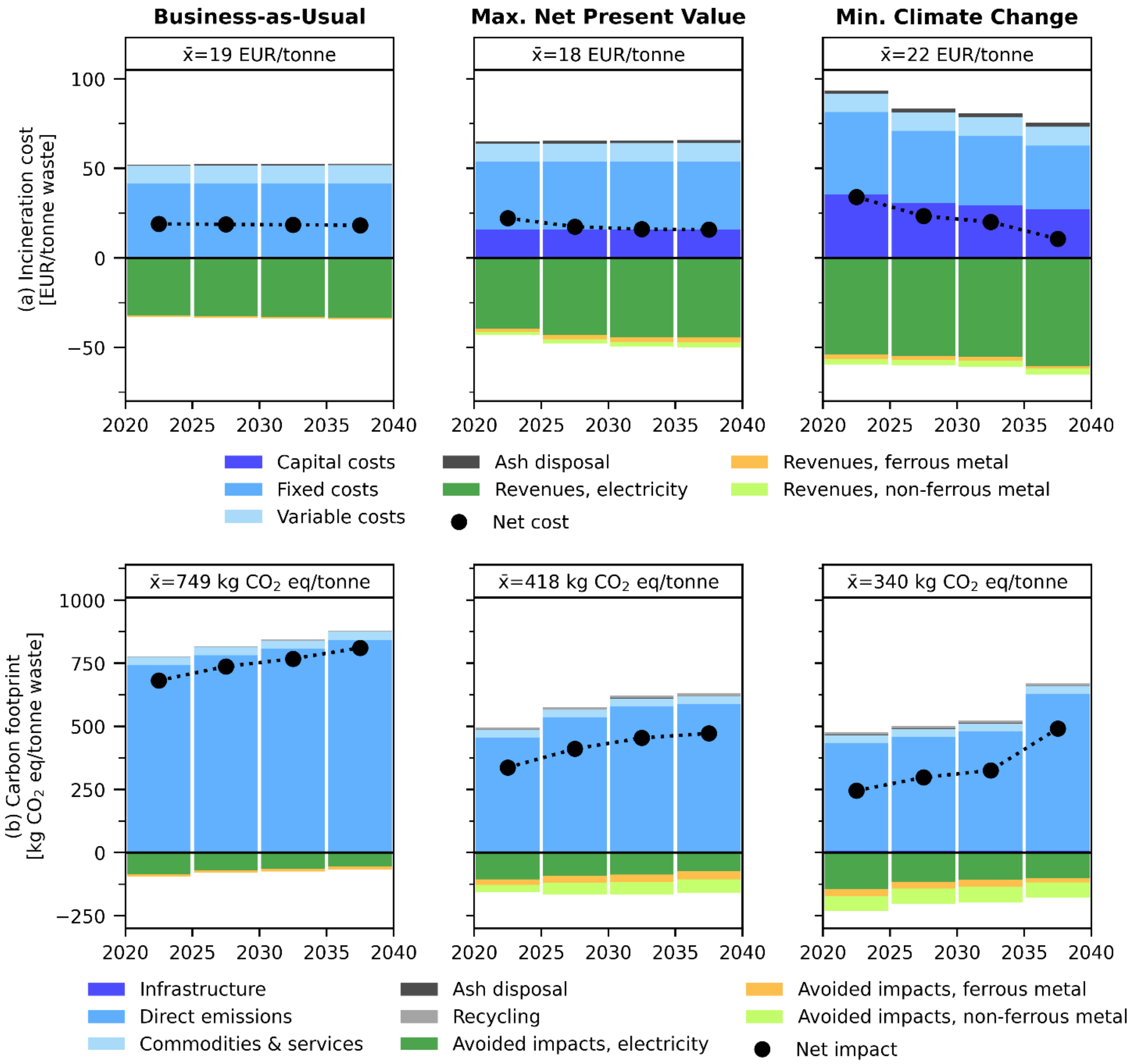

Figure 5-7. Breakdown of costs (a) and carbon footprint (b) of incineration in the Business-as-Usual, Max. Net Present Value, and Min. Climate Change scenarios. The text box shows the averages over 2020-2040. The economic data is harmonized to EUR2019.

The cost of incineration is not particularly affected by the improvement of separate collection and recycling. The net cost remains constant over 2020-2040 in the BAU (at about 20 EUR/tonne), while in the other two scenarios it decreases 25% in the Max. Net Present Value (from 24 to 18 EUR/tonne) and 67% in the Min. Climate Change scenario (from 37 to 13 EUR/tonne). This decrease in costs is primarily explained by the increase in the waste energy content (which increases the production of electricity). In the Min. Climate Change scenario, the cost also decreases due to a higher autilization of the incineration capacity at the end of the decision horizon.

The average carbon footprint of incineration over the decision horizon ranges from 749 kg CO_2 eq/tonne in the BAU to 418 kg CO_2 eq/tonne in the Max. Net Present Value scenario (44% reduction over the BAU) and 340 kg CO_2 eq/tonne in the Min. Climate Change scenario (55% reduction over the BAU) (Figure 5-7b). The BAU shows the highest carbon footprint largely due a higher direct GHG emissions caused by the higher share of plastic in the waste fed to incineration. Additionally, the new incineration facility installed in the Max. Net Present Value and Min. Climate Change scenarios allows reducing impacts due to its higher electrical efficiency and the recovery of aluminum from the bottom ash. The impacts avoided through electricity substitution are reduced because of the increased share of renewables in the Spanish electricity mix (carbon footprint decreases from 188 to 117 kg CO_2 eq/MWh over 2020-2040). In this context, the potential climate benefits of recovering aluminum from the bottom ash become more pronounced since aluminium recycling avoids up to 18,049 kg CO_2 eq/tonne material.

The carbon footprint of incineration increases over time, from 682 to 811 kg CO_2 eq/tonne in the BAU (19% increase), from 337 to 472 kg CO_2 eq/tonne in the Max. Net Present Value scenario (40% increase), and from 245 to 491 CO_2 eq/tonne in the Min. Climate Change scenario (100% increase). Figure 5-8 displays the time development of the carbon footprint as a function of three major variables, namely the fossil carbon content of waste (Figure 5-8a), the electricity production rate (Figure 5-8b), and the carbon footprint of the Spanish electricity mix (Figure 5-8c). These results show that the carbon footprint of incineration increases over time largely because of an increase in the fossil carbon content in the incinerated waste. Over 2020-2040, the fossil carbon content increases 16% in the BAU, 26% in the Max. Net Present Value scenario, and 44% in the Min. Climate Change scenario due to a higher share of plastic and a lower share of paper/cardboard and textile (predominantly biogenic). While the higher share of plastic allows increasing the electricity production rate, the potential climate benefits from electricity substitution are not sufficient to counterbalance the impact of direct emissions. This is partially explained by the low carbon footprint of the Spanish electricity mix.

