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TESIS DOCTORAL

Optimización del ciclo de vida para el diseño sostenible de sistemas circulares de gestión de residuos municipales

PhD THESIS

A life cycle optimization framework for the sustainable design of circular municipal solid waste management systems

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“We do not inherit the Earth from our ancestors;

we borrow it from our children.”

Proverb

CONTENTS

PREFACE

This doctoral dissertation was elaborated as a compendium of papers published in scientific journals. The following four papers constitute the core of the thesis:

1. Cobo, S.; Dominguez-Ramos, A.; Irabien, A. From linear to circular integrated waste management systems: A review of methodological approaches. *Resour. Conserv. Recycl.* **2018**, *135*, 279-295. <https://doi.org/10.1016/j.resconrec.2017.08.003>.
   2018 Journal Impact Factor: 7.044.

2. Cobo, S.; Dominguez-Ramos, A.; Irabien, A. Minimization of resource consumption and carbon footprint of a circular organic waste valorization system. *ACS Sustainable Chem. Eng.* **2018***, 6*, 3493-3501.
   <https://pubs.acs.org/doi/full/10.1021/acssuschemeng.7b03767>.
   2018 Journal Impact Factor: 6.970.

3. Cobo, S.; Dominguez-Ramos, A.; Irabien, A. Trade-offs between nutrient circularity and environmental impacts in the management of organic waste. *Environ. Sci. Technol.* **2018***, 52*(19), 10923-10933.
   <https://pubs.acs.org/doi/full/10.1021/acs.est.8b01590>.
   2018 Journal Impact Factor: 7.149.

4. Cobo, S.; Levis, J.W.; Dominguez-Ramos, A.; Irabien, A. Economics of enhancing nutrient circularity in an organic waste valorization system. *Environ. Sci. Technol.* **2019**, 53(11), 6123-6132. <https://pubs.acs.org/doi/full/10.1021/acs.est.8b06035>.
   2018 Journal Impact Factor: 7.149.

The journal *Resources, Conservation and Recycling* is ranked within the first decile (D1) of the 2018 Journal Citations Report (Science Citation Index) in the Environmental Sciences category, *ACS Sustainable Chemistry & Engineering* belongs to the D1 in the Chemical Engineering category, and *Environmental Science & Technology* is within the D1 of the Environmental Engineering and Environmental Sciences categories.

Selene Cobo graduated in Chemical Engineering at the University of Cantabria in 2014, where Professor Ángel Irabien and Assistant Professor Antonio Domínguez-Ramos engaged her in research in the process systems engineering and sustainability fields. After earning her Master's degree in Chemical Engineering at the University of Cantabria and the University of the Basque Country, she enrolled in the doctoral program in Chemical Engineering, Energy and Processes at the University of Cantabria.

The research that led to the publication of the three first listed papers was entirely conducted in the Development of Chemical Processes and Pollution Control (DEPRO) research group under the guidance of Prof. Irabien and Dr. Domínguez-Ramos. The DEPRO group, whose head is Prof. Irabien, belongs to the Department of Chemical and Biomolecular Engineering, where a variety of projects seeking to transfer innovative and sustainable technologies to the chemical and process industries are carried out.

As part of the requirements for obtaining the International Doctorate mention, the PhD candidate spent three months (April – June 2018) as a visiting scholar in the Department of Civil, Construction and Environmental Engineering at North Carolina State University (NCSU) advised by Research Assistant Prof. James W. Levis. The fourth of the published papers is a direct result of that collaboration.

Moreover, from May to August 2019 Ms. Cobo was a member of the Process-Energy-Environmental Systems Engineering lab at Cornell University, where she learned and applied the fundamentals of bilevel optimization to her work under the supervision of Prof. Fengqi You.

The author has been the recipient of two predoctoral fellowships granted by the University of Cantabria and the Spanish Ministry of Education (code FPU 15/01771). Her visit to NCSU was funded by a predoctoral mobility scholarship awarded by the University of Cantabria and by the research project CTQ2016-76231-C2-1-R, whereas her research at Cornell University was sponsored by the FPU program (code EST18/00007). She gratefully acknowledges this financial support.

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En primer lugar, debo agradecerle a Ángel haberme brindado la oportunidad de continuar mi formación y embarcarme en este proyecto. La idea primigenia de la tesis fue concebida por él; le estoy muy agradecida por haberme proporcionado las herramientas – conocimiento y medios – para poder abordarla, y por la libertad y confianza concedidas para explorar nuevas ideas a medida que la investigación avanzaba.

De Antonio cabe destacar su pasión por la investigación y la sostenibilidad, así como su vocación docente, las cuales son muy apreciadas por sus alumnos y doctorandos. Su sentido de la responsabilidad, constancia y apoyo han facilitado enormemente el desarrollo de la tesis y contribuido a mejorar la calidad del trabajo.

I would like to thank James for his warm welcome at NCSU. His knowledgeable and thorough remarks were very valuable during and after my visit. I am also grateful to Prof. You for the opportunity to conduct research at Cornell University.

Me gustaría subrayar aquí la importante labor de todos los miembros del Departamento en materia de divulgación, que sin duda sirve de inspiración para las nuevas generaciones. El valor de la ciencia al servicio de la sociedad que inculcan desde las aulas ha influido sin lugar a duda en la elaboración de esta tesis.

Precisamente a los compañeros con los que he compartido experiencias (muchas de ellas buenas, otras más amargas) desde que comencé en esas aulas mi paso por la Universidad de Cantabria, les quiero agradecer su ánimo a lo largo de todos estos años. Muy especialmente, a mis compañeras de despacho, que han hecho más llevadero y ameno el trabajo.

A todos a los que he conseguido convencer en el transcurso de esta etapa predoctoral para que reciclen más y mejor, ¡gracias por aguantarme! Las brujis merecen una mención aparte, ya que siempre quieren saber a qué me dedico exactamente y para qué sirve.

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RESUMEN

El concepto de la economía circular surgió en respuesta a la contaminación y al agotamiento de los recursos naturales asociados a nuestro modelo de producción y consumo. En una economía circular, los residuos son considerados un recurso que debe ser valorizado antes de ser reintroducido de nuevo en los ciclos productivos, con lo que se consigue minimizar la extracción de recursos y la acumulación de residuos. Como consecuencia de las políticas de la Unión Europea que priorizan la valorización de los residuos sobre su eliminación, se ha conseguido reducir progresivamente el promedio europeo de residuos sólidos municipales enviados a vertedero, e incrementar las tasas de incineración, reciclaje y compostaje. No obstante, no todos los Estados Miembros siguen la misma tendencia; solo el 30% de los residuos sólidos municipales generados en España fueron reciclados o compostados en el 2016, lo cual constituye una desviación significativa respecto al objetivo impuesto por la Unión Europea de alcanzar una tasa de reciclaje del 50% en el año 2020.

Las herramientas de la ingeniería de procesos y sistemas pueden contribuir al diseño de estrategias de gestión de residuos óptimas que satisfagan los criterios de una economía circular. Sin embargo, todavía no se ha demostrado que la implementación de una economía circular conlleve una disminución en el consumo de recursos y los impactos ambientales, o que no desacelere el crecimiento económico. Consecuentemente, los objetivos de esta tesis son dos: desarrollar un marco metodológico que permita determinar la configuración óptima de sistemas integrados de gestión de residuos y recursos bajo una perspectiva del ciclo de vida, e investigar si la adopción de una economía circular es una medida efectiva para incrementar los beneficios económicos y reducir el consumo de recursos y los impactos ambientales.

El marco metodológico desarrollado – presentado en el primero de los artículos que constituyen esta tesis – está basado en la expansión de los límites de los sistemas integrados de gestión de residuos para incluir los subsistemas involucrados en las etapas previas del ciclo de vida de las sustancias residuales, desde la extracción de recursos hasta la generación de los desechos. Estos sistemas, denominados en la tesis sistemas circulares integrados de gestión de residuos (SCIGR), permiten evaluar las consecuencias de recircular los componentes recuperados de los residuos. Con el objetivo de cuantificar la circularidad de estas sustancias en un SCIGR, se propuso un

nuevo indicador de circularidad, descrito en la tercera publicación de la tesis. En el segundo artículo que compone la tesis se examinó la viabilidad del marco metodológico propuesto aplicándolo a un caso de estudio, la gestión de nutrientes y residuos orgánicos municipales en la Comunidad Autónoma de Cantabria. Para ello, se definieron los límites del sistema y se diseñó una superestructura con procesos unitarios alternativos. El modelo del sistema describe cómo los fertilizantes industriales y los productos que pueden ser recuperados de los residuos orgánicos municipales (compost, material bio-estabilizado, digestato, estruvita y sulfato de amonio) son aplicados al suelo como producto fertilizante para cultivar maíz – el principal cultivo forrajero en Cantabria – de acuerdo a diferentes estrategias, dependiendo de su contenido en nutrientes. Por último, la cuarta publicación recoge un análisis económico del sistema.

El modelo mecanístico del sistema se desarrolló combinando diferentes tipos de software para llevar a cabo el análisis de flujo de materiales y el análisis de ciclo de vida (ambiental y económico) de los procesos unitarios, y para modelar la distribución de nutrientes en el sistema. Se formuló un problema multi-objetivo de programación lineal entera mixta basado en las ecuaciones del modelo. El problema está sujeto a las restricciones del caso de estudio y a las impuestas por la legislación europea. Las funciones objetivo identificadas se clasifican en:

- Funciones objetivo que miden el progreso realizado en la consecución de una economía circular: los indicadores de circularidad de carbono, nitrógeno y fósforo.
- Funciones objetivo que cuantifican diferentes aspectos asociados a la sostenibilidad del sistema: i) uso de recursos: consumo de materias primas no renovables y área de vertedero, ii) impactos ambientales: calentamiento global y eutrofización marina y de agua dulce, iii) rentabilidad económica: costes anuales totales del subsistema de gestión de residuos.

Las soluciones a los problemas de optimización planteados – basadas en la integración de múltiples procesos unitarios – indican cuáles son los procesos unitarios seleccionados y los flujos de materia que deben entrar a cada uno de ellos. Los resultados – sujetos a las hipótesis formuladas – sugieren que mejorar la circularidad de los recursos no implica necesariamente una reducción en el consumo total de recursos naturales o la emisión de cargas ambientales del sistema. Asimismo, puede conducir a un aumento en los costes de gestión de residuos, los cuales se podrían minimizar mediante la cooperación de los diferentes actores que forman parte del sistema circular. Por tanto, el éxito de la economía circular dependerá del equilibrio entre la circularidad de los recursos y sus implicaciones sobre diferentes aspectos de la sostenibilidad; las consecuencias de mejorar la circularidad de los recursos deben ser analizadas caso a caso.

ABSTRACT

The concept of the circular economy arose in response to the pollution and the depletion of natural resources related to our production and consumption model. In the context of a circular economy, waste is viewed as a resource that must be upgraded before being reintroduced back into the production cycles, thereby minimizing resource extraction and waste disposal. As a consequence of the European policies that prioritize waste valorization over disposal, the average fraction of municipal solid waste disposed of in landfills within the European Union has steadily decreased in recent years, as the incineration, recycling and composting rates have risen. Nonetheless, these statistics vary widely among Member States; only 30% of the municipal solid waste generated in Spain was recycled or composted in 2016, which constitutes a significant deviation from the 50% recycling target set by the European Union for year 2020.

Process systems engineering tools can contribute to the design of optimal waste management strategies that fulfill circular economy criteria. However, it has not been demonstrated yet that the implementation of a circular economy will decrease resource consumption and environmental impacts, or that it will not hamper economic growth. Therefore, the objectives of this dissertation are twofold: to develop a methodological framework capable of selecting the optimal configuration of integrated waste and resource management systems under a life cycle perspective, and to investigate whether adopting a circular economy is an effective measure to attain increased economic benefits and a reduction in resource consumption and environmental impacts.

The developed framework – presented in the first paper compiled in this thesis – is based on the expansion of the boundaries of integrated waste management systems to include the upstream and midstream subsystems involved in the previous life cycle stages of the waste components, from resource extraction to waste generation. These systems, categorized as Circular Integrated Waste Management Systems (CIWMSs) in the dissertation, enable assessing the consequences of the recirculation of the recovered waste components into the upstream subsystems. To quantify the circularity of these substances within CIWMSs, a new circularity indicator was proposed and described in the third publication comprised in the dissertation. In the second paper included in this thesis, the developed framework was tested on a case study: the

management of municipal organic waste and nutrients in the Spanish region of Cantabria. The system boundaries were defined and a superstructure containing alternative unit processes that could be integrated into the system was designed. The system model describes how industrial fertilizers and the products that could be recovered from municipal organic waste (compost, bio-stabilized material, digestate, struvite and ammonium sulfate) are applied to the soil to grow corn – the main fodder crop in Cantabria – according to different strategies depending on their nutrient composition. Finally, the economic analysis of the studied system is carried out in the fourth publication.

The bottom-up mechanistic model was developed combining different pieces of software to perform the material flow analysis, life cycle assessment and life cycle costing of the unit processes, and to model the distribution of nutrients within the system. A mixed integer linear programming multi-objective optimization problem was formulated based on the equations of the model. The problem is subject to the restrictions of the case study and those imposed by the European waste legislation. The identified objective functions are classified as:

- Objective functions that measure the progress toward the achievement of a circular economy: the circularity indicators of carbon, nitrogen and phosphorus.
- Objective functions that quantify different aspects of sustainability: i) resource use: the consumption of non-renewable raw materials and the landfill area, ii) environmental impacts: global warming, marine eutrophication and freshwater eutrophication, iii) economics: the total annual costs of the waste management subsystem.

The results of the optimization indicate the selected unit processes and the material flows that must enter each of them; the solutions to the multi-objective optimization problems were based on the integration of multiple unit processes. The findings of the research – subject to the formulated hypotheses – suggest that improving resource circularity does not necessarily entail a decrease in the overall consumption of natural resources or the emission of environmental burdens. Likewise, it can lead to increased waste management costs, which could be minimized through the cooperation of the different actors involved in the circular system. Thus, the successful implementation of a circular economy must rely on a proper balance between resource circularity and its sustainability implications; the consequences of enhancing resource circularity should be analyzed on a case-by-case basis.

CHAPTER 1

GOALS AND SCOPE

"Pollution is nothing but the resources we are not harvesting. We allow them to be dispersed because we've been ignorant of their value."

Richard Buckminster Fuller, American architect (1895-1983)

The imprint that humans are leaving on the planet is so deep that many experts agree that a new geologic era, characterized by substantial irreversible chemical and biological changes derived from human activities, should be defined: the Anthropocene.[1,2]

The release of $555 \cdot 10^9$ metric tons of carbon (C) to the atmosphere since the beginning of the Industrial Revolution has led to a steep increase in the CO_2 atmospheric concentration to over a third above preindustrial levels.[1,2] Setting aside the economic costs of the natural catastrophes associated with the consequential rise in global temperatures, such as coastal flooding,[3] the detrimental effects of climate change are quantifiable on the natural ecosystems[4,5] and human health.[6] The World Health Organization estimates that between 2030 and 2050 global warming will cause around 250,000 annual deaths, approximately 40% of which are due to childhood undernutrition.[7]

Climate change poses a risk to the stability of the global food system because of the lower crop productivities related to warmer temperatures, changes in rain patterns and the increased frequency of extreme weather events.[8] On the other hand, according to the IPCC,[9] the sector of the economy that encompasses agriculture, forestry and other land uses (e.g. the cultivation of bioenergy crops) emits almost a quarter of the anthropogenic greenhouse gases. Thus, agriculture could be at the core of a positive feedback loop exacerbated by the forthcoming increase in the global demand for food: as agriculture intensifies because of the poorer crop yields, more greenhouse gases are emitted, worsening those yields.