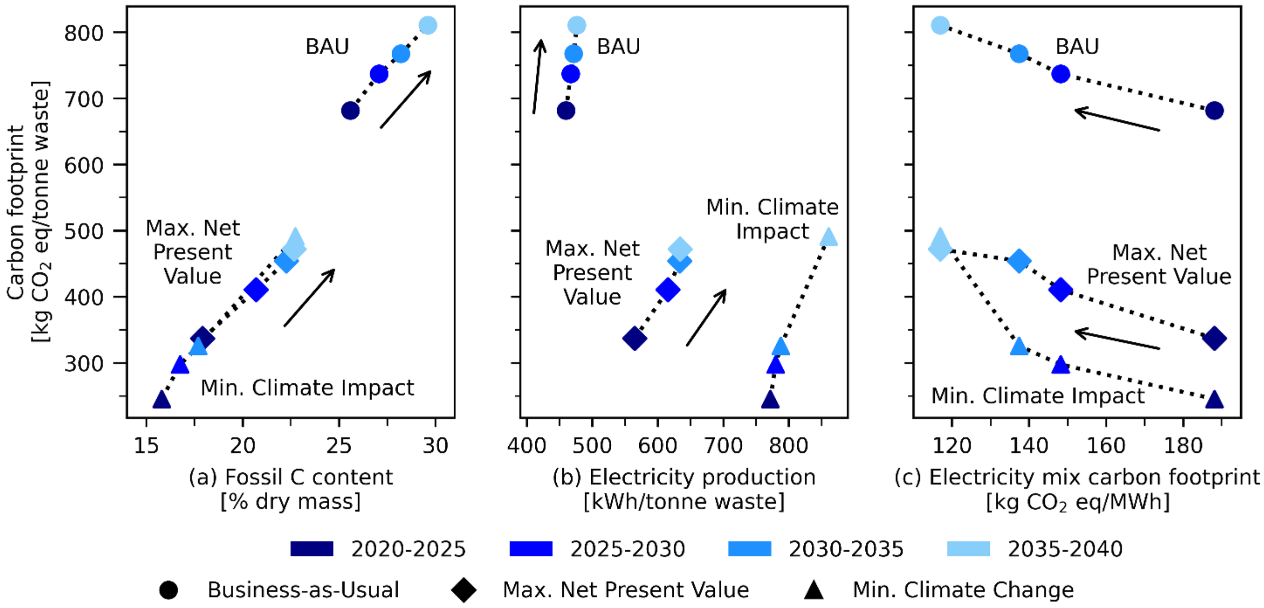

Figure 5-8. Carbon footprint of incineration as a function of (a) the fossil carbon content of waste, (b) the electricity production rate, and (c) the carbon footprint of the electricity mix. Marker colors indicate periods, arrows indicate the evolution with time, and marker style indicates scenarios.

5.4 Final remarks and future work

This final chapter has addressed the economic and climate optimal WTE pathways that could be prioritized in the future (**RQ3**). A multi-period optimization framework was developed to enable a prospective analysis of the capacity, use, economics, and carbon footprint of WTE pathways considering the availability and composition of waste, system constraints, policy targets, and changes in the background system (i.e., the decarbonization of the electricity mix).

The capabilities of this framework have been demonstrated through the case study of Madrid. It was found that achieving 10% landfill by 2035 (as established in the Landfill Directive) demands a more intensive use of incineration. While the results from Chapter 4 already suggest this trend, the use of the optimization framework (instead of a descriptive MFA and LCA framework) allowed revealing possible pathways to achieve this target. Thus, about 54% of the MSW generated over 2020-2040 should be incinerated according to the maximum net present value scenario and 30% according to the minimum climate change impacts scenario, compared with 23% in the business-as-usual scenario. In all scenarios, one or two large-scale incineration facilities (≥ 300,000 tonnes/year of capacity) are needed to reach these rates. Consequently, these results suggest that it is unlikely that separate collection and recycling will cause a shortage in feedstock availability for incineration, which is in agreement with the energy recovery potential estimated in Chapter 3.

The optimization framework presented in this thesis is suitable to prospectively assess the economic and climate performance of WTE technologies. In this regard, the results revelead that the cost of incineration is not particularly affected by future changes in the MSW management system. In fact, changes in the composition of waste and the investment in more efficient facilities would boost the electricity production rate and the revenues from the sale of electricity, which could make incineration even more profitable. This has large implications for the waste management sector since the relatively low costs of incineration is often claimed to prevent the development circular economy businesses. Moreover, the lack of sufficient economic competitiveness is often identified as a major barrier to the replacement of incineration with emerging WTE thermochemical technologies such as gasification and/or pyrolysis (*cf.* Section 1.2.1).

However, the carbon footprint of incineration is likely to increase in the future, chiefly due to an increase in the share of plastic waste entering incineration and the decline in the climate benefits of electricity recovery due to the transition towards a more renewable power mix. In this context, incineration could move far away from the potential climate benefits often claimed. This holds particularly true for existing and new incineration facilities dedicated to electricity production only, the typical situation in Southern Europe. The prospective economics and environmental performance of alternative technological pathways, such as combined heat and power, co-incineration, and incineration coupled with carbon capture, have still to be addressed.

The optimization framework developed in this thesis introduces several novelties, the most relevant being the nonlinear reponse of waste treatment processes to changes in waste composition. There are many ways in which the framework could be improved. The current version covers only climate change impacts, thus disregarding other highly relevant impacts such as those related to human health. There are several reasons behind this decision. First and foremost, the size of the optimization framework would increase significantly if other non-GHG emissions were included. The LCA framework developed in Chapter 4 includes 106 elementary flows (i.e., emissions to air, water, and soil). This would mean 106 new variables per waste treatment facility. The second reason is that, as observed in Chapter 4, the impacts on human health are highly uncertain, whereas the impact assessment method for climate change is well-developed and world-wide agreed. In any case, the implementation of other impact categories should be explored in future research.

Separate collection has been modelled exogenously. This means that the separate collection stage is not optimized but modelled with the approach developed in Chapter 3 and provided to the optimization model as an input parameter. The main reason behind this decision is related to the complexity of modelling the behaviour of people with respect to waste separation. For example, an optimized solution could suggest to separately collect the plastic waste without impurities as this would maximize the recovery of materials and the benefits from recycling. However, it is known from Chapter 3 that the waste streams collected separately do not contain only the target materials but many other unwanted materials. Future research should focus on developing new models to account for the inefficiencies during waste separation at households.

The technology portfolio should certainly be enriched to cover emerging WTE technologies, such as gasification, pyrolysis, hydrothermal carbonization, hydrothermal liquefaction, or ethanol fermentation. These technologies are likely to play an increasingly important role over the next decades and should be considered. However, the development of predictive LCIs for these technologies represents a major barrier as they involve more complex reactions compared with standard WTE technologies (i.e., AD and incineration). Thus, the simple approach adopted throughout this thesis that relates emissions to the physicochemical properties of waste materials may not be suitable to model these emerging technologies. In fact, the LCIs models of these technologies may require nonlinear relationships (Sardarmehni and Levis, 2021) that would significantly increase the size and complexity of the optimization framework. Finally, it is worth noting that the techno-economic characterization of these emerging technologies is also a challenging task since most of them are still in their infancy regarding commercial implementation.

The scope of the optimization framework could be expanded in several ways. The first possibility is to include waste streams available outside the system boundaries. This may include MSW streams from other municipalities or regions and other waste streams, such as sewage sludge, biomass (e.g., forest and agricultural residues), and industrial waste. On the one hand, the inclusion of MSW from other municipalities or regions could provide better outcomes in terms of capacity optimization as more feedstock would be available. On the other hand, previous works have revelead potential operational, economic, and environmental benefits associated with co-processing, such as the co-digestion of food waste and sewage sludge (De Clercq et al., 2017) or the co-incineration in other installations such as cement kilns (Galvez-Martos and Schoenberger, 2014). The second possibility is to include the MSW management system of other municipalities and/or regions. The difference with the first possibility is that, in this case, the waste treatment facilities located in other municipalities and/or regions are also modelled. Both possibilities result particularly relevant for the case study of Madrid since its MSW management system is completely isolated from the rest of the region. The potential benefits of such a regional integration remains to be explored.