Moreover, the large amounts of nitrogen (N) and phosphorus (P) required by industrialized agriculture are a significant source of environmental pollution that is perturbing the global cycles of these elements. Around 120 million metric tons of molecular N (N_2) are converted annually into reactive forms for fertilization purposes.[10] In fact, anthropogenic sources contribute double the natural rate of terrestrial N fixation.[11] The N use efficiency of the most cultivated crops is typically below 40%; the remaining N ends in the atmosphere as N_2O, a powerful greenhouse gas, or leaches into the water bodies causing eutrophication, i.e. excessive biological productivity that may result in oxygen depletion and eventually, decreased biodiversity in the aquatic ecosystems.[11] Regarding the P flows, out of the 20 million metric tons that are annually mined from phosphate rock, around 9 million metric tons are transferred to the ocean each year, contributing to eutrophication.[10]

The disruption of the major biogeochemical cycles combined with the extensive land-use conversion – agriculture alone occupies about 38% of the Earth's terrestrial surface –[12] has triggered an unprecedented biodiversity loss that some scientists have labeled as the start of the Earth's sixth mass extinction.[13]

As Figure 1.1 shows, the human-driven perturbations in the CO_2 atmospheric concentration, the rate of biodiversity loss, the land use and the N and P cycles have already transgressed the planetary boundaries defined by Rockström et al.[14] as the thresholds above which abrupt irreversible environmental change with potentially disastrous consequences for humans could occur.

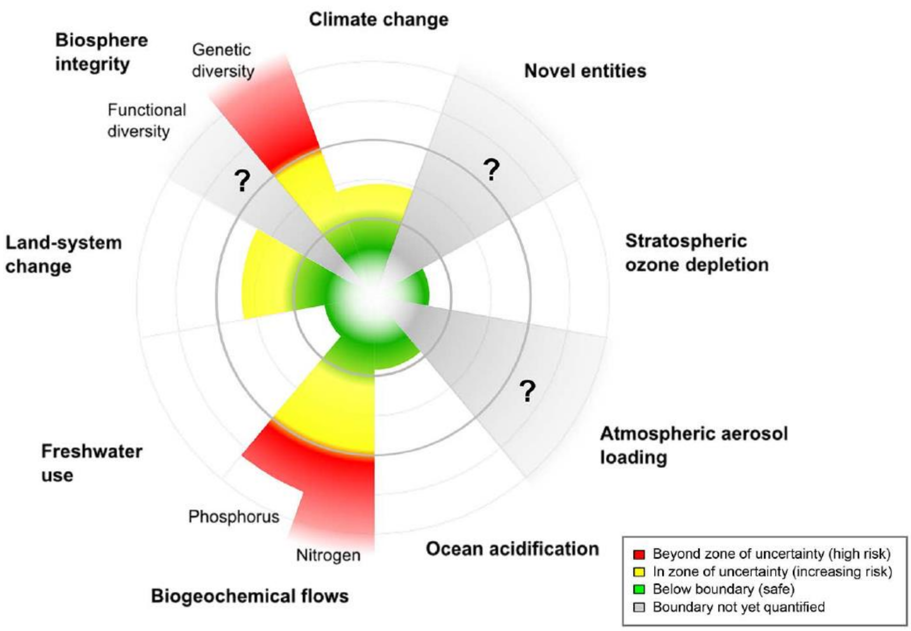

Figure 1.1. Planetary boundaries (adapted from Steffen et al.[15])

The challenge of maintaining the Earth system within a safe operating space while complying with the moral imperative of covering the basic human needs will be aggravated by the projected increase in the world population, which is expected to reach 9.7 billion in 2050.[16]

Revisiting the sustainability concept

Some authors have suggested that in the light of the research that indicates that the stable functioning of the Earth subsystems is a prerequisite for social prosperity, the concepts of sustainability and sustainable development established in the 1987 Brundtland report[17] should be reformulated.[18–20]

Griggs et al.[18] consider the three pillars of sustainability – economic, social and environment – a nested concept, as depicted in Figure 1.2: the economy fulfills a function for society, which lies within the Earth's life-support system. Therefore, they defined sustainable development as "Development that meets the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generation depends".

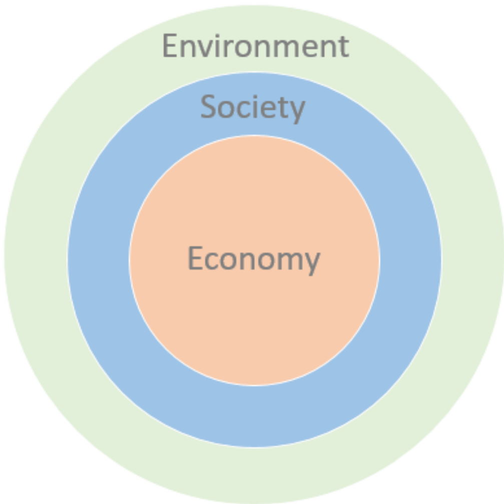

Figure 1.2. New sustainability paradigm

Holden et al.[19] go one step further and claim that "sustainable development constitutes a set of constraints on human behavior, including constraints on economic activity". They conclude that economic growth cannot be the priority of sustainable policies, which should focus on satisfying human needs, ensuring social equity and respecting environmental limits.

Nonetheless, economic growth is one of the 17 Sustainable Development Goals proposed by the United Nations for their 2030 agenda.[21] Recent studies have found that despite the multiple synergies between the Sustainable Development Goals, there are also trade-offs between them;[22,23] pursuing social goals is usually associated with higher environmental impacts.[23,24]

The results of the analysis performed by O'Neil et al.[24] suggested that it is possible to meet the basic universal human needs without transgressing planetary boundaries if the level of resource use per capita decreases. Thus, they recommended a deviation of the Sustainable Development Goals agenda from economic growth, under the implicit assumption that economic growth is intrinsically coupled to resource use, which is supported by other studies.[25,26]

However, given the global trends in resource extraction, shown in Figure 1.3, neither the dematerialization of the economy, (i.e. "the reduction in the quantity of materials used and/or the quantity of waste generated in the production of a unit of economic output",[27] nor the stabilization of the global demand for materials at the expense of a halt in the economic growth, are likely to occur in the near future without the appropriate incentives.[28]

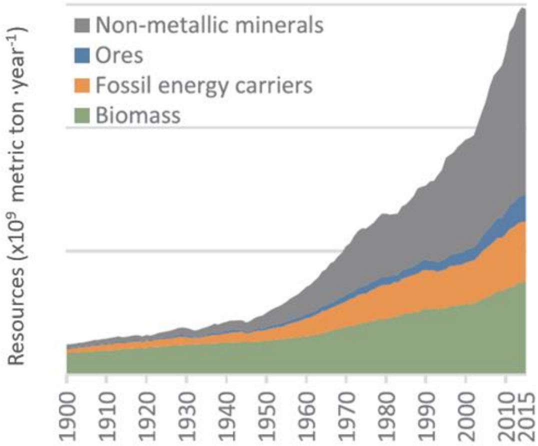

Figure 1.3. Global flows of resource extraction (adapted from Krausmann et al.[29])

Therefore, a novel economic model based on reduced resource consumption and capable of operating within the decision-making space where the environmental and social objectives overlap is needed to attain this new sustainability paradigm.

The coordinated management of waste and resources

The advocates of the transition toward a circular economy claim that the pressure on the natural ecosystems to meet the demand for natural resources could be minimized if the dominant linear trend of extracting, processing, consuming or using and then disposing of raw materials was reversed, so that the materials that are typically considered waste can gain back the status of resource, after undergoing the pertinent upgrading processes.[30,31]

One of the main organisms that seeks to promote the circular economy by collaborating with businesses, government and academia, is the Ellen MacArthur Foundation.[32] It shares its vision with other international initiatives, like the Circle Economy,[33] in which several international businesses and institutions participate, or, at the national level, the Spanish Circular Economy Foundation.[34]

The seminal thinkers and supporters of the circular economy were inspired by the high efficiency of the systems found in nature.[35] As an illustration, the average terrestrial ecosystem recycles P roughly 50 times before it is lost in freshwaters.[36]

The cradle-to-cradle design philosophy proposed by McDonough and Braungart,[37] which envisions production systems where materials emulate the cyclical biological metabolism of nutrients, contributed to setting the foundations for the circular economy concept.

Furthermore, the principles of the circular economy are interwoven with those of the industrial ecology discipline,[38] which views industrial systems as entities analogous to the natural ecosystems, whose exchange of resources with the surrounding environment can be optimized.[39] Nonetheless, whereas the focus of industrial ecology is on the industrial performance, the scope of the circular economy is broader.

Some authors have taken a critical stance toward the theoretical basis of the circular economy, under the premise that the natural ecosystems work sub-optimally. The detractors argue that in accordance with the second law of thermodynamics, the subsystems that conform the biosphere generate disorder or waste.[40,41] As in nature, the inefficiencies of the recycling technologies do not allow completely closed loops of materials. Throughout this thesis, the term “circular economy” is not taken literally as a perfectly closed system that does not rely on the consumption of external virgin resources. Instead, it refers to an economic model that simultaneously optimizes waste and resource management to minimize the consumption of natural resources.

Another argument against the basis of the circular economy is that natural ecosystems require a continuous supply of solar energy to maintain their complexity. Likewise, energy is required to create value from downgraded materials.[40,41] The skeptics reason that, unless technologies capable of efficiently harvesting energy from renewable sources are implemented, the better environmental profile of circular systems with respect to the traditional linear systems is questionable. Hence, the adoption of circular practices for the management of natural resources should be parallel to the expasion of renewable energy sources, whose barriers, according to recent studies that explored the feasibility of a 100% renewable energy mix, ”are primarily social and political, not technological or economic”.[42–44]

Regarding the connection of the circular economy with the sustainability concept, it has not been well established yet. It is widely assumed in the literature – without solid substantiation – that the improvement in resource efficiency derived from the implementation of a circular economy will decouple environmental impacts from economic growth,[45,46] whereas the social implications of the circular economy, like more manual labor, are often overlooked.[47–49] Clearly, more research is needed in this area.

Therefore, the efforts to achieve sustainability should not focus solely on enhancing the circularity of resources. The 7 Rs rule – Regulation, Reducing, Reusing, Recycling, Recovering, Rethinking and Renovation – summarizes the available tools to minimize the dependence on raw materials.[50] The sale of services rather than products,[51] waste prevention,[52] the promotion of reuse and product life extension strategies,[53] measures to minimize product weight and the design of products for disassembly[35] should be simultaneously encouraged. In this respect, the commitment of companies to sustainability should not be underestimated.

For instance, Adidas® has accomplished a sustainable business model manufacturing running shoes and other sportswear made from upcycled plastics that Parley's Ocean Plastic™ collects from the ocean.[54]

The exchange of resources between traditionally separate industries – also known as industrial symbiosis –[55] is another approach to the collective minimization of resource consumption. The first industrial symbiosis network reported in the literature is the eco-industrial park of Kalundborg (Denmark),[45] where four companies (a power plant, an oil refinery, a biotech and pharmaceutical company and a producer of plasterboard) exchange water and steam.[56]

These examples illustrate how the key to implementing successful strategies for resource recovery lies on the supply chain design.[57,58] Indeed, most of the environmental impact of companies comes from their supply chain or from the consumers' use phase.[59] Thus, a significant reduction in the overall environmental impacts of businesses would require coordinated global policies, which private companies cannot control.

On the contrary, the public sector is in a privileged position to coordinate the sustainable management of waste and resources. Some efforts have already been made to develop new policies that promote the circularity of resources and a framework to quantify the performance of a circular economic model.

Monitoring the circular economy

The European Commission recently proposed a set of ten indicators to measure the progress toward the attainment of a circular economy.[60] The indicators belong to the following thematic areas: production and consumption, waste management, secondary raw materials and competitiveness and innovation. The value of these metrics for the European Union and Spain are compiled in Table 1.1.

The production and consumption indicators include waste generation rates, the self-sufficiency for raw materials – which reveals how independent the European Union is from the rest of the world in terms of raw materials – and the investment in green public procurement, which has not been quantified yet.

Regarding the indicators associated with the use of secondary raw materials, they are divided into two categories: the contribution of recycled materials to the demand of raw materials, and the trade in recyclable raw materials. The end-of-life recycling input rates and the circular material use rate belong to the first sub-category. The end-of-life recycling input rate indicates how much of the input of materials into the production system comes from recycled materials, whereas the circular material use indicator measures the share of material recovered and fed back into the economy. Although these indicators provide insight into the materials metabolism, they do not deliver any information about the flows of raw materials that are prevented from being extracted as a consequence of the reintroduction of the secondary materials into the production cycles. Because of the typical worse quality of the secondary materials,[61,62] a given amount of a secondary material cannot replace the same amount of virgin material to perform the same function. On the other hand, the indicators that show the trade in recyclable raw materials illustrate how the exports of secondary materials significantly exceed the imports, which gives rise to questioning the capacity of the European Union to absorb the secondary materials that it generates.

The competitiveness and innovation indicators comprise the private investment in tangible goods, job creation, the gross value added and the number of patents related to the circular economy. Finally, the waste management indicators are based on the recycling rates of specific waste streams.

Table 1.1. Indicators to monitor the progress toward a circular economy (source: Eurostat[63])

INDICATOR	EU	Spain	
PRODUCTION AND CONSUMPTION			
1. Self-sufficiency of raw materials	36.4	N/A	% (2016)
2. Green public procurement	N/A	N/A	
3. Waste generation			
Generation of municipal waste per capita	487	462	kg per capita (2017)
Generation of waste excluding major mineral wastes per GDP unit	66	62	kg per 10^3 € (2016)
Generation of waste excluding major mineral wastes per domestic material consumption	13.3	17.3	% (2016)
4. Food waste	80	N/A	x10^6 metric ton(2016)
WASTE MANAGEMENT			
5. Recycling rates			
Recycling rate of municipal waste	46.4	33.5	% (2017)
Recycling rate of all waste excluding major mineral waste	55	46	% (2014)
6. Recycling/recovery for specific waste streams			
Recycling rate of overall packaging	67.2	70.3	% (2016)
Recycling rate of plastic packaging	42.4	45.5	% (2016)
Recycling rate of wooden packaging	39.8	67.1	% (2016)
Recycling rate of e-waste	35.6	37.4	% (2015)
Recycling rate of bio-waste	81	71	% (2017)
Recycling rate of construction and demolition waste	90	79	% (2016)
SECONDARY RAW MATERIALS			
7. Contribution of recycled materials to raw materials demand			
End-of-life recycling input rates	12.4	N/A	% (2016)
Circular material use rate	11.7	8.2	% (2016)
8. Trade in recyclable raw materials			
Imports from non-EU countries	5.90	0.58	x10^6 metric ton (2017)
Exports to non-EU countries	36.72	1.30	x10^6 metric ton (2017)
Intra EU trade	52.23	5.65	x10^6 metric ton (2017)
COMPETITIVENESS AND INNOVATION			
9. Private investment, jobs and gross value added related to circular economy sectors			
Gross investment in tangible goods	0.12	0.09	% of GDP (2016)
Persons employed	1.73	2.02	% of employment (2016)
Value added at factor cost	0.98	1.02	% of GDP (2016)
10. Number of patents related to recycling and secondary raw materials	338.17	20.52	(2014)

Although these metrics are intended to reflect the implications of the circular economy at the macro level, waste management takes place at the local or regional level. Thus, municipalities are at the center of the logistic interplay between the different actors that could benefit from the adoption of circular economy strategies (waste managers, citizens, manufacturers, etc.).

Policy development for the management of municipal solid waste

The city of San Francisco constitutes the paradigmatic example of this vision. It has set into motion an ambitious plan to divert zero waste to landfills by 2020. In 2014 the city had already accomplished an impressive 80% recycling and composting rate through the implementation of measures such as financial incentives (the more mixed waste the residents generate, the more they pay for the waste management services), the ban on styrofoam, polystyrene foam and plastic water bottles, or the mandatory recycling and composting for all residents and businesses.[64]

Unfortunately, San Francisco constitutes the exception rather than the rule. Around a third of the 2.01 billion metric tons of municipal solid waste generated worldwide in 2016 was openly dumped,[64] and it is estimated that over 250,000 metric tons of waste plastic particles is floating at sea.[65] This is evidence that although waste management is crucial to achieve sustainable local communities, it also has global consequences; in fact, it accounts for about 5% of global greenhouse gas emissions.[64]

The annual municipal solid waste generation is expected to reach 3.40 billion metric tons by 2050, as a result of the rise in the world population and the income levels.[64] Hence, waste management will become a progressively pressing issue, and as such, it is tackled by different international institutions, such as the United Nations, which consider waste management the transversal connector of Sustainable Development Goals 11, focused on sustainable cities and communities, and 12, which targets responsible consumption and production.[21]

These objectives are aligned with the Europe 2020 strategy,[66] the Raw Materials Initiative[67] and the EU action plan for the circular economy,[68] which pursue sustainable growth within a “resource efficient Europe”. To facilitate the transition to more sustainable material management and a circular economy model within the European Union, Directive 2018/852[69] establishes that at least 70% of packaging waste must be recycled by 2030, whereas Directive 2018/850[70] bans landfilling separately collected waste and establishes that the fraction of municipal solid waste that is landfilled must be reduced to 10% by 2035. Moreover, Directive 2018/851,[71] which amends Directive 2008/98/EC[72] on waste, lays down measures aimed at preventing waste, reducing the adverse impacts of waste generation and management and improving resource efficiency.

Although municipal solid waste represents only between 7% and 10% of the total waste generated in the European Union, its mixed composition makes it particularly challenging to manage.[71] Directive (EU) 2018/851[71] encourages the application of the waste hierarchy (depicted in Figure 1.4), which was designed to prioritize waste prevention and management options.

As Figure 1.5 shows, the mean rates of waste disposal and material and energy recovery in the European Union follow the waste hierarchy. However, that is not the case for all the Member States. In Spain only 30% of the municipal solid waste generated in 2016 was recycled or composted, although the municipal solid waste generation rate (1.21 kg·person^{-1}·day^{-1}) was slightly lower than the average of the European Union (1.30 kg·person^{-1}·day^{-1}).[73] These statistics vary widely even within the same country. For instance, in the Spanish region of Cantabria, 41% of the municipal solid waste generated in 2014 (at a rate of 1.49 kg·person^{-1}·day^{-1}) was recycled or composted.[74]

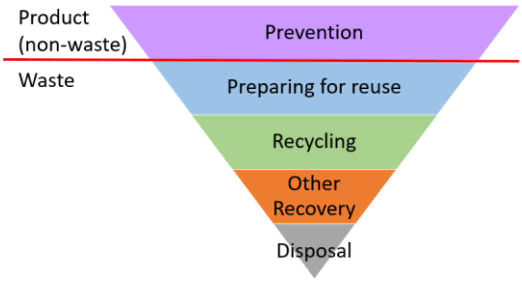

Figure 1.4. Waste hierarchy

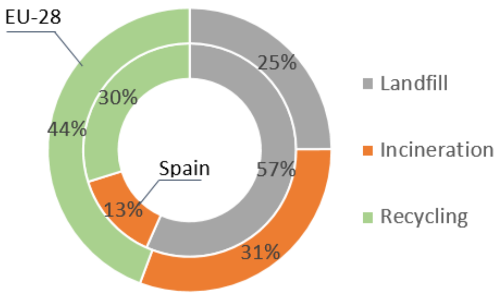

Figure 1.5. Landfill, incineration and recycling (material recycling, composting and anaerobic digestion) rates in the EU-28 and Spain in 2016 (source: Eurostat[73])

The cases of Sweden and Belgium are particularly remarkable. They have developed a strong incineration infrastructure; approximately half of the municipal solid waste generated in these countries is incinerated, whereas the other half is recycled. Consequently, they send to landfill below 1% of municipal solid waste.[73]

Figure 1.6 displays how in the period 1995-2016 the fraction of municipal solid waste disposed of in landfills within the European Union decreased as the incineration, recycling and composting rates rose, an indication of the progressive adoption of the waste hierarchy.

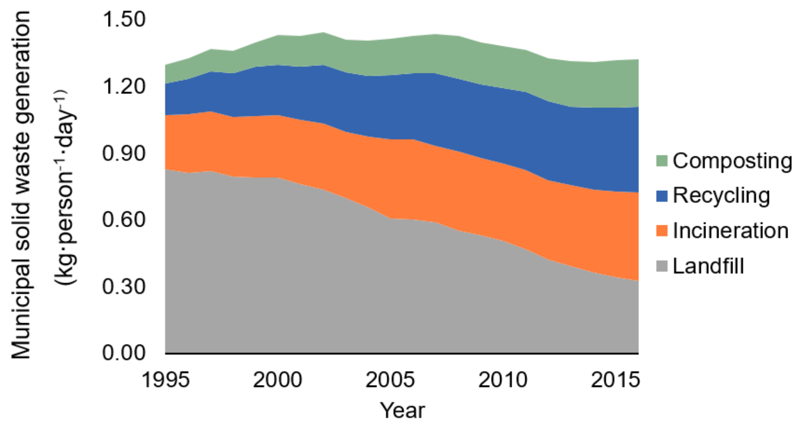

Figure 1.6. Evolution of the municipal solid waste management alternatives in the EU-28 (source: Eurostat[75])

The gradual improvement of the recycling rates is compulsory for the Member States; Directive (EU) 2018/851[71] set increasingly high recycling targets (shown in Figure 1.7) for the municipal solid waste generated in the period 2020-2035. Based on the comparison of these targets with the recycling rates of several European regions, they seem ambitious but attainable goals. Whereas some countries – like Spain – must make a considerable effort to catch up with the recycling rates of the rest of the European Union, Germany – with a 66% recycling rate in 2016 – already surpassed the 2035 recycling objective.[76]

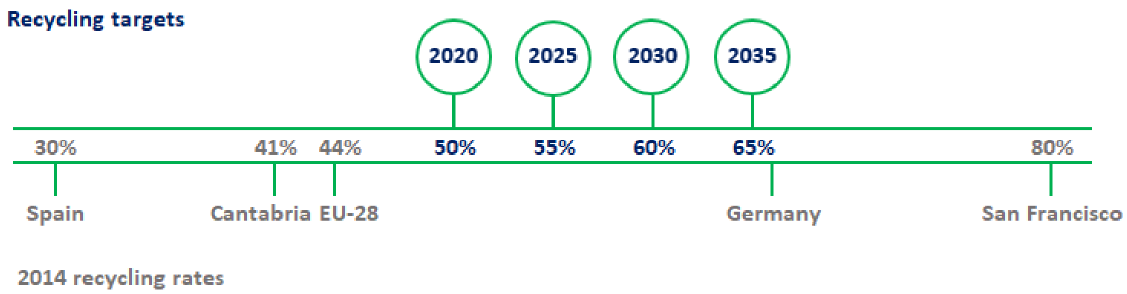

Figure 1.7. Recycling targets set by Directive (EU) 2018/851[71] (blue) and 2014 recycling rates (grey), including material recycling, composting and anaerobic digestion

Despite the fact that the motivation for this policy is clear, the term “recycling rate” is rather dubious and usually misconceived by the public. According to Directive 2008/98/EC,[72] “recycling means any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material” Additionally, Directive 2008/98/EC[72] establishes that waste ceases to be waste after it has undergone a recovery operation, if it can fulfill a function without adverse environmental or human health impacts and there is a market or demand for it.

Essentially, this means that Member States are allowed to report recycling rates on the basis of the output of sorting facilities – instead of the *actual* recycling processes – as long as there are buyers willing to purchase the sorted items. This definition opened the door to massive exportations; 90% of the plastic and 65% of the paper collected for recycling in Spain in 2011 was exported to China.[77,78]

After the Chinese 2018 import ban on materials recovered from solid waste – alleging environmental concerns and the poor quality of the imported materials – the global markets are struggling to adjust. Considering China's substantial capacity to absorb these materials – it has imported a cumulative 45% of total plastic waste since 1992 –[79] this ban will have global cascading effects on the recycled materials market, and it will certainly hamper the attainment of the recycling targets set by the European Union.

The European Union cannot pretend that waste diversion equals recycling any more. This should be seized as an opportunity to invest in new European recycling facilities and improved quality standards, as opposed to finding new markets.

To reduce the contamination levels of the secondary raw materials, the source separation of the different waste fractions that compose municipal solid waste is mandatory within the European Union; Member States are obliged to set up separate collection systems for paper, metal, plastic and glass since 2015.[71]

In addition to that, Directive 2018/851[71] states that bio-waste ("biodegradable garden and park waste, food and kitchen waste from households, offices, restaurants, wholesale, canteens, caterers and retail premises and comparable waste from food processing plants") must be either separated and recycled at source (e.g., by means of home or public composters) or separately collected from other types of waste by the end of 2023. Furthermore, as from the beginning of 2027, municipal bio-waste entering aerobic or anaerobic treatment will only count as recycled if it has been separately collected or recycled at source.[71] The divergence in the dates indicates that the policy-makers are aware that it is highly unlikely that all Member states will effectively put into practice a bio-waste source separation and collection system in such a short period of time. Although there are not any reliable statistics on the average bio-waste source separation rates, a study (summarized in Figure 1.8) found that the mean bio-waste source separation rate across the capitals of the European Union was 16% in 2014.[80]

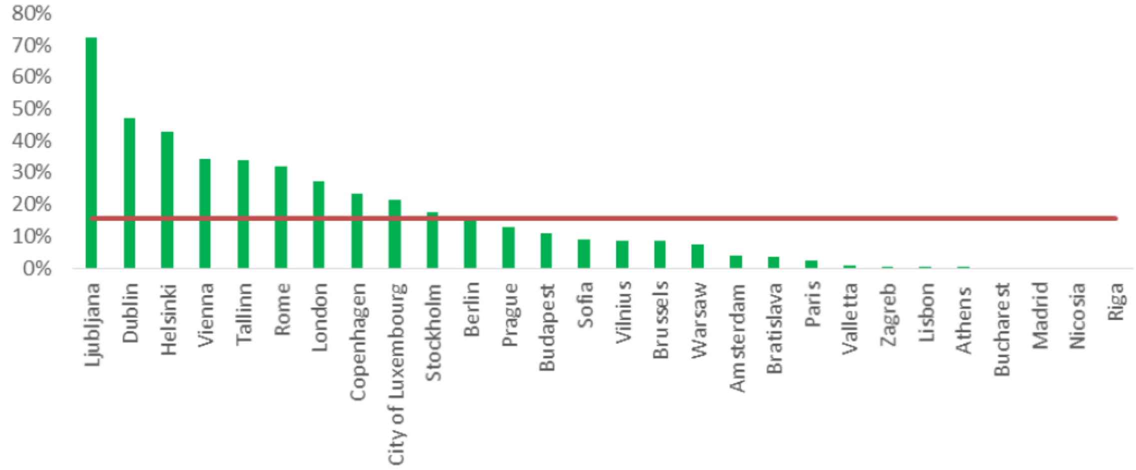

Figure 1.8. Bio-waste source separation rates in the EU-28 capital cities in 2014 (adapted from BIPRO/CRI[80])

Nonetheless, Directive 2008/98/EC[72] already encouraged Member States to ensure the separate collection of bio-waste prior to composting or anaerobic digestion. To that end, the Spanish Law 22/2011[81] about waste and polluted soil, made a distinction between the bio-stabilized material generated in the composting process of the bio-waste collected in the mixed waste stream, and the compost produced from the source-separated bio-waste. Since the application of this law, waste managers are not allowed to sell bio-stabilized material as compost.

It has been demonstrated that the source separation of bio-waste reduces the risks associated with the subsequent agricultural application of the stabilized organic material, like the transfer of heavy metals and organic pollutants to the soil.[82–84] Moreover, since fewer pretreatment operations are required for the source-separated bio-waste, the generation of bioaerosol and malodor – which is usually abated with biofilters –[85] is minimized.[83]

State-of-the-art processes for bio-waste recycling

Composting is the most widespread process for bio-waste recycling, in which aerobic microorganisms decompose organic matter, releasing CO_2, water and heat.[86] The resulting compost is a product rich in nutrients and used for soil amendment; i.e., it can improve soil quality and support plant life by increasing the soil nutrient levels.

The most typical composting systems are windrow and tunnel composting. In windrow composting, the bio-waste is laid out in parallel rows and periodically turned to enhance the oxygen diffusion through the material.[86] Tunnel composting is a more controlled process that consists of a conveyor enclosed inside a sealed casing that moves the waste across a tunnel, through which air is blown.[86]

The alternative bio-waste recycling process is anaerobic digestion. In this process, microorganisms degrade organic matter in the absence of oxygen, generating biogas, a mixture of approximately 65% vol. CH_4 and 35% vol. CO_2.[87]

The digestate (the remaining product) is dissolved in water in varying proportions, depending on whether the digester is operated in wet (total solids below 15% wt.) or dry conditions.[88] The advantage of wet operation is that it facilitates the contact between the microorganisms, the organic matter and the compounds generated in the successive biochemical reactions. However, the costs of dry anaerobic digestion are lower, because of the more energy-intensive product dehydration and the larger reactor volumes related to wet anaerobic digestion.

On the other hand, anaerobic digesters can operate either in the range of thermophilic (55 ^{o}C - 60 ^{o}C) or mesophilic (35 ^{o}C - 40 ^{o}C) temperatures. The former accelerates the kinetics of the process and reduces the presence of undesired pathogens in the digestate.[88] Nonetheless, the energy required to heat up the reactor, especially in wet systems, could be substantial in the thermophilic regime.[89]

The high moisture content of the digestate increases its transport and spreading costs, which makes it less attractive than compost for its soil amendment and fertilizing properties, although in some cases it is also hard for waste managers to find farmers interested in purchasing compost.[90–92]

The main advantage of anaerobic digestion over composting is that it also enables energy recovery, usually through the direct combustion of biogas, although it has been suggested that upgrading the biogas to biomethane for the transport sector is more environmentally beneficial.[93] However, the most efficient route to recover energy from bio-waste is incineration; the primary energy replacement of incineration is up to 16 times higher than that of anaerobic digestion.[94]

Germany is the European country where anaerobic digestion is more prevalent; the biogas produced from the anaerobic digestion of different organic matrices represented 51 TWh of the

German primary energy production in 2011, whereas it only accounted for 0.96 TWh of the Spanish primary energy supply.[95]

Although there are several small-scale plants for the anaerobic digestion of sewage sludge, manure and other wastes from the agro-food industry in Spain, only six anaerobic digestion plants for the valorization of the organic fraction of municipal solid waste are documented. They are compiled in Table 1.2.

Table 1.2. Spanish anaerobic digestion plants that handle the organic fraction of municipal solid waste

Plant	Location	Capacity (metric ton·year^{-1})
La Paloma[96]	Madrid	108,175
Las Dehesas[97]	Madrid	161,000
Ecoparc de Barcelona[98]	Barcelona	12,725
Ecoparc 2 de Montcada i Reixac[99]	Barcelona	100,000
Can Barba[100]	Barcelona	25,000
Ecoparque Gran Canaria Norte[101]	Gran Canaria	21,000

To maximize nutrient recovery from the products derived from anaerobic digestion, several technologies in different stages of development are currently being investigated.[102–104] Among them, the recovery of ammonia (NH_3) as ammonium sulfate ($(NH_4)_2SO_4$) by air stripping, and struvite ($NH_4MgPO_4 \cdot 6H_2O$) precipitation by means of the addition of magnesium compounds, are deemed feasible methods to recover nutrients from the liquid digestate (the liquid phase that is separated from the solid fraction of the digestate).[104] The latter has been proven successful at the recovery of P and N from the anaerobic digestion effluents of the potato processing and dairy industry.[105]

The potential of nutrient recovery

It has been estimated that the nutrients present in the food waste collected in municipal bio-waste and half of the food waste generated by other sources could replenish 4% of N, P and potassium (K) used in chemical fertilizers worldwide.[35] That is a minor fraction compared to the total flows of N, P and K contained in food, animal and human waste streams, which altogether represent 2.7 times the nutrients present in the industrial fertilizers currently used.[106]

Therefore, nutrient recovery constitutes an opportunity to restore the natural fluxes of the biogeochemical cycles, reducing the excessive removal and release of nutrients at different stages.[48] Moreover, the proper management of the nutrients recovered from organic waste could play a significant role in securing the food supply for the growing population, which will become progressively challenging as the global phosphate reserves diminish.[107]

Recent studies concluded that the optimization of the current nutrient management practices is a condition to feeding the future generations in a sustainable manner, although the synergistic combination of multiple strategies, such as dietary shifts, precision agriculture, increasing water use efficiency and reducing food waste will be required.[108–110]

Mueller et al.[110] found that closing the yield gaps of agricultural systems to 100% of attainable yields could increase the worldwide production of corn, wheat and rice 64%, 71% and 47% respectively. Figure 1.9 shows the predominant factors – fertilizer use and irrigation – that deter different regions from reaching the maximum yield of corn; 73% of the underachieving areas could close cereal yield gaps by increasing nutrient inputs.