The optimization framework is sufficiently generic and flexible to be used for other case studies. However, as usually occur in the field of industrial ecology, the availability of data becomes a major barrier. It is worth noting that the City Council of Madrid provides relatively well developed and documended data on the MSW management system (e.g., waste generation and composition and facilities mass balances). This availability of data has greatly facilitated the modelling work made in this thesis. Other smaller municipalities and/or regions may still lack the capacity of providing such a comprehensive information. Thus, the assessment would have to rely on lower data quality and larger amount of assumptions that can compromise the representativeness of the outcomes.

Chapter 6

Conclusions

The goal of this thesis was to assess the energy recovery potential of municipal solid waste (MSW) and the economic and environmental optimal waste-to-energy (WTE) pathways within the context of an increasingly circular economy. To achieve this goal, tailored tools for the systematic assessment and optimization of MSW management systems were developed and applied to the case study of Madrid. The principal results and conclusions for the research questions addressed in this thesis are presented in the following.

RQ1 What will the energy recovery potential of MSW be under future scenarios of increased separate collection and recycling?

A higher separate collection and recycling of MSW do not necessarily compromise the energy recovery potential. For the specific case study of Madrid, it has been found that the gross energy recovery potential of WTE feedstocks ranges from 12,436 TJ in 2019 to 8,942-11,327 TJ by 2040, depending on the scenario. Thus, even under the most promising scenario for recycling, the gross energy recovery potential decreases only by 28%. This relatively low impact can be attributed to inefficiencies in separate collection and the limited efficiency of materials recovery at automatized sorting facilities. Overall, the future availability and characteristics of feedstocks were found to be adequate to sustain the operation of large-scale WTE facilities. Yet, the implementation of waste prevention measures or alternative collection schemes that could further improve separate collection, such as deposit-refund or "pay-as-you-throw" (PAYT) schemes, may have an impact on the energy recovery potential and the WTE sector.

RQ2 What are the potential environmental consequences of phasing-out incineration in the MSW management system?

Phasing-out the existing incineration facility of Madrid could reduce some of the environmental impacts associated with MSW treatment, including climate change, acidification, terrestrial and freshwater eutrophication, photochemical ozone formation, human toxicity–cancer effects, and ecotoxicity. It was found that phasing-out incineration could reduce the climate change impacts despite that this decision intensifies landfilling. While this finding may seem counter-intuitive, it is largely explained by the continuous improvement of the separate collection of highly biodegradable materials, such as food waste, paper, and cardboard. Phasing-out incineration would also avoid the production and disposal of fly ashes (i.e., residues from cleaning the exhaust gases), which are a large source of impacts on human health and ecotoxicity due to their potential leacheability, and, consequently migration of heavy metals towards ground and surface water bodies. Within this context, the substitution of the existing incineration facility with a new more energy efficient facility was proved not to introduce notable differences from an environmental perspective. Despite the potential benefits in terms of the impact categories assessed in this thesis, it is worth noting that phasing-out incineration jeopardizes the realization of the 10% landfill target by 2035. Other environmental risks associated with landfilling, such as the potential pollution of rivers and oceans with plastics, have been not investigated in this thesis. Thus, the 10% landfill goal should not be disregarded, even though achieving this goal may require structural changes in the MSW management system beyond the use or not of incineration.

RQ3 What are the economic and/or environmental optimal WTE pathways that could be prioritized in light of the future availability and characteristics of MSW?

This research question motivated the development of a multi-period optimization framework that allows determining time-dependent waste treatment capacities and flows according to economic and climate objectives and subject to waste availability and composition, system constraints and restriction, policy targets, and changes in the background system (e.g., the decarbonization of the electricity mix). The application of this framework to the case study of Madrid revealed that an optimal MSW management system that achieves 10% landfill by 2035 requires a more intensive use of incineration. While incineration could become even more economically attractive in the future, the carbon footprint is likely to increase with the potential of moving this WTE pathway far away from the climate benefits often claimed. Anaerobic digestion (AD) was found attractive to minimize the climate change impacts, but not to minimize the costs of the system, for which composting is preferred. The deployment of AD could be affected by landfilling restrictions due to the potential high amount of rejects produced during the pretreatment, especially if the separate collection of organic waste is relatively inefficient.

Based on the work presented in this thesis, envisioning a future with a relevant role for WTE appears highly likely even within the context of an increasingly circular economy. Therefore, future work should prioritize the enrichment of the technology portfolio of the optimization framework to cover emerging WTE technologies, notably gasification and pyrolysis. These technologies are likely to play an increasingly important role over the next decades and should be considered. The development of predictive life cycle inventories that account for the response of these technologies to changes in waste composition is key. Furthermore, the following two modelling aspects deserve special attention given their high impact on the energy recovery potential: (1) improving the projection of MSW generation with more complex forecasting methods and (2) a more exhaustive modelling of separate collection, including the consideration of alternative collection schemes.

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Appendices

Appendix A: Waste materials properties

Table A-1. Biological and physicochemical properties of waste materials based on average values from the literature.

Material	**Moisture content** (% wet mass)		**Lower heating value (LHV)** (MJ/kg dry mass)	
	Value	**Source**	**Value**	**Source**
Food waste	69	*1, 2, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, 16, 17, 18, 19, 20, 24*	15	*3, 4, 8, 10, 12, 16, 20, 23*
Green waste	58	*2, 5, 7, 8, 10, 11, 12, 14, 15, 17, 20, 24*	14	*3, 4, 8, 10, 12, 20, 23*
Cellulose	60	*6, 7, 8, 20, 24*	18	*4, 8, 20*
Paper	11	*2, 5, 6, 8, 10, 11, 12, 15, 16, 17, 19, 20, 24*	14	*3, 4, 8, 10, 12, 16, 20, 23*
Cardboard	11	*5, 6, 7, 8, 12, 15, 16, 17, 19, 20, 24*	15	*3, 4, 8, 12, 16, 20, 23*
PET	6	*8, 18, 20, 24*	27	*4, 8, 20, 23*
HDPE	6	*6, 8, 18, 20, 24*	42	*4, 8, 20, 23*
LDPE	22	*6, 8, 18, 20, 24*	34	*4, 8, 20, 23*
PP	6	*8, 18, 20, 24*	41	*4, 23*
Other plastic packaging	8	*1, 8, 10, 11, 12, 16, 18, 19, 20, 24*	28	*4, 8, 10, 16, 20, 23*
Non-packaging plastic	5	*1, 8, 10, 11, 12, 16, 18, 19, 20, 24*	31	*4, 8, 10, 16, 20, 23*
Cartons and alike	15	*6, 8, 18, 20*	20	*3, 4, 8, 20*
Glass	2	*2, 5, 8, 12, 16, 17, 18, 19, 20, 24*	0	*3, 4, 8, 12, 16, 20*
Ferrous metal	4	*2, 5, 8, 12, 16, 17, 18, 19, 20, 24*	0	*3, 4, 8, 12, 16, 20*
Non-ferrous metal	3	*2, 5, 8, 12, 16, 17, 18, 19, 20, 24*	0	*3, 4, 8, 12, 16, 20*
Textile	14	*2, 5, 6, 8, 10, 11, 12, 15, 17, 18, 19, 20, 24*	19	*4, 8, 20, 23*
Wood	12	*5, 8, 10, 11, 15, 16, 17, 18, 19, 20, 24*	17	*4, 8, 16, 20, 23*
Others	15	*2, 5, 7, 8, 12, 19, 20, 24*	8	*4, 8, 20*

Source: 1: IEA Bioenergy (2018a); 2: Milutinović et al. (2017); 3: Götze et al. (2016a); 4: Götze et al. (2016b); 5: Jeswani and Azapagic (2016); 6: Krause et al. (2016); 7: Naroznova et al. (2016); 8: Turner et al. (2016); 9: Styles et al. (2016); 10: Hla and Roberts (2015); 11: Arafat and Jijakli (2013); 12: Minoglou and Komilis (2013); 13: Andersen et al. (2011); 14: Boldrin et al. (2011); 15: US EPA (2011); 16: Giugliano et al. (2011); 17: Lagerkvist et al. (2011); 18: Montejo et al. (2011); 19: Cherubini et al. (2009); 20: Riber et al. (2009); 21: Staley and Barlaz (2009); 22: Stichnothe and Azapagic (2009); 23: US EIA (2007); 24: IPCC (2006c); 25: US EPA (2003); 26: Eleazer et al. (1997)

Table A-1. *(continued)*

Material	Volatile solids (VS) content (% dry mass)		Biochemical methane potential (BMP) (m^3 CH_4/tonne VS)		Methane content of biogas (% volume biogas)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	83	*1, 3, 4, 6, 7, 8, 10, 11, 20, 25*	390	*1, 7, 26*	58	*Buswell equation*
Green waste	77	*3, 4, 7, 10, 11, 20, 25*	145	*7, 21*	52	*Buswell equation*
Cellulose	90	*4, 6, 7, 20*	209	*7*	53	*Buswell equation*
Paper	84	*3, 4, 6, 7, 8, 10, 11, 20, 25*	267	*7, 26*	56	*Buswell equation*
Cardboard	86	*3, 4, 6, 7, 8, 10, 11, 20, 25*	284	*7, 26*	57	*Buswell equation*
PET	93	*4, 8, 20*	0	–	0	–
HDPE	98	*4, 20*	0	–	0	–
LDPE	94	*4, 20*	0	–	0	–
PP	96	*4*	0	–	0	–
Other plastic packaging	94	*4*	0	–	0	–
Non-packaging plastic	95	*3, 4, 8, 10, 11, 20*	0	–	0	–
Cartons and alike	94	*3, 4, 6, 8, 20*	211	*7, 26*	58	*Buswell equation*
Glass	1	*3, 4, 8, 20*	0	–	0	–
Ferrous metal	1	*3, 4, 20*	0	–	0	–
Non-ferrous metal	3	*3, 4, 20*	0	–	0	–
Textile	87	*4, 6, 8, 10, 11, 20*	225	*21*	59	*Buswell equation*
Wood	88	*4, 8, 10, 11, 20*	82	*21*	52	*Buswell equation*
Others	35	*4, 8, 20*	0	–	0	–

Table A-1. *(continued)*

Material	**Carbon content** (% dry mass)		**Biogenic carbon** (% total carbon)		**Hydrogen content** (% dry mass)		**Oxygen content** (% dry mass)		**Nitrogen content** (% dry mass)		**Sulfur content** (% dry mass)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	45	*4*	45	*4*	7.30	*4*	33	*4*	3.19	*4*	0.61	*4*
Green waste	36	*4*	36	*4*	5.26	*4*	34	*4*	2.13	*4*	0.40	*4*
Cellulose	43	*4*	43	*4*	6.85	*4*	47	*4*	1.12	*4*	0.27	*4*
Paper	39	*4*	39	*4*	6.08	*4*	35	*4*	0.31	*4*	0.07	*4*
Cardboard	41	*4*	41	*4*	6.89	*4*	40	*4*	0.30	*4*	0.11	*4*
PET	64	*4*	64	*4*	5.03	*4*	28	*4*	0.16	*4*	0.06	*4*
HDPE	83	*4*	83	*4*	15	*4*	2	*4*	0.16	*4*	0.09	*4*
LDPE	80	*4*	80	*4*	14	*4*	2	*4*	0.71	*4*	0.08	*4*
PP	81	*4*	81	*4*	14	*4*	2	*4*	0.26	*4*	0.03	*4*
Other plastic packaging	78	*4*	78	*4*	10	*4*	2	*4*	0.42	*4*	0.04	*4*
Non-packaging plastic	83	*4*	83	*4*	12	*4*	2	*4*	1.64	*4*	0.08	*4*
Cartons and alike	44	*4*	44	*4*	6.20	*4*	35	*4*	0.27	*4*	0.09	*4*
Glass	1	*4*	1	*4*	0.14	*4*	1	*4*	0.04	*4*	0.08	*4*
Ferrous metal	3	*4*	3	*4*	0.24	*4*	2	*4*	0.11	*4*	0.02	*4*
Non-ferrous metal	5	*4*	5	*4*	0.76	*4*	1	*4*	0.21	*4*	0.02	*4*
Textile	53	*4*	53	*4*	6.92	*4*	34	*4*	2.10	*4*	0.63	*4*
Wood	48	*4*	48	*4*	6.76	*4*	46	*4*	1.17	*4*	0.42	*4*
Others	14	*4*	14	*4*	1.85	*4*	14	*4*	0.48	*4*	0.41	*4*

Table A-1. *(continued)*

Material	Phosphorus content (% dry mass)		Potassium content (% dry mass)		Chlorine content (% dry mass)		Fluorine content (% dry mass)		Bromine content (% dry mass)		Antimony content (% dry mass)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	7.00E-01	*4*	7.84E-01	*4*	7.83E-01	*4*	2.53E-03	*4*	1.00E-03	*4*	1.80E-05	*4*
Green waste	1.34E+00	*4*	8.70E-01	*4*	2.97E-01	*4*	4.35E-03	*4*	1.28E-03	*4*	1.18E-04	*4*
Cellulose	6.43E-02	*4*	2.44E-01	*4*	4.57E-01	*4*	1.22E-03	*4*	1.17E-03	*4*	4.57E-05	*4*
Paper	1.89E-02	*4*	8.10E-02	*4*	8.26E-02	*4*	3.03E-03	*4*	1.03E-03	*4*	9.79E-06	*4*
Cardboard	2.27E-02	*4*	8.39E-02	*4*	9.16E-02	*4*	2.54E-03	*4*	1.49E-03	*4*	1.97E-05	*4*
PET	4.70E-03	*4*	2.04E-02	*4*	1.39E-01	*4*	1.00E-03	*4*	1.00E-03	*4*	2.63E-02	*4*
HDPE	8.71E-03	*4*	2.00E-02	*4*	5.86E-02	*4*	1.04E-03	*4*	1.00E-03	*4*	3.38E-04	*4*
LDPE	3.12E-02	*4*	6.39E-02	*4*	4.57E-01	*4*	3.80E-03	*4*	1.00E-03	*4*	7.20E-04	*4*
PP	2.68E-02	*4*	3.22E-02	*4*	7.58E-02	*4*	1.51E-03	*4*	1.00E-03	*4*	5.68E-04	*4*
Other plastic packaging	2.94E-02	*4*	3.97E-02	*4*	9.65E-01	*4*	1.76E-03	*4*	1.15E-02	*4*	4.08E-03	*4*
Non-packaging plastic	2.19E-03	*4*	2.08E-02	*4*	6.27E-01	*4*	3.59E-03	*4*	3.99E-03	*4*	1.55E-03	*4*
Cartons and alike	1.87E-02	*4*	6.97E-02	*4*	1.05E-01	*4*	2.19E-03	*4*	1.36E-03	*4*	5.40E-03	*4*
Glass	1.25E-02	*4*	5.18E-01	*4*	8.04E-03	*4*	3.92E-03	*4*	1.00E-03	*4*	2.14E-03	*4*
Ferrous metal	1.82E-02	*4*	4.64E-02	*4*	1.80E-01	*4*	1.07E-03	*4*	1.00E-03	*4*	5.15E-04	*4*
Non-ferrous metal	1.74E-02	*4*	6.08E-02	*4*	1.65E-01	*4*	1.93E-03	*4*	1.00E-03	*4*	8.58E-05	*4*
Textile	3.78E-02	*4*	9.93E-02	*4*	6.57E-01	*4*	3.10E-03	*4*	3.70E-03	*4*	5.31E-03	*4*
Wood	1.21E-02	*4*	8.34E-02	*4*	6.65E-02	*4*	1.00E-03	*4*	1.00E-03	*4*	9.72E-05	*4*
Others	2.74E-01	*4*	1.87E+00	*4*	5.88E-01	*4*	2.10E-02	*4*	1.00E-03	*4*	2.94E-04	*4*