Nonetheless, they estimated that by eliminating fertilizer overuse, the global N and P application on corn, wheat and rice could be reduced by 28% and 38% respectively without decreasing current yields.

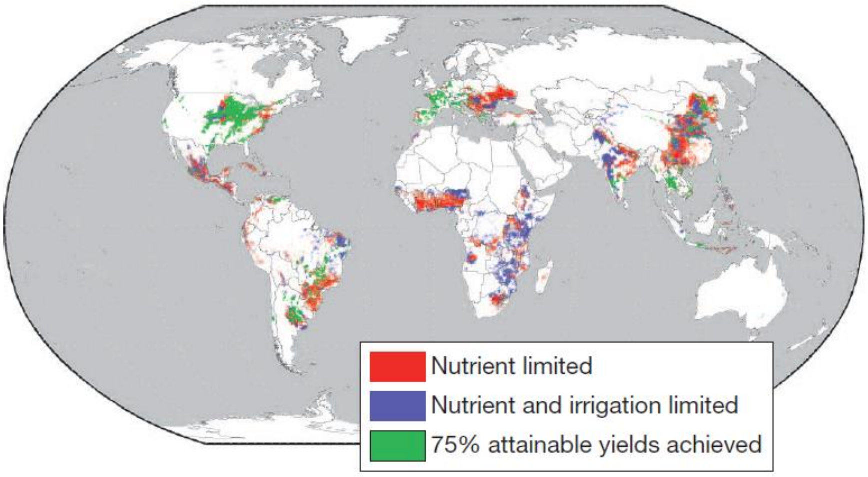

Figure 1.9. Management factors limiting yield-gap closure to 75% of the attainable yield of corn in different regions (adapted from Mueller et al.[110])

These results corroborate that sustainable agriculture relies on a delicate equilibrium: too few nutrients lead to low crop productivity, but too many nutrients are the cause of environmental pollution.[108]

Thus, to fully understand the implications of the exchange of nutrients between food and waste systems, robust systems engineering tools that enable the comprehensive design, analysis, simulation and optimization of complex systems are needed.

THE CONTRIBUTIONS OF PROCESS SYSTEMS ENGINEERING TO SUSTAINABILITY

A system is an entity encompassing a set of interacting parts and delimited from the environment by its boundary. The inputs to the system represent the influence of the environment on the system, and the outputs reflect the behavior of the system.[111]

Process systems engineering applies systems thinking, which provides a holistic framework to address complexity, to the engineering of systems comprising physical, chemical and/or biological processing operations.[112] Analysis and synthesis are the cornerstone of process systems engineering.[111,112] Systems analysis aims at understanding the behavior of systems,[113] whereas process synthesis consists in integrating a process flowsheet that will convert the given inputs into outputs to meet a series of specifications.[113]

Some of the standard process systems engineering methods, like process integration – a common practice in the industrial sector to minimize resource consumption by reusing secondary streams carrying residual heat, cold, mechanical work or water – could be easily applied to further exploit the value of waste materials and components.[114]

The automation of process integration by means of the combination of life cycle assessment (LCA) – the methodology that quantifies the environmental impacts of products, processes and services throughout their entire life cycle – and optimization-based process synthesis should guarantee, as long as the problem is properly formulated, the minimization of the overall environmental impacts of the designed processes.[115,116] Failing to account for the environmental impacts of a process across its entire life cycle may lead to solutions that decrease the local environmental impacts of the process at the expense of increasing the environmental burdens in other stages of its life cycle.[116]

Since environmental improvements usually entail economic costs, a compromise between the economic and environmental performance of systems is desirable. The combination of LCA and multi-objective optimization – a mathematical programming technique to concurrently optimize multiple objective functions – enables decision-makers to balance the environmental and economic objectives.[117,118]

Solving such complex multi-criteria problems was computationally impossible a few decades ago; it is now feasible because of the recent advances in optimization theory and software applications.[115] The first published paper describing how the LCA methodology is embedded within an optimization framework dates back to 1995.[119] Two years later the same authors applied multi-objective optimization to determine the optimal design and scheduling of processes from the dairy industry.[120] This early research demonstrated that LCA can successfully be combined with optimization techniques to satisfy different sustainability criteria.

The improvement in the computer-aided systems engineering capabilities, and specially the development of novel optimization algorithms capable of solving increasingly more complex problems, is likely to accelerate the synthesis of sustainable processes in the future.[112,116]

Furthermore, experts agree that the research frontiers in process systems engineering will broaden the scope of the systems under study beyond those that are traditionally considered to pertain to the chemical engineering discipline. Therefore, the process systems engineering tools aimed at finding model-based solutions to systems problems could play a major role in addressing some of the challenges faced by humanity today.[111,112]

The development of models that simultaneously describe the behavior of natural and technical systems is a particularly hard task, because of its multi-disciplinary nature, but it could significantly improve our understanding of the consequences of human activities on the environment. Examples of this type of comprehensive models are the REMIND-R model, which comprises a macroeconomic, an energy and a climate module,[121] or the MAgPIE model, which links agricultural production to its impact on the environment,[122] both developed at the Potsdam Institute for Climate Impact Research.

Planning for the sustainable and coordinated management of waste and resources will require a consistent methodological framework – still not developed – that reflects the interactions between the natural ecosystems that provide resources and receive environmental impacts and the technical systems where resources are processed, consumed and managed as waste.

OBJECTIVES OF THE THESIS

This dissertation, motivated by the urgent need to find sustainable solutions to the problems described in this chapter, pursues two main objectives:

- To propose a methodological framework to determine the optimal configuration of integrated waste and resource management systems under a life cycle perspective, facilitating the decision-making processes.

- To test the hypothesis that the implementation of a circular economy is a valid strategy to achieve a more sustainable production and consumption model in terms of resource consumption, environmental impacts and economic benefits.

The second objective will be accomplished by means of the application of the proposed methodological framework to a case study: the management of municipal organic waste (the bio-waste generated by households and private businesses and collected by municipalities) in the Spanish northern region of Cantabria, which has a population of 580,300 and an area of 5,326 km^2.[123] The selected case study is of interest because at the time of writing the regional waste management system needs to be retrofitted to comply with the European legislation.

To attain these two main objectives, five specific objectives were set:

1. To define the characteristics and boundaries of "Circular Integrated Waste Management Systems" (CIWMSs) aiming at resource recovery.

2. To identify the strengths and weaknesses of the methodologies deployed for the design and analysis of integrated waste management systems that handle municipal solid waste, and how they can be applied to the design and analysis of sustainable CIWMSs.

3. To design a superstructure containing alternative unit processes for the treatment and valorization of municipal organic waste and nutrient management.

4. To formulate an optimization problem based on the model of the superstructure that quantifies the resource consumption and the environmental and economic performance of the unit processes.

5. To propose an indicator to measure the circularity of waste components within CIWMSs.

Regarding the structure of the document, Chapter 2 covers the followed methodological approach and the basics of the applied tools. Each subsection of Chapter 3 (Chapters 3.1 – 3.4) correspond to one of the four published papers.

Chapter 3.1 is based on the findings of a critical literature review, and it addresses specific objectives 1 and 2. The practical implementation of the methodological framework proposed in Chapter 3.1 is described in Chapters 3.2 – 3.4; the basic model of the system under study is first introduced in Chapter 3.2, and it is progressively improved in Chapters 3.3 and 3.4, fulfilling specific objectives 3 and 4. The circularity indicator targeted by specific objective number 5 is presented in Chapter 3.3.

The problem formulated in Chapter 3.2 aims at minimizing the carbon footprint, the land use and the consumption of raw materials of the system, whereas Chapter 3.3 explores the trade-offs between the minimization of the climate change and eutrophication impacts, and the maximization of nutrient circularity within the system. Chapter 3.4 focuses on the economic dimension of the model; it investigates how improving nutrient circularity affects the profitability of the system. Finally, Chapter 4 summarizes the conclusions and limitations of the research.

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CHAPTER 2

METHODOLOGY AND FUNDAMENTALS

"If I have seen further, it is by standing upon the shoulders of giants."

Isaac Newton, English physicist and mathematician (1642-1727)

The methodology followed to attain the objectives of the thesis is based on these steps:

- Definition of the system boundaries.
- Development of a model that describes the system behavior.
- Formulation of the optimization problem.
- Interpretation of results.

The tools applied throughout the thesis and the basic concepts on which they rely are described in this section.

MATHEMATICAL PROGRAMMING

To optimize the synthesis of a process system, a superstructure – a representation containing all the alternative designs of the system –[1] must be developed. The model describing the superstructure must satisfy the mass and energy balances and the capacity constraints imposed by the topology of the superstructure. The solution of the optimization problem will determine the value of the variables that optimize the defined objective function. They are either discrete variables that represent the unit processes within the superstructure that should integrate the flowsheet, or continuous variables that indicate the size and the operating conditions of the unit processes.[1]

This type of problem formulation leads to MIP (Mixed Integer Programming) problems, which can be linear (MILP) or non-linear (MINLP). The disadvantage of working with non-linear programming (NLP) problems is that they may be non-convex, which gives rise to multiple local optima instead of a single global optimum. NLP problems are convex if they have a convex objective function and a convex feasible region.[2] The difference between convex and non-convex functions is illustrated in Figures 2.1 and 2.2. In the latter, where a convex non-linear function is represented, the global minimum can be clearly identified.

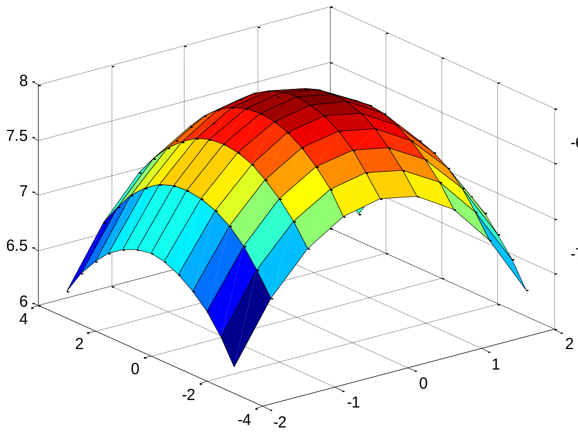

Figure 2.1. Non-convex function

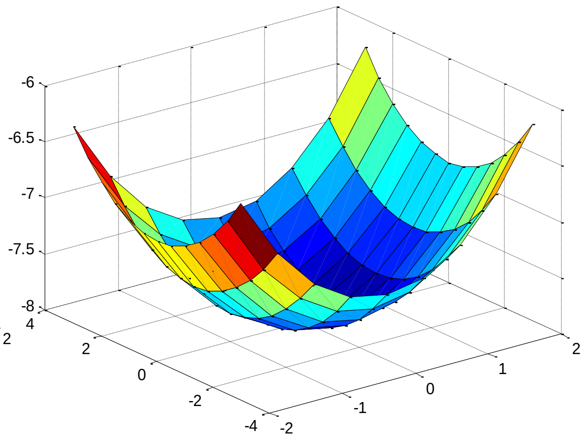

Figure 2.2. Convex function

Most methods to solve MIP and NLP problems rely on the branch and bound algorithm,[2,3] which divides the feasible region of continuous variables into sub-regions. The sub-problems associated with each sub-region are sequentially solved. Then, the solutions to each sub-problem are compared and the non-optimal solutions are eliminated.

There is a variety of commercial modeling platforms for the optimization of problems based on algebraic equations, such as GAMS, AMPL or AIMMS.[3] The model presented in this dissertation was implemented in GAMS (General Algebraic Modeling System) 24.7.1.,[4] which offers a wide range of solvers, i.e., computer codes for solving specific types of optimization problems.

The CPLEX solver, deployed in this research, is a variation of the branch and bound algorithm in which the branching of the variables is performed using cutting planes to restrict the size of the solution domain.[5]

Multi-objective optimization

Multi-objective optimization is a mathematical programming tool that finds, among the entire set of feasible solutions of the problem, those that are better than the others in at least one objective.[6] The solutions of a multi-objective optimization problem are known as Pareto-optimal, Pareto-efficient, non-inferior or non-dominated solutions. The image of the efficient solutions is called Pareto front, Pareto curve, or Pareto surface, and its shape indicates the nature of the trade-off between the objective functions.[6]

The Pareto front resulting from the minimization of two hypothetical objective functions is shown in Figure 2.3, where the points that delimit the curve, A and B, are the solution to the single-objective optimization of objective functions f_1 and f_2 respectively. The region below the Pareto curve comprises the set of infeasible solutions (among which is the ideal point that minimizes both objective functions), whereas the region above the Pareto front contains the sub-optimal solutions. The worst possible solution, the Nadir point, represents the upper bounds of the objectives in the Pareto-optimal set.[6]

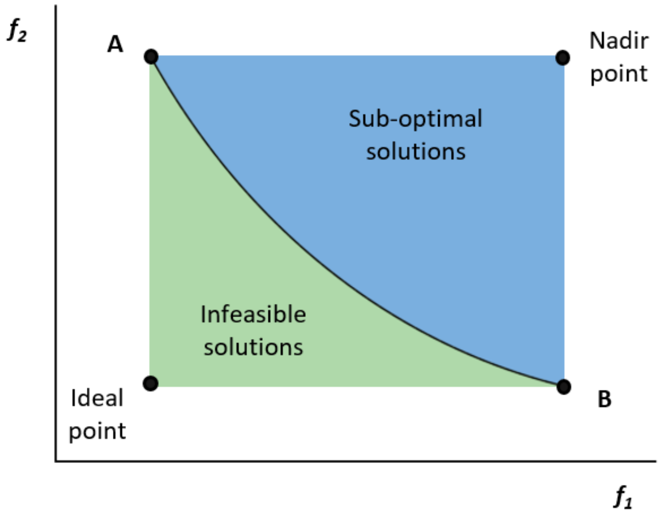

Figure 2.3. Pareto front

There are numerous strategies to solve multi-objective optimization problems. The Ɛ-constraint method, applied in this thesis, is one of the most commonly used methods.[7] To solve a multi-objective optimization problem with k objective functions by means of the Ɛ-constraint method, first each objective function is optimized separately. The solutions to these k single-objective optimization problems (points A and B in Figure 2.3) show the ranges of values that each objective function can take within the set of Pareto-optimal solutions.

Then, one objective function is selected, and the others are transformed into inequality constraints. The set of Pareto-optimal solutions is obtained solving this single-objective optimization problem *n* times, for different equally spaced values of the upper bounds of the inequality constraints within the ranges of values that the objective functions can take.

The number *n* of single-objective optimization problems that must be solved for a multi-objective optimization problem with *k* objective functions, in which *p* values are provided for the upper bound of each inequality constraint, is calculated with Equation 2.1:[7]

$$n = k + p^{k-1} \qquad \text{(Equation 2.1)}$$

Therefore, the computational complexity of the Ɛ-constraint method grows exponentially with the number of objectives. Another drawback of this method is that some of single-objective optimization problems may lead to unfeasible or repeated Pareto solutions, which is the reason it is increasingly hard to visualize the trade-offs between the objective functions as the number of objective functions increase.