Table A-1. *(continued)*

Material	Arsenic content (% dry mass)		Barium content (% dry mass)		Beryllium content (% dry mass)		Cadmium content (% dry mass)		Cobalt content (% dry mass)		Copper content (% dry mass)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	2.62E-04	*4*	2.28E-03	*4*	1.92E-04	*4*	7.27E-06	*4*	3.07E-04	*4*	1.09E-03	*4*
Green waste	6.60E-04	*4*	1.11E-02	*4*	1.57E-04	*4*	5.34E-05	*4*	1.33E-04	*4*	3.99E-03	*4*
Cellulose	1.81E-04	*4*	1.19E-03	*4*	1.45E-04	*4*	5.51E-06	*4*	3.42E-05	*4*	1.14E-03	*4*
Paper	1.83E-04	*4*	2.97E-03	*4*	1.15E-04	*4*	2.00E-05	*4*	9.13E-05	*4*	4.72E-03	*4*
Cardboard	1.51E-04	*4*	9.55E-03	*4*	1.75E-04	*4*	1.73E-05	*4*	2.90E-04	*4*	4.00E-03	*4*
PET	2.15E-04	*4*	2.47E-03	*4*	2.19E-05	*4*	4.56E-05	*4*	5.62E-04	*4*	1.13E-03	*4*
HDPE	2.34E-04	*4*	9.92E-03	*4*	2.90E-04	*4*	4.48E-05	*4*	5.86E-05	*4*	5.90E-04	*4*
LDPE	2.92E-04	*4*	2.81E-02	*4*	3.26E-05	*4*	1.29E-04	*4*	1.24E-04	*4*	3.89E-03	*4*
PP	2.53E-04	*4*	3.71E-03	*4*	1.95E-04	*4*	2.94E-04	*4*	5.39E-04	*4*	2.36E-03	*4*
Other plastic packaging	2.37E-04	*4*	5.83E-03	*4*	7.55E-05	*4*	8.23E-04	*4*	1.25E-04	*4*	2.10E-03	*4*
Non-packaging plastic	3.20E-04	*4*	5.06E-02	*4*	2.24E-05	*4*	4.90E-05	*4*	5.55E-04	*4*	1.12E-02	*4*
Cartons and alike	1.65E-04	*4*	7.72E-03	*4*	1.41E-04	*4*	2.34E-05	*4*	3.45E-04	*4*	9.35E-03	*4*
Glass	2.00E-02	*4*	1.17E-01	*4*	1.52E-04	*4*	5.04E-05	*4*	4.73E-04	*4*	4.46E-03	*4*
Ferrous metal	1.81E-03	*4*	3.34E-02	*4*	1.34E-04	*4*	9.44E-06	*4*	2.19E-03	*4*	2.06E-02	*4*
Non-ferrous metal	1.76E-04	*4*	2.05E-03	*4*	1.35E-04	*4*	2.42E-05	*4*	2.74E-04	*4*	1.23E-01	*4*
Textile	8.24E-04	*4*	2.43E-02	*4*	1.96E-04	*4*	7.36E-05	*4*	1.46E-04	*4*	2.52E-02	*4*
Wood	4.46E-03	*4*	4.51E-03	*4*	2.00E-04	*4*	3.96E-05	*4*	1.39E-04	*4*	4.70E-03	*4*
Others	1.21E-03	*4*	1.07E-01	*4*	2.25E-04	*4*	7.45E-05	*4*	2.96E-03	*4*	4.78E-03	*4*

Table A-1. *(continued)*

Material	Chromium content (% dry mass)		Lead content (% dry mass)		Mercury content (% dry mass)		Molybdenum content (% dry mass)		Nickel content (% dry mass)		Selenium content (% dry mass)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	9.29E-04	*4*	1.41E-04	*4*	7.57E-07	*4*	1.03E-04	*4*	1.13E-03	*4*	4.77E-03	*4*
Green waste	1.49E-03	*4*	8.24E-04	*4*	2.66E-06	*4*	3.34E-04	*4*	6.39E-04	*4*	6.48E-03	*4*
Cellulose	8.87E-04	*4*	3.01E-04	*4*	2.61E-06	*4*	6.77E-05	*4*	1.02E-03	*4*	4.18E-03	*4*
Paper	5.05E-04	*4*	5.51E-04	*4*	8.57E-07	*4*	9.44E-05	*4*	1.77E-04	*4*	3.54E-03	*4*
Cardboard	5.38E-04	*4*	9.39E-04	*4*	2.85E-06	*4*	7.89E-05	*4*	2.11E-04	*4*	3.51E-03	*4*
PET	8.66E-04	*4*	1.41E-04	*4*	2.40E-07	*4*	6.82E-05	*4*	8.44E-04	*4*	0.00E+00	*4*
HDPE	1.31E-03	*4*	1.06E-04	*4*	2.40E-07	*4*	1.63E-04	*4*	1.29E-03	*4*	4.95E-05	*4*
LDPE	3.28E-03	*4*	4.45E-03	*4*	7.33E-07	*4*	5.89E-04	*4*	8.89E-04	*4*	0.00E+00	*4*
PP	2.73E-03	*4*	2.20E-03	*4*	3.67E-07	*4*	7.38E-05	*4*	1.90E-03	*4*	0.00E+00	*4*
Other plastic packaging	1.38E-03	*4*	8.68E-04	*4*	5.70E-07	*4*	1.14E-04	*4*	1.02E-03	*4*	1.73E-04	*4*
Non-packaging plastic	4.45E-03	*4*	4.59E-03	*4*	3.32E-07	*4*	1.34E-04	*4*	5.43E-03	*4*	9.15E-05	*4*
Cartons and alike	1.32E-03	*4*	8.50E-04	*4*	2.18E-06	*4*	8.83E-05	*4*	6.33E-04	*4*	2.94E-03	*4*
Glass	3.44E-02	*4*	1.02E-02	*4*	2.40E-07	*4*	2.42E-04	*4*	7.18E-04	*4*	6.34E-03	*4*
Ferrous metal	2.59E-02	*4*	3.60E-04	*4*	2.58E-07	*4*	1.97E-03	*4*	2.27E-02	*4*	6.52E-03	*4*
Non-ferrous metal	1.48E-02	*4*	2.43E-03	*4*	2.53E-07	*4*	3.11E-04	*4*	6.07E-03	*4*	6.53E-03	*4*
Textile	3.72E-02	*4*	7.85E-03	*4*	2.76E-06	*4*	1.40E-04	*4*	1.24E-03	*4*	4.86E-03	*4*
Wood	6.42E-03	*4*	6.85E-04	*4*	8.93E-07	*4*	7.23E-05	*4*	1.11E-03	*4*	4.95E-03	*4*
Others	4.42E-03	*4*	2.20E-02	*4*	2.50E-06	*4*	1.69E-04	*4*	3.62E-03	*4*	5.01E-03	*4*

Table A-1. *(continued)*

Material	Silver content (% dry mass)		Thallium content (% dry mass)		Tin content (% dry mass)		Vanadium content (% dry mass)		Zinc content (% dry mass)	
	Value	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**	**Value**	**Source**
Food waste	8.17E-03	*4*	3.68E-05	*4*	4.12E-04	*4*	9.79E-05	*4*	4.83E-03	*4*
Green waste	8.49E-04	*4*	6.12E-05	*4*	2.92E-04	*4*	5.73E-04	*4*	3.28E-02	*4*
Cellulose	4.22E-03	*4*	1.95E-05	*4*	8.89E-05	*4*	9.53E-05	*4*	3.29E-03	*4*
Paper	7.29E-05	*4*	1.94E-05	*4*	4.75E-05	*4*	1.70E-04	*4*	2.09E-03	*4*
Cardboard	6.10E-05	*4*	1.80E-05	*4*	6.45E-05	*4*	1.97E-04	*4*	3.52E-03	*4*
PET	4.73E-04	*4*	2.09E-05	*4*	3.18E-04	*4*	4.11E-05	*4*	4.14E-04	*4*
HDPE	1.75E-03	*4*	2.89E-05	*4*	1.13E-03	*4*	1.02E-04	*4*	6.52E-03	*4*
LDPE	4.80E-04	*4*	2.35E-05	*4*	1.21E-03	*4*	1.79E-04	*4*	1.28E-02	*4*
PP	1.13E-03	*4*	2.48E-05	*4*	2.05E-04	*4*	9.49E-05	*4*	3.51E-03	*4*
Other plastic packaging	5.38E-04	*4*	2.09E-05	*4*	2.99E-03	*4*	1.45E-04	*4*	1.08E-02	*4*
Non-packaging plastic	4.83E-04	*4*	2.17E-05	*4*	1.84E-03	*4*	7.58E-05	*4*	3.07E-02	*4*
Cartons and alike	1.48E-04	*4*	2.03E-05	*4*	2.29E-04	*4*	6.36E-04	*4*	4.33E-03	*4*
Glass	1.17E-03	*4*	5.32E-05	*4*	1.83E-03	*4*	7.45E-04	*4*	1.06E-02	*4*
Ferrous metal	1.17E-04	*4*	5.17E-05	*4*	2.45E-01	*4*	8.12E-04	*4*	1.22E-02	*4*
Non-ferrous metal	1.19E-04	*4*	5.20E-05	*4*	2.32E-03	*4*	9.61E-03	*4*	3.24E-02	*4*
Textile	8.32E-03	*4*	3.74E-05	*4*	7.08E-04	*4*	4.28E-04	*4*	2.41E-01	*4*
Wood	8.49E-03	*4*	3.82E-05	*4*	1.71E-04	*4*	6.84E-05	*4*	9.86E-03	*4*
Others	7.57E-03	*4*	5.75E-05	*4*	1.66E-03	*4*	3.99E-03	*4*	4.04E-02	*4*

Appendix B: Supporting information for Chapter 4

Table B-1. Gross and net electrical efficiency of the incineration facility of Madrid over 2008-2019.

	Input waste (t/year)	**Electricity generation potential** (MWh/year)	**Gross electricity generation** (MWh/year)	**Internal consumption** (MWh/year)	**Net electricity generation** (MWh/year)	**Gros electrical efficiency** (% LHV)	**Net electrical efficiency** (% LHV)
2019	332,980	1,275,889	228,627	62,572	162,088	16.85%	12.70%
2018	328,680	1,268,680	222,922	64,827	170,014	16.64%	13.40%
2017	314,035	1,284,309	197,562	64,559	171,601	15.44%	13.36%
2016	270,035	1,251,746	189,727	58,856	153,630	17.24%	12.27%
2015	257,605	1,083,750	177,577	53,450	130,192	16.91%	12.01%
2014	241,730	986,770	145,163	49,640	117,337	14.73%	11.89%
2013	242,123	985,168	166,977	42,633	102,530	16.92%	10.41%
2012	265,919	1,049,867	183,642	51,797	125,780	16.95%	11.98%
2011	307,140	1,100,525	212,486	54,202	135,524	16.98%	12.31%
2010	315,130	1,279,846	236,160	57,681	139,881	18.39%	10.93%
2009	311,295	1,339,532	234,841	59,903	163,019	18.51%	12.17%
2008	313,064	1,357,057	224,660	62,020	166,607	17.61%	12.28%
Avg	**291,645**	**1,348,294**	**199,247**	**56,845**	**144,850**	**16.74%**	**12.22%**

Note: The electricity generation potential was calculated from the mass of waste and an average lower heating value of 14.7 MJ/kg. The gross and net electrical efficiencies were calculated as the ratio between the gross and net generation and this potential.
Source: Data was obtained from the annual reports produced by the Madrid City Council (www.madrid.es).

Table B-2. Transfer coefficients calculated from on-site measured emissions at the incineration facility of Madrid.

Substance	E_R (mg/Nm^3)	E_M (kg/t waste)	F_W (t waste/year)	F_Q (kg element/year)	TC_{FG} (% input chemcial)
HCl	3.50E+00 (1)	3.36E-02 (2)	326,954	Cl: 1,076,060	Cl: 0.99
SO_2	7.00E-01 (1)	6.71E-03 (2)	326,954	S: 429,633	S: 0.26
HF	1.38E+02 (1)	1.32E-03 (2)	326,954	F: 7,391	F: 5.56
Cd	1.10E-03 (1)	1.05E-05 (2)	326,954	Cd: 211	Cd: 1.64
Cu	–	4.12E-05 (3)	326,954	Cu: 29,437	Cu: 0.04
Cr	–	4.55E-04 (3)	326,954	Cr: 26,208	Cr: 0.52
Hg	5.00E-4 (1)	4.79E-06 (2)	326,954	Hg: 3	Hg: 58.04
Ni	–	5.39E-06 (3)	326,954	Ni: 2,611	Ni: 0.06
Pb	–	4.05E-05 (3)	326,954	Pb: 327	Pb: 3.70
Zn	–	1.48E-03 (3)	326,954	Zn: 206,049	Zn: 0.21

Source: [1] Pérez et al. (2018); [2] Calculated as indicated above; [3] Margallo (2014)

Notes: $\mathbf{E_R}$: emission level per volume of flue gas at the reference oxygen level (11% dry vol.).

$\mathbf{E_M}$: emission level per tonne of waste incinerated. This value can be calculated from E_R as follows:

$$E_M = E_R \cdot \frac{21 - O_R}{21 - O_M} \cdot V_{FG}$$

where O_M is the actual oxygen level in the flue gas (7% dry vol.), O_R is the reference oxygen level in the flue gas (11% dry vol.), and V_{FG} is the volume of flue gas produced per tonne of waste incinerated (6,849 m^3/tonne waste).

$\mathbf{F_W}$: annual mass of waste used as reference.

$\mathbf{F_Q}$: annual mass of chemical elements in the waste used as reference.

$\mathbf{TC_{FG}}$: transfer coefficient of chemical elements to the flue gas during incineration, calculated as follows:

$$TC_{FG} = \frac{E_M \cdot F_W \cdot \frac{M_{element}}{M_{compund}}}{F_Q}$$

where $M_{element}$ and $M_{compound}$ are the molecular weight of the compound and the chemical element.

Table B-3. Spanish electricity generation mix in 2019, 2025, 2030, and 2040 based on the Spanish energy and climate integrate plan (Ministerio para la Transición Ecológica, 2020).