LIFE CYCLE THINKING

Life cycle thinking is the conceptual framework that provides a system-level holistic view of the production and consumption processes involved in the life cycle of products and services. Several methodologies based on the life cycle thinking approach have been developed to address the different sustainability dimensions, the most applied of which is life cycle assessment (LCA).

LCA is a standardized systematic and iterative method to quantify the emissions, resource consumption and environmental impacts associated with goods and services throughout their entire life cycle; i.e., from the cradle (resource extraction) to the grave (waste disposal).[8-10] Figure 2.4 summarizes the phases of an LCA and the iterations between them.

The first step of an LCA, the goal and scope definition, requires establishing the functional unit (the reference to which all the inputs and outputs of the life cycle inventory and the subsequent analyses are related), the time and space limits of the analysis and the unit processes comprised within the system under study. This stage also involves selecting an LCA modeling approach (attributional or consequential), and strategies to address multi-functionality (if needed), which are discussed in Chapter 3. After the emissions and resources consumed by each unit process are quantified in the life cycle inventory, an impact assessment is performed.[9]

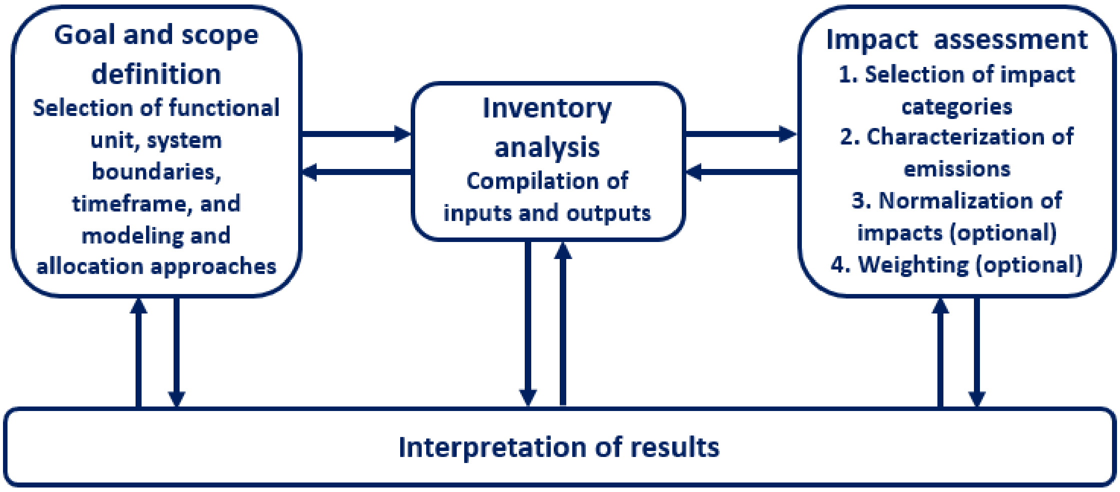

Figure 2.4. Overview of the LCA phases (adapted from ISO 14040[8])

In the life cycle impact assessment phase, the environmental burdens emitted and the resources consumed by the studied system are aggregated into impact categories, according to the substances ability to contribute to each impact category, and they are then converted into indicators (e.g., kg CO_2-eq) by means of characterization methods. These characterization methods model the impact of each emission according to the underlying environmental mechanisms. They are based on the definition of substance-specific and time-dependent characterization factors that express the impact of elementary flows in terms of the unit of the category indicators.[10]

The existing characterization methods are not fully harmonized yet, and different characterization methods do not consider the same impact categories. Many methods make a distinction between midpoint indicators, which focus on single environmental problems (climate change, eutrophication, acidification, ozone depletion, etc.), and endpoint indicators, which show the environmental impact on higher aggregation levels, such as the effect on human health, biodiversity or resource depletion.[11]

The impact categories analyzed in this dissertation by means of ReCiPe 1.11, one of the most common characterization methods, are climate change, marine eutrophication and freshwater eutrophication. This method lets the life cycle practitioner select one of the three pre-defined cultural perspectives (individualist, hierarchist and egalitarian) associated with different time horizons (20, 100 and 500 years), according to which the characterization factors are calculated.[11] The most frequently used hierarchist perspective was applied in this thesis.

Several LCA models of waste management systems have been developed since the 1990s (ORWARE, WRATE, WISARD, etc.).[12] The EASETECH (Environmental Assessment System for Environmental Technologies) model, implemented in the EASETECH 2.3.6 software,[13] was selected to perform the LCA of the waste management unit processes studied in this dissertation. It is one of the most widely used waste LCA models, and besides providing LCA results based on the waste composition, it can perform material flow analyses (MFA);[14] i.e., it determines the quantity of each component in all the elementary flows of the system.

Regarding the application of the life cycle thinking framework to economic and social analyses, life cycle costing (LCC)[15,16] and social LCA[17] methodologies have been developed, although they are still not standardized. They rely on the quantification of costs and social indicators in all the stages of the life cycle.

Nonetheless, the definition of the goal and scope of the LCC analysis might differ from those of the LCA, since several stakeholders with opposed economic interests might be involved in different stages of the life cycle. Beyond considering the divergent economic interests of the actors involved in the case study, the social dimension of sustainability was not directly assessed in this thesis. The economic analysis of the studied system was based on the LCC models implemented in SWOLF (Solid Waste Optimization Lifecycle Framework), developed by researchers at North Carolina State University.[18]

BIOGEOCHEMICAL MODELING

The biogeochemical cycle of a chemical element comprises all its transport and transformation pathways throughout the ecosystems. According to Li,[19] a biogeochemical model is a mathematical expression of the spatially and temporally differentiated environmental forces that drive biogeochemical reactions in ecosystems.

Modeling biogeochemical cycles is extremely challenging because of the large number of environmental variables causing biogeochemical reactions, and the complex feedbacks and interactions between them.[19] Moreover, many soil emissions are the result of microbial processes that exhibit a high degree of temporal and spatial variability.[20] The main biogeochemical processes relevant to this dissertation are outlined here.

When the soil organic C decomposes, C is partially lost as CO_2. Dissolved organic carbon (DOC) is produced as an intermediate during decomposition, and it can be immediately consumed by the soil microbes as the basic material for cell synthesis and energy. Meanwhile, the decomposed N is partially mineralized by microbes to NH_4^+, which is then subject to nitrification. NH_4^+ and NO_3^- are subsequently taken up by plants as they grow.[21]

Nitrification – the main soil process contributing to the production of NO and N_2O – is the microbial oxidation of NH_4^+ to NO_2^- and NO_3^- under aerobic conditions, whereas under the anaerobic conditions caused by rainfall or irrigation, NO_2^- and NO_3^- are reduced to N_2 by denitrifying bacteria.[19]

On the other hand, CH_4 is a product of the biological reduction of CO_2 or organic carbon mediated by anaerobic microbes that are only active when the soil redox potential is very low. Conversely, CH_4 is oxidized by aerobic methanotrophs in the soil.[19]

A few models describing nutrient biogeochemical cycles have been developed in the last three decades (DAYCENT,[22] Daisy,[23] etc.) Among them, DNDC (Denitrification-Decomposition) has been reported to be extensively tested.[24]

The DNDC 9.5 graphical interface was used in this thesis to predict the crop yield, C sequestration, nitrate and P leaching losses and emissions of C and N gases that arise from different nutrient management strategies. DNDC is a process-oriented simulation model of C and N biogeochemistry in agroecosystems first described in 1992 by Li et al.[25] The model does not account for the complete P biogeochemical cycle; however, it describes the daily soil P profile, the crop demand and uptake of P, and the P loss through leaching flows.[24]

The model consists of two components. The first component is composed of three sub-models: soil climate, crop growth and decomposition, whose input parameters are the characteristics of the ecological drivers (climate, soil, vegetation and farming practices). The integration of these sub-models predicts daily soil temperature, moisture, pH, redox potential (Eh) and substrate concentration profiles (DOC, NH_4^+, NO_3^- and NO_2^-).

The second component, consisting of the nitrification, denitrification and fermentation sub-models, predicts daily fluxes of CO_2, CH_4, NH_3, NO, N_2O and N_2 based on the soil environmental variables calculated in the first component of the model.

The model, whose structure is depicted in Figure 2.5, is based on physico-chemical and biological equations, as well as empirical equations generated from laboratory studies.

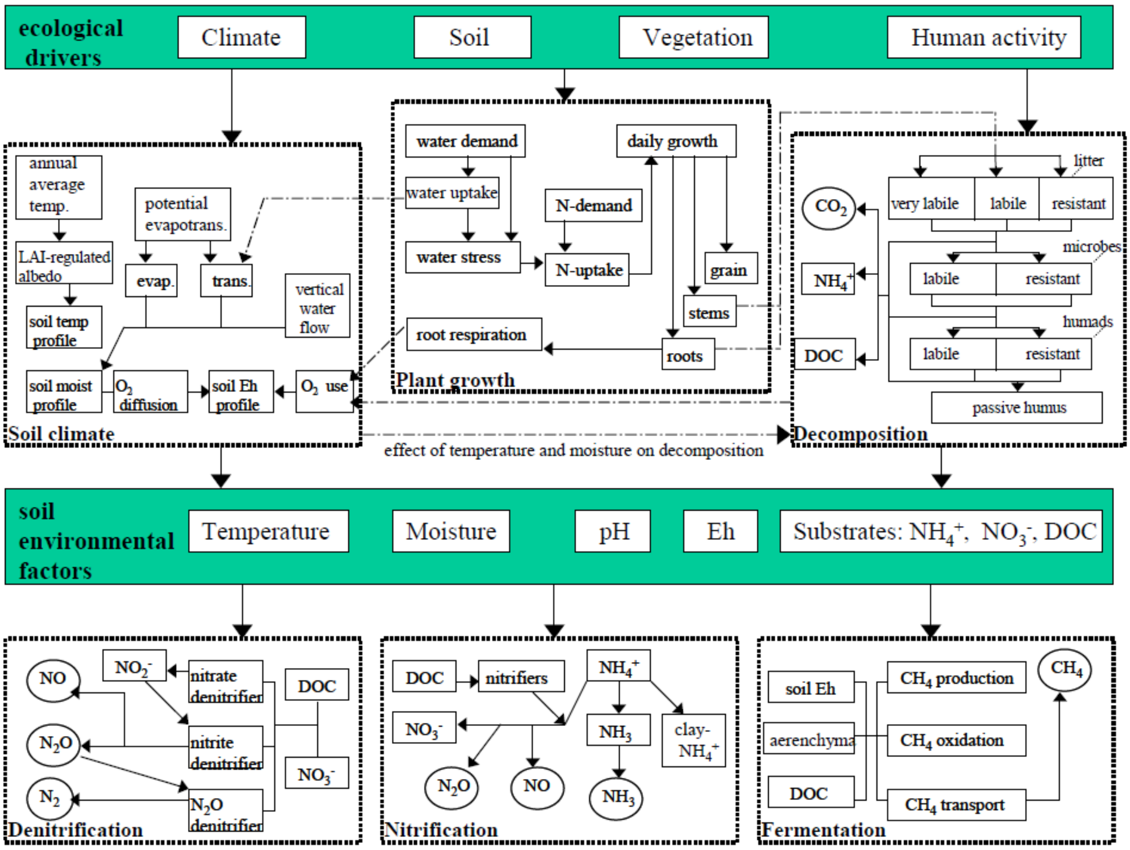

Figure 2.5. Overview of the DNDC model structure (adapted from the DNDC user's guide[26])

METHODOLOGICAL SEQUENCE

To integrate the tools and methodologies described in this chapter within the same framework, a bottom-up mechanistic model of the studied system was developed combining the selected MFA, LCA, LCC and biogeochemical modeling tools, and an optimization problem based on the model was subsequently formulated. The flows of information between the software, which were not automated, are summarized in Figure2.6.

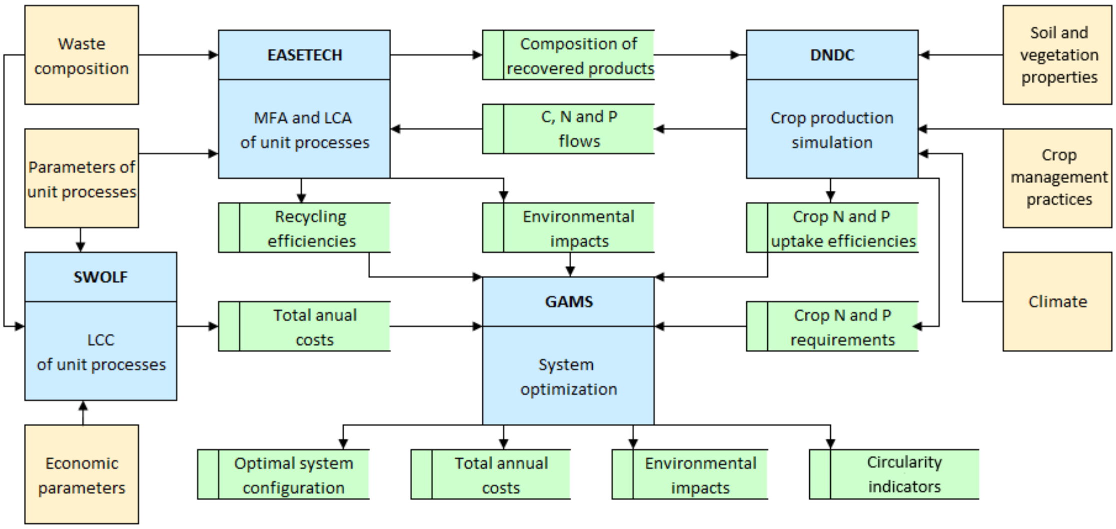

Figure 2.6. Data flow diagram

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CHAPTER 3

RESULTS AND DISCUSSION

"We cannot solve our problems with the same thinking we used when we created them."

Albert Einstein, German/American physicist (1879-1955)

The system analysis tools applied to design and assess the performance of linear Integrated Waste Management Systems were reviewed in Chapter 3.1 in order to identify the weak spots of these methodologies, the difficulties of applying them to Circular Integrated Waste Management System (CIWMSs), and the topics that could benefit from further research. The findings of the literature review provided the basis to develop a methodological framework for the analysis of CIWMSs.

This methodological framework was subsequently applied to a case study, the management of municipal organic waste in Cantabria. Figure 3.1– elaborated with the STAN (subSTance flow ANalysis*)* 2.6.801 software –[1] depicts the configuration of the mechanical biological treatment plant located in Cantabria in 2014, and the estimated flows of carbon (C), nitrogen (N) and phosphorus (P) associated with the mixed waste processed in the facility.

The model of a superstructure comprising alternative unit processes to manage the municipal organic waste generated in Cantabria was presented in Chapter 3.2. The flows of organic waste that are sent to each unit process were optimized according to these objective functions, which were minimized: the carbon footprint of the system, the occupied landfill area and the consumption of non-renewable raw materials. The multi-objective optimization of the problem proved that increasing the circularity of resources does not necessarily entail that the overall consumption of natural resources and the emission of environmental burdens of the system decrease.

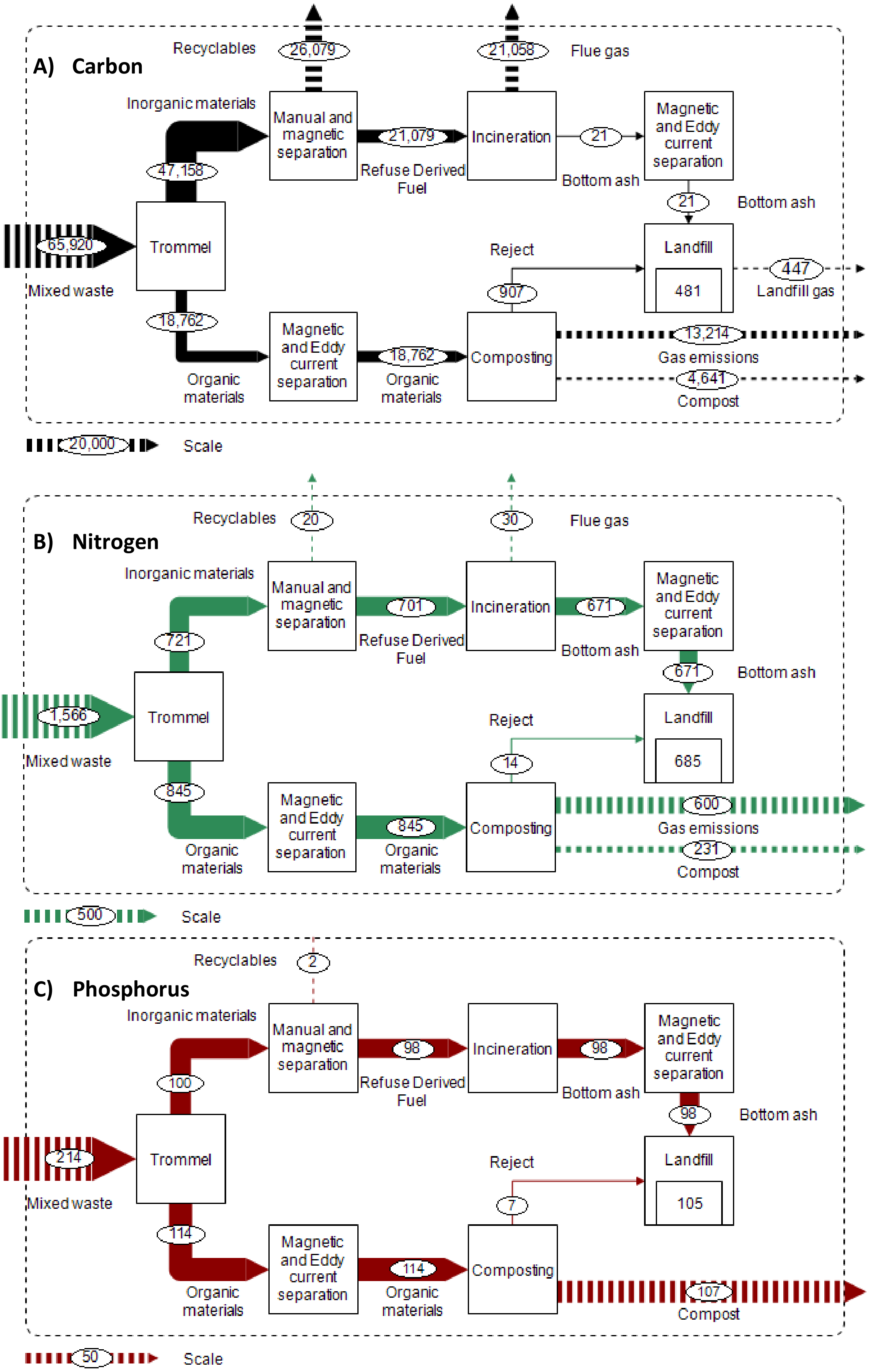

Figure 3.1. Flows of C (Figure 3.1.A), N (Figure 3.1.B) and P (Figure 3.1.C) in the Cantabrian mechanical biological treatment plant (ton·year^{-1}). Estimation performed with the data from the Cantabrian waste management plan[2] and the elemental composition of the waste components provided by the EASETECH (Environmental Assessment System for Environmental Technologies) 2.3.6 software.[3]

This idea was further investigated in Chapter 3.3, where an indicator that quantifies the circularity of waste components was proposed. In this section the model was expanded to include different unit processes for the application of the recovered products as soil amendment. The problem was optimized according to six objective functions: the circularity indicators of C, N and P, which were maximized, and their associated environmental impacts (global warming, marine eutrophication and freshwater eutrophication), which were minimized.

Finally, the economic consequences of enhancing nutrient circularity within the studied system were analyzed in Chapter 3.4. The model was optimized to find the system configurations that minimize the total annual cost and the global warming impacts, and maximize the circularity indicators of N and P.

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CHAPTER 3.1

CIRCULAR INTEGRATED WASTE MANAGEMENT SYSTEMS

"There are no separate systems. The world is a continuum. Where to draw a boundary around a system depends on the purpose of the discussion."

Donella H. Meadows, American environmental scientist (1941-2001)

Chapter 3.1 corresponds to the following paper:

Cobo, S.; Dominguez-Ramos, A.; Irabien, A. From linear to circular integrated waste management systems: A review of methodological approaches. *Resour. Conserv. Recycl.* **2018**, *135*, 279-295; DOI: 10.1016/j.resconrec.2017.08.003.

"Resources within planet Earth are finite by nature. Natural resources whose formation roots in other geologic periods, like mineral deposits, cannot be renewed in human timescales and thus their reservoirs are bound to eventually become depleted if their consumption continues.[1, 2] On the other hand, natural stocks subject to biological cycles (a population of trees for example) yield a sustainable flow of valuable goods and services (such as wood and CO_2 removal from the atmosphere) on a continuous basis.[3] Nonetheless, since the early 1970s some renewable natural resources are being exploited faster than they can be renewed.[4] As a matter of fact, it would take 1.64 planets to regenerate in one year the natural resources consumed in 2016.[5] This figure is expected to worsen because of the projected population increase and the improved acquisition levels of the emerging economies.[6,7]

If the consumption of raw materials rises, so does waste generation.[8] Around 1.3 billion metric tons of Municipal Solid Waste (MSW) are annually produced in cities all over the world,[9] and a significant amount of the waste produced in low and lower-middle income countries is disposed of in open dumps[9] lacking measures to prevent safety and environmental hazards. Under the assumption that every metric ton of MSW generated in cities worldwide could be stored in 1 m^3 of sanitary landfill,[10] a landfill volume equivalent to that of 347,000 Olympic swimming pools would be required every year. Accordingly, policies against landfills are mostly motivated by a lack of space, particularly in the highly populated areas of Europe and Asia, where landfills are more likely to interfere with other land uses like agriculture.[11]

In fact, waste valorization might help us overcome one of the most pressing global challenges: securing the food supply. Waste has been suggested as a plausible source to recover phosphorus,[12,13] an essential nutrient to the metabolism of plants and by extension to agriculture, whose remaining accessible reserves could run out as soon as 50 years from now.[14]

Hence, as the principles of industrial ecology dictate, resource and waste management are key to meeting the future needs of society in a sustainable manner. Waste prevention activities or policies such as restricting planned obsolescence in electronic products and measures like minimizing product weight or design for disassembly[15] will contribute to tackle these issues.

A reduction in the consumption of natural resources and the amount of waste generated could also be accomplished if a shift to circular economic and production systems, mimicking the self-sustaining closed loop systems found in nature, such as the water cycle, was put into practice. A circular economy aims at transforming waste back into a resource, by reversing the dominant linear trend of extracting, processing, consuming or using and then disposing of raw materials, with the ultimate goal of preserving natural resources while maintaining the economic growth and minimizing the environmental impacts.[16,17]

In a circular economy the reduction in the environmental impacts, such as global warming, is due to the improvement in resource and energy efficiencies. For instance, it has been demonstrated that the production of secondary aluminum from scrap consumes less than 5% of the energy needed in the production of primary aluminum;[18] this entails that the emission of up to 19 metric tons of equivalent CO_2 to the atmosphere could be avoided per metric ton of aluminum that is recycled instead of produced from the mineral ore.[19]

Given all the benefits that the circularity of resources has to offer, the reasonable question to pose is how society and industry can successfully transition to a circular economy. The straightforward answer from an engineering point of view is through the design of efficient Circular Integrated Waste Management Systems (CIWMSs) that link resource processing and waste treatment, and allow the potential of waste to be fully exploited. A CIWMS is expected to produce not only materials, but also energy and nutrients; additionally, it could deliver certain chemicals.

Therefore, a trade-off between the functions of a CIWMS is unavoidable. A thorough analysis must be carried out prior to the design stage of a CIWMS so that it can assist in the decision-making process. As the analytical framework supported by systems thinking can provide a holistic view on the sustainability challenges that arise from the interconnections between the components of an Integrated Waste Management System (IWMS),[20,21] so far manifold papers applying a systems-oriented approach to waste management have been published.

That is the reason only the most recent papers focusing on the analysis of IWMSs have been addressed in this study. The aim of this chapter is to conduct a critical and comprehensive review of the studies published since 2011 that analyze IWMSs whose input is MSW, in order to gain insight into the strengths and shortcomings of the methodologies currently being applied, and to identify their applicability to a sustainable CIWMS targeting resource recovery. To the best of the authors' knowledge, an IWMS has never been analyzed from the perspective of a circular economy before. The novelty of this review is that the characteristics of a CIWMS are defined, the potential pitfalls of applying the current methodologies deployed in the analysis of linear IWMSs to a CIWMS are identified and possible methodological improvements are proposed.

This review is structured as follows: first, the methodology applied in the selection of the reviewed papers is described. Second the state-of-the-art technologies and processes for IWMSs are outlined, along with their potential restraints to the development of a circular economy. Third, the characteristics of a CIWMS are defined. Next, the methodologies currently applied to analyze IWMSs are briefly described and the hottest topics regarding the methodological aspects of the analysis of IMWSs are subsequently identified. Finally, the conclusions drawn from the findings of the study are summarized, with special emphasis on the Life Cycle Assessment (LCA) methodology.

METHOD

77 papers analyzing IWMSs that treat MSW and published after 2010 were identified by means of the Scopus database.[22] They are listed in Table 3.1.1 (at the end of the chapter). The systematic review method was conducted applying three different keyword strings: i) *municipal solid waste*, *integrated*, *system* and *analysis,* ii) *municipal solid waste*, *integrated*, *system* and *methodology*, iii) *municipal solid waste*, *integrated*, *system* and *(sustainable* or *sustainability)***.**

The papers focusing on the analysis of scenarios regarding alternative waste treatment technologies or processes were excluded from the review.

Once the technological obstacles faced by CIWMSs and the limitations of the methodologies applied for the analysis of IWMSs were detected in the reviewed studies, the search criteria were expanded to cover the specific topics of interest. Those additional papers are listed throughout the document.

TECHNOLOGICAL BACKGROUND

Prior to the proposal of guidelines for the analysis of CIWMSs that enhance the circularity of resources and enable the transition to a circular economy, it is mandatory to recognize the technological restrictions to the implementation of such a system. They are outlined in this section.

Quality and value of recycled materials

The market penetration of recycled materials is highly dependent on their physical and chemical characteristics, which will determine their price. However, not all the existing recycling technologies enable a fair competition between virgin and secondary materials, because their quality might differ.

Recycling technologies either downgrade or upgrade the materials in respect to the quality of the virgin materials. Downgrading implies that the properties of the recycled material are not as good as those of the virgin material. Instead, upgrading technologies improve the quality of the waste materials at least up to the quality of the virgin materials.

In closed-loop recycling, the material is recycled into the same product system and the inherent properties of the recycled material are maintained virtually identical to those of the virgin material. Oppositely, in open-loop recycling the material is recycled into a different product system and its inherent properties may or may not differ to those of the virgin material.[23] Closed-loop recycling is not equivalent to infinite recycling; materials can be used and later recycled within a closed-loop system for a number of times, until microstructural changes in the material or the accumulation of chemical elements and compounds hamper its further reuse.[24]

A case of closed-loop recycling occurs when a glass bottle is recycled into a glass jar, because the glass jar could be recycled back into a glass bottle with the same functionality as the original one,[25] whereas recycling PET bottles into PET fibers is an example of open-loop recycling;[26] it is an irreversible process.

Recycling processes can be further classified as downcycling or upcycling processes. Downcycling has been defined as the recycling of materials into lower value products.[27] The use of wrought scrap in cast products, due to their ability to accommodate higher silicon contamination, is considered downcycling. On the contrary, if the waste materials are recycled into products of higher value, the recycling process is called upcycling.[28] Upcycling involves a change in the fundamental properties of the material, like its physical structure or its chemical composition. Novel approaches to upcycling described in the literature entail chemical[28, 29] or biological transformation.[30] Figure 3.1.1 compiles the types of recycling processes according to the quality of the recycled materials and the value of the resulting recycled products in respect to the original materials and products.

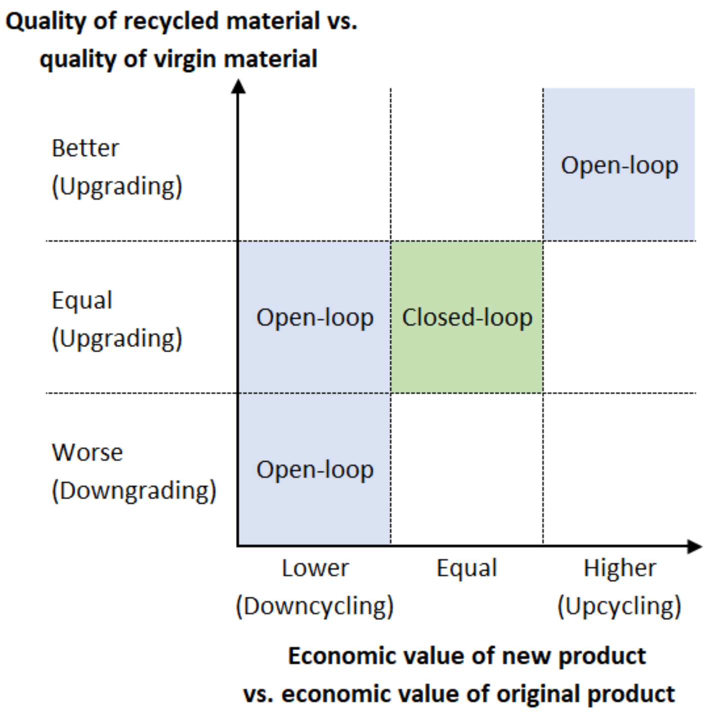

Figure 3.1.1. Classification of recycling processes

Although downgrading and upgrading are often used as synonyms of downcycling and upcycling, Figure 3.1.1 shows that is not necessarily true: a waste material may be upgraded to maintain its original function, and later used to manufacture a product of lower value than the original one. The confusion regarding the terminology has recently been intensified by Geyer et al.,[31] who question the usefulness of making a distinction between open and closed-loop recycling.

State-of-the-art technologies and processes for IWMSs

Regarding the technical and economic factors that hinder the complete separation and recycling of materials,[32-34] the concentration of the valuable materials in the discarded products and wastes is one of the critical parameters that will determine the feasibility of the recovery process;[35] several authors agree that the *unrecyclability* of some materials stems from the combination of small quantities of multiple materials in one product, like a smartphone.[36, 37] Hence the need to design systems that contemplate the valorization of all the materials within a given product. Clearly, the solution to this challenge relies on the development of more efficient sorting and disassembly technologies, along with the implementation of policies that promote the separate collection of these wastes.

One strategy that has been proposed to tackle the limitations of the current recycling technologies is to store in landfills the waste that cannot be properly separated or recycled until the pertinent technologies have been developed up to the point that they enable the recovery of the remaining secondary raw materials in waste,[38] which is the prime purpose of landfill mining, along with energy recovery from the stored waste.[39] Although several environmental and economic assessments of landfill mining have been performed so far,[40-42] more applied research is needed before the most sustainable pathway to landfill mining is agreed upon.[43]

Even though recycling efficiencies reached their full potential in the future, MSW is a complex heterogeneous mix of materials, and that prevents it from being treated by a single technology.[44] It is important to make a distinction between waste treatment; i.e, the set of processes seeking to minimize the environmental impacts of waste in order to comply with the pertinent regulations, and waste valorization, which concerns the transformation of waste into a product capable of providing society with a valuable service. However, a given waste management system can provide both functions, i.e., waste treatment and waste valorization.