Power generation technology	**Contribution to the electricity mix (%)**			
	2019	**2025**	**2030**	**2040**
Hard coal	4.86	–	–	–
Hydro, pumped storage	0.63	1.91	3.46	4.29
Hydro, reservoir	8.16	7.93	7.05	5.80
Hydro, run-of-river	1.32	1.28	1.14	0.94
Natural gas, combined cycle	21.18	11.32	9.40	2.32
Natural gas, co-generation	11.35	5.66	4.10	0.72
Natural gas, conventional	1.10	0.97	0.86	0.73
Nuclear, boiling water reactor	3.44	3.03	1.16	0.00
Nuclear, pressure wáter reactor	17.96	15.84	6.05	0.00
Oil	1.09	1.01	0.89	0.77
Solar thermal, parabolic trough	1.94	4.55	6.55	8.57
Solar thermal, tower	0.04	0.10	0.15	0.19
Wind, onshore	20.81	30.21	34.54	42.89
Biogs, co-generation	0.34	0.37	0.37	0.40
Wood, co-generation	1.39	2.35	3.22	4.02
Solar photovoltaic, open ground	3.55	12.70	20.37	27.80
Incineration	0.85	0.77	0.67	0.56

Appendix C: Supporting information for Chapter 5

Table C-1. Lower and upper design capacity of the new waste treatment facilities implemented in the optimization framework.

Facility	**Capacity range** (tonnes/year)	**Assumptions and source**
New MRF for packaging waste	25,000–100,000	Capacity range based on the techno-economic analysis performed by Cimpan et al. (2016)
New RDF facility	45,000–360,000	Based on the capacity of mechanical-biological treatment (MBT) plants in Spain, which ranges from 45,257 to 362,488 tonnes/year (Gallardo, 2014)
New composting	10,000–300,000	Minimum capacity is 10,000 tonnes/year based on data for German composting plants (Pinasseau et al., 2018), while the maximum capacity is the same as for AD facilities.
New anaerobic digestion	25,000–300,000	Capacity range based on Smith et al. (2021)
New small incineration New medium incineration New large incineration	50,000–100,000 100,000–250,000 250,000–450,000	Classification of incineration by capacity according to the Best Available Techniques (BAT) document (Neuwahl et al., 2019): • small plants: < 100,000 tonnes/year • medium plants: 100,000 to 250,000 tonnes/year • large plants: > 250,000 tonnes/year
New RDF incineration	250,000–450,000	Same capacity range as for large incineration was assumed.
New landfill	20,000–2,000,000	Capacity range based on Kaza and Bhada-Tata (2018).

Note: MRF: material recovery facility. RDF: refuse-derived fuel.

Table C-2. Minimum and maximum capacity utilization rate of the waste treatment facilities implemented in the optimization framework.

Facility	**Capacity utilization rate** (% capacity installed)	Assumptions and source
MRF for residual waste MRF for packaging waste	50–91% 50–91%	The economic viability of MRFs is highly sensitive to the availability of feedstock. Operating below the treatment capacity entails large economic losses (Cimpan et al., 2016). Over the period 2015-2019, the minimum capacity utilization rate of MRFs in Madrid was around 50% for MRFs for residual waste and 33% for MRFs for packaging waste. These values indicate that facilities operated well below the design capacity. Herein, a 50% minimum capacity utilization rate was considered
RDF facility	50–91%	Same assumption as for MRFs.
Composting	10-91%	A 10% minimum capacity utilization rate was considered.
Anaerobic digestion	50–91%	The capacity utilization rate of existing anaerobic digestion (AD) facilities in Madrid ranged from 64% to 74% over the period 2015-2019. Similar figures have been proposed in the literature. Wolf et al. (2009) found that the minimum capacity utilization rate for 70 AD plants in Europe was around 60%. Hochloff and Braun (2014) propose a 50% minimum capacity utilization rate. Herein, a 50% minimum capacity utilization rate was considered
Incineration	70–91%	The average capacity utilization rate of the existing incineration facility in Madrid over the period 2008-2019 was 89%, while the minimum capacity utilization rate was 73%. Jaunich et al. (2019) propose an 80% minimum capacity utilization rate for incineration facilities operating in the U.S. Herein, a 70% minimum capacity utilization rate was considered.
Landfills	0–100%	Landfills are always available to receive waste. Therefore, their minimum capacity utilization rate is 0%

Note: MRF: material recovery facility. RDF: refuse-derived fuel.

Table C-3. Modelling of the linear capital costs functions for the new waste treatment facilities implemented in the optimization framework.

Facility	Modelling approach
New MRF for packaging waste	The capital costs function of new MRFs for packaging waste was developed with data from Cimpan et al. (2016). This study calculated capital costs between 7.2 and 22 million EUR2019 for four facilities with capacities ranging from 25,000 to 100,000 tonnes/year
New RDF facility	The capital costs function of a new RDF facility was developed with the scaling factor method due to the scarce information available in the literature. The scaling factor method establishes that the capital costs at any capacity can be approximated from the capital costs and the capacity of a reference facility as follows (Ng and Phan, 2021; Sadhukhan et al., 2014): $$\frac{COST_{analysis}}{COST_{ref}} = \left(\frac{CAP_{analysis}}{CAP_{ref}}\right)^{R}$$ where $CAP_{analysis}$ is the analyzed capacity, $COST_{analysis}$ are the capital costs of the facility at the analyzed capacity, CAP_{ref} and $COST_{ref}$ are the capacity and capital costs of the reference facility, respectively, and R is the scaling factor. The capacity and capital costs of the reference RDF facility were retrieved from Voss et al. (2021). According to this work, an RDF facility with 110,00 tonnes/year of capacity has a capital cost of about 9.3 million EUR2019. The scaling factor was assumed the same as for a new MRF for packaging waste, which was calculated equal to 0.73 based on data from Cimpan et al. (2016).
New composting	The capital costs functions of new composting facilities were developed with the scaling factor method described above. For windrow composting, the facility described in van Haaren et al. (2010) was used as the reference. This facility has a capacity of 40,000 tonnes/year and a capital cost of 2.9 million EUR2019. For tunnels composting, the reference facility was obtained from Mayer et al. (2020) corresponding to a facility with 27,300 tonnes/year capacity and a capital cost of 2.4 million EUR2019. In both cases, a scaling factor of 0.6 was applied after Mayer et al. (2020)
New anaerobic digestion	The capital costs of new anaerobic digestion (AD) facilities include the costs of the pre-treatment system, digesters, the energy conversion device (CHP units or upgrading system), and the digestate composting facility. The capital costs of the pre-treatment, digesters, CHP units, and upgrading at each capacity level were calculated based on exponential equations with the form $y = ax^{b}$, where y are the capital costs (EUR), x is the capacity (tonnes/year), and b is the scaling factor. The exponential equations were adopted from Sanscartier et al. (2012) for the pre-treatment system, from Smith et al. (2021) for the digesters, and from Bauer et al. (2013) for the upgrading system. The capital costs of the composting were calculated with the linear function developed above.

Table C-3. *(continued)*

Facility	**Modelling approach**
New incineration	The capital costs functions for small, medium, and large new incineration facilities were developed based on the Best Available Techniques (BAT) document published by the European Commission (Neuwahl et al., 2019). This document provides average unitary capital costs for incineration within the European context ranging from 726 to 382 EUR/tonne-year corresponding to capacities between 100,000 and 600,000 tonnes/year.
New landfill	The capital costs function of a new sanitary landfill includes the costs of the infrastructure required, such as land and machinery (e.g., LFG and leachate recovery systems), plus the costs of the post-closure final cap. The capital costs of landfill infrastructure range from 1 million USD for a capacity of 20,000 tonnes/year to 50 million USD for a capacity of 2,000,000 tonnes/year (Kaza and Bhada-Tata, 2018). The post-closure capital costs were assumed 30% of the capital costs (Kaza and Bhada-Tata, 2018).