An MSW management system focused on valorization must include a subsystem for materials sorting. The paper, cardboard, plastics, glass, aluminum and iron present in MSW are usually sorted in material recovery facilities and sent to recycling industries, where they are upgraded to be reintroduced into the market. For further information about the quality of recyclables and their recovery efficiencies in commingled and single-stream waste, the reader should refer to Cimpan et al.[45] There are several options for the valorization of both the inorganic and organic remaining materials. The alternative treatments to recycling the inorganic fraction of waste such as leftover plastic or textiles are the waste-to-energy processes like incineration, gasification or pyrolysis; the most developed and widespread of which is incineration.[46] These thermochemical processes can also be applied to the organic fraction of waste. The biological processes of anaerobic digestion and composting enable the organic matter to be looped back into the system as fertilizer (digestate or compost);[47] thus, they are considered recycling processes. In fact, anaerobic digestion is a strategy to simultaneously recover nutrients from the solid digestate and energy from the biogas produced by the microorganisms.[48]

Furthermore, new processes to valorize the organic fraction of waste are being proposed. The fermentation of organic waste has been suggested as a method to produce hydrogen.[49] Another example is the enzymatic liquefaction process proposed to separate the solid non-degradable materials that can be upgraded to Refuse Derived Fuel from a bioliquid that can be digested to produce biogas.[50] In addition to those, a number of processes to produce valuable chemicals such as levulinic acid[51] from organic waste or Refuse Derived Fuel have arisen; these are upcycling processes that fall within the category of waste refineries. Several authors propose to gasify waste in order to obtain syngas, a precursor to either the catalytic synthesis of methanol or the production of hydrocarbons via the Fischer Tropsch process.[52-55] Of the above-mentioned processes, the only one at large scale is operated by the company Enerkem, with a production capacity of 38,000 m^3 of methanol per year.[56]

Materials recycling or energy recovery?

In the specific case wherein the current state of the technologies allows a residual material to undergo either a recycling or an energy recovery process, materials recovery is usually encouraged; the Waste Framework Directive[57] states that, unless adequately justified by LCA, the EU Member States must follow the waste management hierarchy, according to which materials recycling takes precedence over energy recovery.

However, whereas the vast majority of studies agree that landfill is the least desired waste management alternative from an environmental point of view,[58-64] and there is also consensus on the claim that waste prevention and reuse are the cleanest and most efficient policies, the performed literature review reveals an ongoing debate on the final destination of the recyclable fractions of waste:[65-67] Should they be reintroduced into the production cycles, as new products or compost, or be sent to energy recovery facilities? The answer will greatly depend on the composition of the waste stream, which will determine its heating value and thus, its energy recovery potential. Furthermore, the assumptions made in the analysis, the system boundaries set and the local characteristics of the specific case study, will determine the optimal valorization strategy.

Cossu[68] analyzed the reasons behind the promotion of recycling. It causes the preservation of natural resources inasmuch as they are being extracted to a lesser degree. Moreover, a reduction in the amount of waste that needs to be properly managed or disposed of gives rise to cost savings in treatment processes. Nevertheless, the assumption that the economic costs and environmental impacts of material recycling are lower than those related to the extraction and processing of the virgin raw materials cannot be substantiated without a thorough analysis.

In the context of a globalized market, one of the factors that play a key role to the detriment of materials recycling is the long transport distances that they must go through to reach their end-users,[67] which has both environmental and economic drawbacks. Additionally, Massarutto et al.[69] proved that if a critical recycling rate (the ratio between the recycled materials and the waste generated) is exceeded, the economic benefits from recycling do not compensate its costs. Their study was based on the assumption that the quality of the collected materials worsens as the separation levels (the ratio between the source separated waste and the total amount of generated waste) increase, which was verified with data from waste management systems.

Several other authors have emphasized the importance of assessing the effect of increasing the recycling rates on the quality of the materials.[68,70-72] Some studies concluded that higher separation levels are not indicative of better materials quality.[72,73] On the contrary, systems focusing on quality rather than on quantity are likely to outperform the others.

An example of the damaging effects of recycling can be found in the steel manufacturing industry. The increased use of secondary materials in the steel making process causes an accumulation of elements such as copper, which hardens steel decreasing its quality and making it necessary to dilute the amount of recycled scrap.[71] The counter-effect of dilution is that it reduces the market demand for recyclables.[74] Hence, as Loughlin and Barlaz[75] pointed out, recycling policies must make sure that the supply of recycled materials matches the demand.

Particular attention must be paid to the potential hazards of recycling because of human exposure to pollutants and toxic compounds. Bisphenol A was found in an array of waste paper samples, possibly as a consequence of the recycling of secondary waste paper.[76] Recycling has also been recently pointed as a potential source of phthalates in plastics;[77] as a consequence, the application of recycled plastics in products sensitive to phthalate content, such as toys and food packaging, must be restricted.

The risk for human health is in fact the main argument that the detractors of energy recovery technologies hold, despite the fact that the thermochemical processes and anaerobic digestion are a means to simultaneously reduce the volume and mass of solid waste and produce heat and electricity. Incineration has been traditionally regarded by the public opinion as a threat to human health and the environment, because of the high concentrations of heavy metals, dioxins and furans present in the flue gases prior to the development of the current sophisticated Air Pollution Control Systems.[78] However, with the state-of-the art technologies, these pollutants do not pose a risk any longer, since they are well below the air emission limit values established by the European legislation, which are quite restrictive in comparison to those of other countries.[79]

Furthermore, several studies report that savings on the environmental impacts can be achieved displacing conventional energy sources by MSW.[80,81] Hence the importance of linking the analysis of the energy and waste management systems,[82] as Eriksson and Bisaillon[83] and Münster et al.[84] did.

The competition between materials recycling and energy recovery is of particular interest for those materials such as cardboard and plastic with high calorific values,[67] which make them attractive fuels for heat and electricity production, whereas deliberately subjecting the incombustible materials, i.e. metals and glass, to energy recovery processes seems pointless. However, a fraction of the metals that cannot be separated by mechanical and magnetic methods can be recovered after the incineration process, because of their enhanced concentration in the residual ash.[85]

Taking into account all the considerations described above, it is reasonable to conclude that materials recycling and energy recovery should complement each other to meet the local demands; even in the utopian scenario wherein it is technologically and economically feasible to completely close the material loops, there might still be a demand for virgin materials, not only because of their higher quality, but also because of social objections.

FRAMEWORK FOR THE ANALYSIS OF CIWMSs

The precise definition of a CIWMS is instrumental to the development of a framework that relies on that concept. The previously discussed barriers to the development of CIWMs should provide a basis for the delimitation of their system boundaries and the definition of their functions. These notions, which are based on the principles of the cradle-to-cradle design,[86] are explored to a greater extent in this section.

Previous application of the circular economy approach to the design of IWMSs

Although specific guidelines for the design and assessment of CIWMSs from a systems perspective have not been found in the literature, Arena and Di Gregorio[70] proposed a series of principles, consistent with the targets of the circular economy, that IWMSs should follow: "An integrated and sustainable waste management system should be defined and developed according to the following criteria: i) to minimize use of landfills and ensure that no landfilled waste is biologically active or contains mobile hazardous substances (…); ii) to minimize operations that entail excessive consumption of raw materials and energy without yielding an overall environmental advantage; iii) to maximize recovery of materials, albeit in respect of the previous point; and iv) to maximize energy recovery for materials that cannot be efficiently recycled, in order to save both landfill volumes and fossil-fuel resources".

Proposed definition of CIWMSs

A description of the concepts of IWMSs and CIWMSs is provided in this section. An IWMS denotes a system whose main input is waste and comprises a number of processes to sort this waste and give each waste fraction the most appropriate treatment according to its chemical composition and the desired function of the system outputs. However, this definition corresponds to that of a linear IWMS, like the one shown in Figure 3.1.2. If an IWMS is to be studied from the perspective of a circular economy and waste prevention, this definition is incomplete. A CIWMS is a type of IWMS that seeks to enhance the circularity of resources by strengthening the link between waste treatment and resource recovery. Thus, CIWMSs can be considered an instrument that enables fulfilling the goals of a circular economy. The definition of CIWMSs could also apply to a system that focuses on just one waste fraction, such as organic waste.

The purpose of a sustainable CIWMS is to achieve the maximum economic profit and benefits for society at the expense of the minimum environmental impacts and consumption of natural resources. Under this perspective, materials upcycling is favored over downcycling. To accomplish these sustainability goals, the maximum amount of waste is expected to be valorized to expand its lifetime, so that it can serve a function to society. This entails that the amount of waste sent to landfill is minimized, although landfills cannot be totally replaced[87] because all the other subsystems generate certain amount of waste that the current technologies cannot valorize.

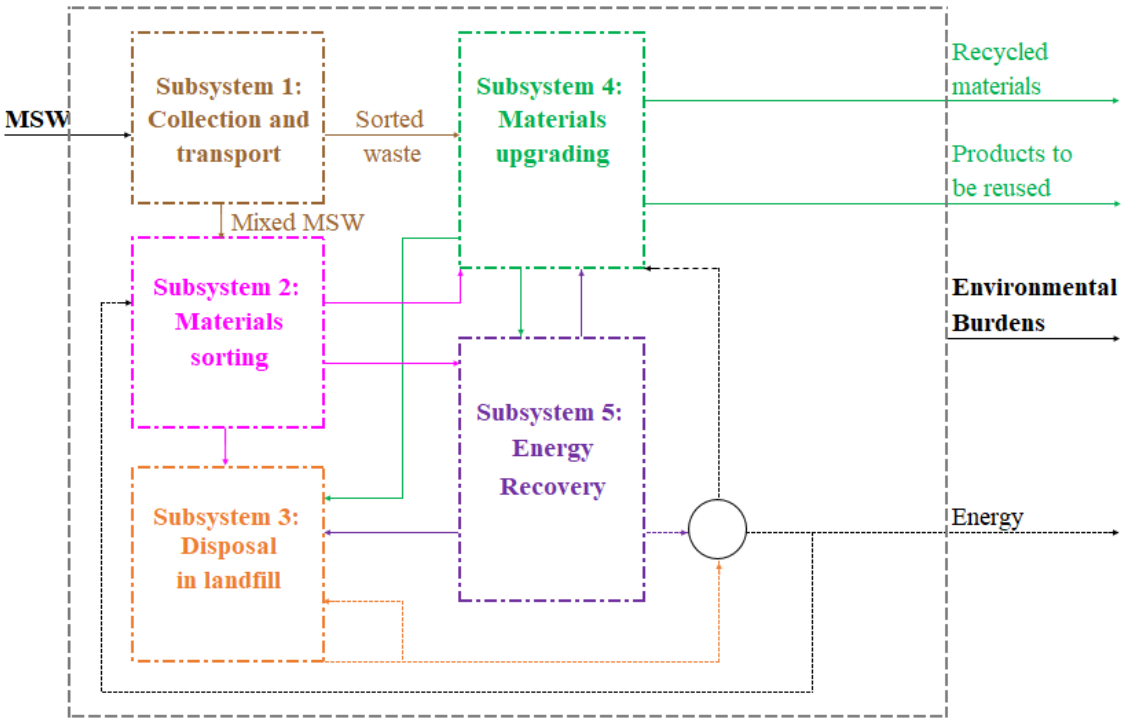

Figure 3.1.2. Configuration and boundaries of a linear IWMS

A CIWMS can be as complicated as the designers wish, but a CIWMS that manages mixed MSW would ideally deliver materials, energy and nutrients. It could also supply some chemicals, a relatively novel approach to waste management. The waste refinery concept, analogous to that of an oil refinery but taking waste as a feedstock, has gained popularity in recent years.[88] A waste refinery is a type of IWMSs wherein chemical reactions take place to upcycle mixed waste or a fraction of waste into marketable chemicals.

Configuration and boundaries of a CIWMS

A CIWMS should encompass the subsystems that connect the transformation of raw materials into waste with the waste treatment subsystems, so that the consequences of the recirculation of the materials into the upstream subsystems can be fully accounted for. A CIWMS that relies to a lesser extent on the consumption of virgin raw materials would result from the connection of the upstream subsystems with those of a traditional linear IWMS, as shown in Figure 3.1.3. As many transport subsystems as necessary should be added to the system depicted in Figure 3.1.3 for each particular case under study. From an LCA perspective, the subsystems 0-2, which comprise the upstream and midstream processes, constitute the background system of the model, whereas the remaining downstream subsystems, which concern those processes under the control of the decision-maker,[89] belong to the foreground system.

These system boundaries intend to capture the whole life cycle of the materials that compose waste, including the stages concerning the consumption of the services derived from the transformation of the natural resources extracted from the ecosystems. Once consumed, some products such as food or cosmetics leave the system as air emissions or wastewater. On the other hand, many products like textiles and furniture provide a service for a time period without being consumed. It is worth mentioning that the primary raw materials delivered by subsystem 0 cannot be compared to the secondary materials produced in subsystem 6 on a mass basis; the comparison must be based on the functions provided by those materials. For instance, 1 kg of primary aluminum might not be functionally equivalent to 1 kg of recycled aluminum, because of their different chemical composition and physical properties.

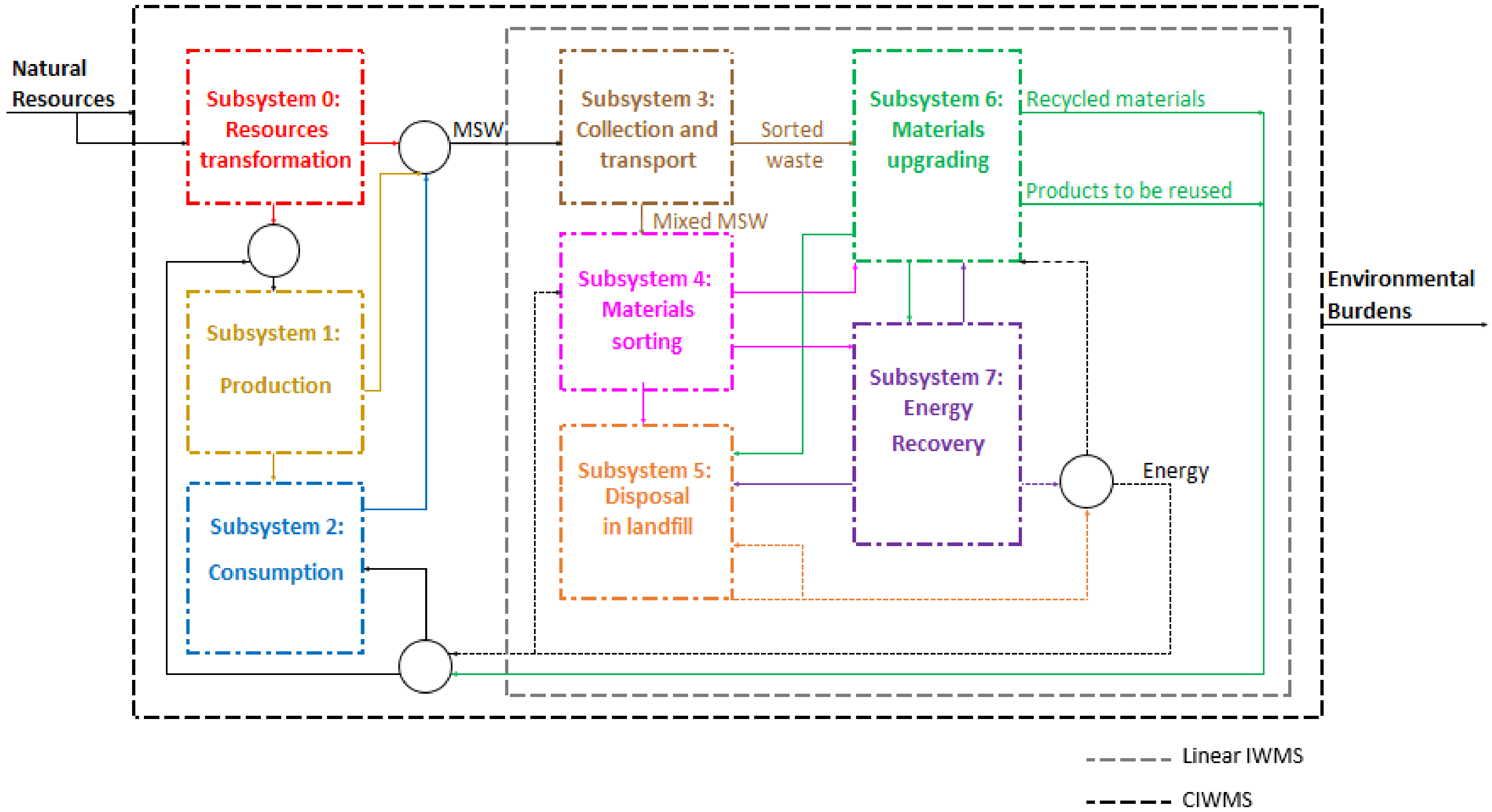

Figure 3.1.3. Configuration and boundaries of a CIWMSs

Figure 3.1.4 illustrates the exchanges between a CIWMS and the surrounding ecosystems, and how a CIWMS can transform one type of environmental burden (waste) into a resource that might displace the consumption of virgin resources that would provide the same function.