Note: MRF: material recovery facility. RDF: refuse-derived fuel.

Table C-4. Data used to develop linear fixed operating costs functions for the waste treatment facilities implemented in the optimization framework.

Facility	**Maintenance annual cost** (% CAPEX)	**Insurance annual cost** (% CAPEX)	**Number of workers**	**Average gross salary** (EUR/year)	**Supervision costs** (% of salary)	**Salary overhead** (% of salary + supervision costs)
MRF for packaging waste	3.00% [1]	0.50% [1]	$y = 1.5365 \cdot x^{0.3028}$, where y is the number of workers and x is the treatment capacity (tonnes/year) [a]	43,441 [6]	25% [7]	50% [7]
MRF for residual waste	3.00% [1]	0.50% [1]				
RDF facility	3.00% [1]	0.50% [1]				
Anaerbic digestion	3.00% [2]	0.15% [2]	17 [5]			
Composting	2.70% [3]	1.00% [3]	3 [b]			
Small incineration	3.00% [4]	3.00% [4]	16 [2, 4]			
Medium incineration	3.00% [4]	3.00% [4]	24 [2, 4]			
Large incineration	3.00% [4]	3.00% [4]	37 [2, 4]			
RDF incineration	3.00% [4]	3.00% [4]	37 [2, 4]			

Note: [a] The number of workers at MRFs for packaging waste increases nonlinearly with treatment capacity, from 30 workers for 25,000 tonnes/year capacity to 44 workers for 50,000 tonnes/year capacity and 54 workers for 100,000 tonnes/year capacity (Cimpan et al., 2016). The nonlinear regression equation that determines the number of workers as a function of treatment capacity was built based on these figures. This equation was used also for the MRFs for residual waste and the RDF facility. [b] Composting is not particularly labor intensive. 3 full-time equivalent workers were considered (regardless capacity), one per processing step (i.e., waste receiving, waste turning, and compost management). MRF: material recovery facility. RDF: refuse-derived fuel.
Source: [1] Andreasi Bassi et al. (2020); [2] Martinez-Sanchez et al. (2015); [3] Andreasi Bassi et al. (2021); [4] Yassin et al. (2009); [5] Smith et al. (2021); [6] Hogg et al. (2015); [7] Towler and Sinnott (2013)

Table C-5. Purchasing prices of the commodities and services consumed by the waste treatment facilities implemented in the optimization framework.

Commodity/service	Units	Source	2020-2025	2025-2030	2030-2035	2035-2040
electricity	EUR/MWh	*1*	108.8	109.1	109.4	109.7
diesel	EUR/kg	*2*	1.02	1.10	1.15	1.20
heat, light fuel oil	EUR/MJ	*3*	0.025	0.027	0.028	0.030
water, tap	EUR/kg		0.001	0.001	0.001	0.001
water, decarbonized	EUR/kg		0.001	0.001	0.001	0.001
quicklime	EUR/kg	*4*	0.11	0.11	0.11	0.11
activated carbon	EUR/kg	*4*	1.03	1.03	1.03	1.03
ammonia	EUR/kg	*4*	0.30	0.30	0.30	0.30
titanium dioxide	EUR/kg	*5*	105	105	105	105
steel wire	EUR/kg	*6*	2.37	2.37	2.37	2.37
transport	EUR/tkm	*7*	2.37	2.37	2.37	2.37
wastewater treatment	EUR/kg	*8*	0.00	0.00	0.00	0.00

Source: [1] Electricity prices for industrial consumers projected from historical data (Eurostat, 2021); [2] Navas-Anguita et al. (2019); [3] Same price as for diesel; [4] Tonini et al. (2020); [5] Ecoinvent database v3.7.1 (Wernet et al., 2016); [6] Ng and Phan (2021); [7] Andreasi Bassi et al. (2020); [8] Included in the fixed operating costs of each facility

Table C-6. Revenue from the sale of recyclable materials, compost, and energy.

Product type	**Product**	**Units**	**Source & notes**	**2020-2025**	**2025-2030**	**2030-2035**	**2035-2040**
recyclable material	paper, sorted and baled	EUR/tonne	*1, a*	114	114	114	114
recyclable material	cardboard, sorted and baled	EUR/tonne	*1, a*	81	81	81	81
recyclable material	PET, sorted and baled	EUR/tonne	*2, a*	264	264	264	264
recyclable material	HDPE, sorted and baled	EUR/tonne	*2, a*	378	378	378	378
recyclable material	LDPE, sorted and baled	EUR/tonne	*2, a*	12	12	12	12
recyclable material	PP, sorted and baled	EUR/tonne	*2, a*	125	125	125	125
recyclable material	mix plastic, sorted and baled	EUR/tonne	*2, a*	103	103	103	103
recyclable material	cartons & alike, sorted and baled	EUR/tonne	*3, a*	7	7	7	7
recyclable material	glass, sorted and baled	EUR/tonne	*1, a*	25	25	25	25
recyclable material	ferrous metal, sorted and baled	EUR/tonne	*1, a*	162	162	162	162
recyclable material	non-ferrous metal, sorted and baled	EUR/tonne	*1, a*	980	980	980	980
organic fertilizer	compost	EUR/tonne	*b*	0	0	0	0
energy	electricity	EUR/MWh	*c*	70	70	70	70
energy	biomethane	EUR/m^3	*1*	0.42	0.42	0.42	0.42

Note: [a] Projecting the revenues from the sale of recyclable materials up to 2040 is very challenging if not impossible (Hogg et al., 2015). Consequently, it was assumed that revenues will remain constant. [b] Market prices for compost in the EU ranges from a symbolic price (e.g., 1 EUR/tonne) up to 14 EUR/tonne for high quality compost. It is a common practice for waste managers to cover the costs of transporting and spreading the compost. For this reason, the hypothesis that compost is given away to farmers at a price of 0 EUR/tonne is common in the literature (Cobo et al., 2019). [c] Revenue from electricity sales was assumed to be fixed at 70 EUR/MWh which is guaranteed for the entire decision horizon (Hogg et al., 2015).
Source: [1] Hogg et al. (2015); [2] Andreasi Bassi et al. (2020); [3] Ip et al. (2018)

Table C-7. Net electrical efficiency of new incineration facilities as a function of the treatment capacity.

Facility size	**Capacity** (tonnes/year)	**Gross electrical efficiency** (%)	**Net electrical efficiency** (%)	**Source**
Small (50,000–100,000 tonnes/year)	44,348	24.3	20.8	Lombardi et al. (2015)
	44,348	15.5	13.3	Lombardi et al. (2015)
	50,000	–	18.3	Yassin et al. (2009)
	100,000	–	20.0	Lombardi et al. (2015)
	100,000	–	15.0	Lombardi et al. (2015)
	100,000	–	22.4	Yassin et al. (2009)
		Average	**18.3**	
Medium (100,000–250,000 tonnes/year)	135,652	24.6	20.5	Lombardi et al. (2015)
	170,294	–	20.6	Lombardi et al. (2015)
	230,000	–	20.7	Turconi et al. (2011)
		Average	**20.6**	
Large (250,000–450,000 tonnes/year)	813,913	32	28.5	Lombardi et al. (2015)
	700,000	–	26.0	Ardolino et al. (2020)
	551,728	27.0	24.0	Lombardi et al. (2015)
	530,000	34.5	30.0	Lombardi et al. (2015)
	450,000	–	24.2	Turconi et al. (2011)
	450,000	–	24.0	Dong et al. (2018a)
	257,599	28.0	25.0	Lombardi et al. (2015)
		Average	**26.0**	