The scope of a CIWMS that manages mixed MSW is so broad that the only systems within the technosphere that it might be related to are the wastewater and the industrial waste treatment systems. Those systems are outside the scope of the study of the CIWMS shown in Figure 3.1.3 and thus, the consequences of the decisions affecting those systems will not be considered

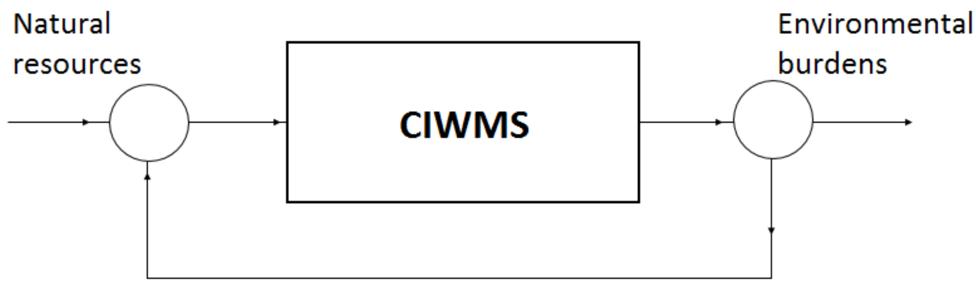

Figure 3.1.4. Overview of the exchanges between a CIWMS and the ecosystems

Link between industrial symbiosis and CIWMSs

According to Chertow,[90] industrial symbiosis engages traditionally separate industries in a collective approach to competitive advantage involving the physical exchange of materials, energy, water, and/or by-products. The keys to industrial symbiosis are collaboration and the synergistic possibilities offered by geographic proximity. Thus, the proposed CIWMS is analogous to an industrial symbiotic system, in the sense that a resource exchange network can be stablished. Nonetheless, although industrial symbiotic systems could play a major role in the circular economy, the concept of a CIWMS is much broader; it is not restricted to nearby industrial systems, but it also includes waste managers, consumers and the supply chains.

Hence, the generic methodological approaches proposed in the literature to assess the performance of industrial symbiotic systems[91, 92] should not be, a priori, extended to CIWMSs.

Recommended tools for the analysis of CIWMSs

Because of the wide range of existing technologies to manage waste, process engineers must carefully study the available possibilities at the design phase of a CIWMS. The superstructure that might emerge after considering process integration could be quite complex. Thus, the selection of the optimum configuration of the system is not a trivial matter, and it might require mathematical programming techniques. Moreover, since the chemical composition of waste will determine the type of processes that it can be subjected to, it can be concluded that the design of a CIWMS should be based on mathematical programming and Material Flow Analysis (MFA), so that the circularity of materials is warranted. The combination of these tools with scenario analysis techniques that assess the consequences of changes in waste composition and quantities or possible technological improvements, could be a valid strategy to account for the dynamic variables that might fluctuate during the studied time horizon.

On the other hand, the assessment of the performance of a CIWMS must analyze all its sustainability dimensions. The sustainability criteria regarding the economic and social dimensions of CIWMSs are at least as important as the environmental aspects and must be likewise assessed; nonetheless, they will not be deeply discussed in this critical review.

METHODOLOGIES APPLIED IN THE LITERATURE

Regarding the methodological approaches reported to be applied in the literature, Chang et al.[21] and Juul et al.[82] classified the system analysis tools that have the potential to assist in the design of IWMSs and the decision-making processes as:

- System engineering models, which focus on supporting the design of the system. These are simulation models, optimization models, forecasting models, cost-benefit analysis or multi-criteria decision-making (MCDM).
- System assessment tools. They focus on assessing how an existing system performs. LCA, MFA and risk assessment are examples of such tools.

Coupling these two types of methodologies is recommended not only because it will lead to a better understanding of the IWMS,[93] but also because the sustainability analysis of an IWMS requires an integrated approach; the applied methodologies should complement each other so that all the sustainability dimensions can be properly evaluated and the economic, environmental and social objectives are balanced.

Another strategy that has been suggested to support the decision-making process is taking a participatory approach. This can be done by either asking multiple stakeholders to participate in the analysis,[65] or by applying a game-theoretic approach that seeks the fair distribution of benefits and costs.[94]

The methodological approaches applied in the 77 reviewed papers are shown in Figure 3.1.5. Whereas over one third of the reviewed papers focus solely on the environmental impacts associated with the IWMS (all of them by means of LCA), only one study relies solely on an economic assessment, based on Life Cycle Costing (LCC).[69] More information on the application of LCC to waste management systems can be found in Martinez-Sanchez et al.'s paper.[95]

Over one fifth of the reviewed studies assessed more than one sustainability dimension. A few papers,[84,96-98] combine the LCA methodology and optimization techniques to broaden the scope of the study and include other sustainability criteria. Mirdar-Haridani et al.[99] combined optimization and social LCA. Multi-objective optimization, applied in some of the reviewed papers,[96,100-105] is possibly the most adequate technique to take into account all the sustainability criteria. Oppositely, other authors[63,106] combined LCA with a set of indicators to account for the other sustainability dimensions of an IWMS.

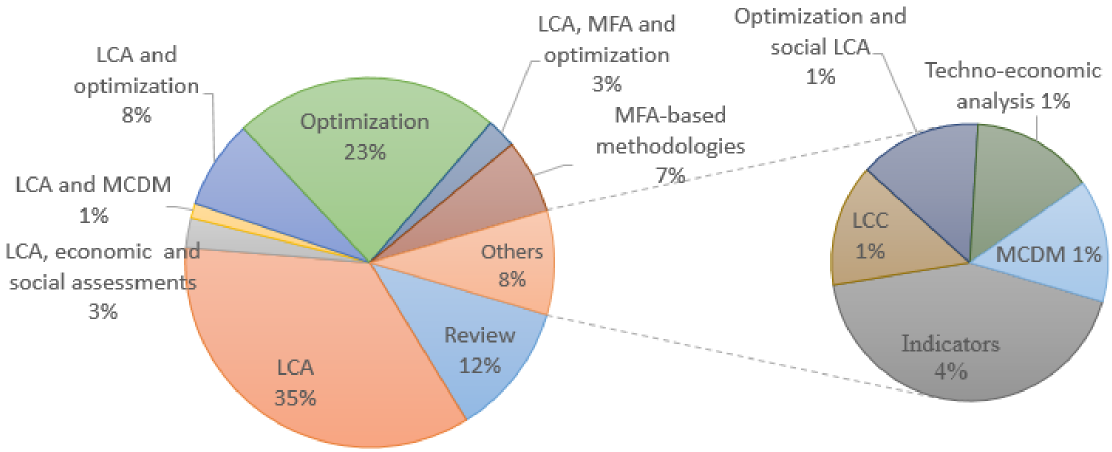

Figure 3.1.5. Methodological approaches applied in the reviewed studies

On the other hand, MFA and/or Substance Flow Analysis (SFA) enable us to explicitly consider the waste characteristics and thus provide a more detailed description of the system under study and track each waste fraction throughout the system. Additionally, Energy Flow Analysis (EFA), which was applied in two studies,[107,108] might prove useful to determine the most suitable valorization treatment to each waste fraction.

So far, the theoretical framework required to combine LCA, multi-objective optimization and MFA techniques has only been described by Vadenbo et al.[104,105] although the methodology was not applied to an IWMS.

HOT TOPICS

The most discussed methodological aspects in the reviewed studies and the challenges and possibilities of their application to the design and assessment of CIWMSs are presented in this section aiming at providing some helpful and critical insights into the development of a theoretical framework for the analysis of CIWMSs.

Accounting for waste prevention

Wastage of goods and products is a tremendous global challenge; taking the food supply and consumption chains as an example, around one third of the food produced for human consumption worldwide is currently lost or wasted.[109]

Waste prevention stands at the top of the waste management hierarchy, as a strategy to be implemented in the life cycle stage prior to waste generation that seeks to minimize the depletion of natural resources and its subsequent environmental burdens. The term *waste prevention* refers to any measures taken before a substance, material or product become waste, that reduce: a) the quantity of waste, b) the adverse impacts of the generated waste and c) the content of harmful substances in materials and products.[57]

Nevertheless, the analysis of waste prevention activities in the framework of LCA has not been normalized yet; only a few studies outline the methodological steps to follow,[110-112] concurring that this is an active area of research.

LCA models of waste management typically calculate the environmental burdens on a waste mass basis. This is the most straightforward option to choose the functional unit. However, it makes this approach inadequate for the comparison of scenarios including waste prevention strategies, given that the amount of waste produced varies among them.[113] Moreover, these models usually rely on the “zero burden approach”, which does not include the upstream processes within the system boundaries because it is assumed that their primary function is not to produce waste and thus none of the environmental burdens generated in the upstream processes are associated with it. Nonetheless, if different amounts of waste are produced in each scenario, the zero burden approach cannot be considered because the contribution of the upstream processes to the overall environmental impacts of the system will differ.[114] Consequently, a proper methodological approach to deal with waste prevention activities from a life cycle perspective should define:

- A functional unit that accounts for the amount of waste prevented.
- System boundaries that include the upstream processes involved in waste generation.

Another issue that must be considered when waste prevention activities are being accounted for is the allocation procedure of the environmental impacts among the products or services delivered by the IWMS. Applying the direct substitution approach in order to avoid allocation among several products is not recommended, given that negative results might be obtained, leading to the erroneous conclusion that a greater amount of waste leads to less environmental impacts.[115]

Cleary[110] recommends an attributional approach with system expansion to account for the upstream processes associated with waste production, arguing that a consequential approach does not consider waste prevention as a waste management strategy functionally equivalent to the others in the waste management hierarchy, since no environmental burdens are attributed to waste prevention activities; that is to say, it simply quantifies the consequences of reducing the waste inputs in the system. Only Gentil et al.[111] claim to apply a consequential LCA model. These authors expand the system boundaries to the upstream processes related to the waste generation processes, although they acknowledge that the cascading effects of waste prevention should have been further assessed.

All of the above-mentioned studies define the functional unit as the sum of the waste managed through conventional methods and the amount of waste prevented, although nuances in the applied approach can be found among the studies.

Quantifying biogenic carbon

Whether biogenic CO_2 emissions are considered neutral or an environmental burden to an IWMS will have a significant influence on the results and conclusions drawn from the analysis. Since studies relying on different assumptions are hard to compare, it is imperative to standardize this matter, not only within the waste management sector.

The EPA[116] defines biogenic CO_2 emissions as CO_2 emissions related to the natural carbon cycle, as well as those resulting from the combustion, harvest, digestion, fermentation, decomposition, or processing of biologically based materials. It is worth remarking that the origin of fossil fuels, produced millions of years ago, is also biological.[117]

The first difficulty that arises when calculating the carbon footprint of a given IWMS is the differentiation between biogenic and fossil carbon. A rigorous MFA should be performed in order to trace back the carbon source and identify the carbon sinks. Carbon (biogenic or not) may be released as an environmental burden or remain in the anthroposphere, in any of the following forms:

- Emissions to the atmosphere. In the presence of oxygen, carbon is oxidized to CO_2. Under anaerobic conditions carbon is reduced to CH_4.
- Wastewater pollution and landfill leachate wherein carbon is present in a variety of organic compounds.
- Sequestered carbon in landfills or in soil amendment products (compost and digestate).

It must be highlighted that the distinction between an environmental burden and the accumulation of a substance in the IWMS under study is often unclear; the system boundaries need to be precisely established at the definition of the scope of the work.

Within an efficiently designed IWMS water is not considered a final carbon sink. After the adequate treatment, the carbon present in the leachate leaves the liquid phase as CO_2 or CH_4,[118] whereas the carbon in wastewater is distributed between the gaseous emissions and the sludge,[119] being the latter subsequently treated as solid waste. Even though Griffith et al.[120] estimate that up to 25% of the carbon content in wastewater is of fossil origin, it is widely assumed that the totality of carbon is biogenic, and thus it is typically not accounted for.[119]

Although emissions from leachate treatments are estimated in some of the reviewed papers,[62, 96] none of them made express reference to the carbon source. The reviewed articles that accounted for biogenic CO_2 are shown in Table 3.1.2. The procedure followed to determine the carbon origin is not clearly stated in many cases. Whereas Tabata et al.[98] and Vergara et al.[121] consider that biogenic CO_2 is derived from the biogenic fraction of waste, only Manfredi et al.[62] and Turner et al.[122] explicitly consider the fraction of biogenic carbon in the input waste.

Regarding the stored carbon in landfills and the carbon emissions to the atmosphere, for the specific case in which an LCA is performed with the objective of comparing different scenarios but there is no interest in knowing the values of their individual carbon footprints, Christensen et al.[123] proved that, provided that the assumptions concerning biogenic CO_2 emissions and carbon sequestration are consistent (considering biogenic CO_2 emissions either neutral or not neutral) and the system boundaries are clearly established, the emission ranking of scenarios remains the same.

As can be seen in Table 3.1.2, biogenic CO_2 emissions are assigned a GWP factor (expressed as kg of equivalent CO_2 per kg of emitted CO_2) of zero in most studies, which implies that no environmental impacts in terms of climate change potential are attributed to them. Applying this GWP is analogous to expanding the system boundaries to include the upstream processes of photosynthesis. Thus, unless biogenic CO_2 is being stored, the CO_2 that is captured during the growth of biomass and comes into the system, is balanced with the biogenic CO_2 that leaves the system, achieving carbon neutrality. For the sake of coherence, a negative GWP must be assigned to the carbon that is captured in the photosynthetic processes and remains sequestered in the system. Nonetheless, as Vergara et al.[121] point out, by applying this procedure only the environmental benefits of the upstream processes are being taken into account, disregarding their environmental burdens. As a consequence, this approach might lead to higher environmental credits than burdens, entailing that landfills and soil amendment products contribute to climate change mitigation.[122]

To correct this incoherence, the carbon flows that connect the system to the environment (primarily as CO_2 and CH_4) must be inventoried. If the system boundaries are expanded to include the upstream processes, once the elemental composition of the waste and products is known, the incoming carbon flows can be easily calculated: every mole of biogenic carbon present in the products, waste and emissions originates from a mole of CO_2 that was absorbed by biomass in the photosynthetic process. Afterward, the carbon flows that come into the system must be subtracted from the carbon flows that leave the studied system.

This systematic approach allows applying the same GWP (1 kg CO_2-eq·kg CO_2^{-1}) to CO_2 emissions from scenarios with different system boundaries, regardless of the CO_2 origin.

The proposed procedure, which relies on the waste composition provided by the MFA, ensures that the CO_2 removed from the atmosphere, whose carbon eventually leaves the system as CH_4, is accounted for. The studies compiled in Table 3.1.2 make no express reference to a correction in the GWP of biogenic CH_4, when CH_4 constitutes a significant fraction of the outlet stream of some technologies that process biogenic waste, such as anaerobic digestion.

Table 3.1.2. GWP and other methodological considerations regarding biogenic carbon in the reviewed papers

	Biogenic CO_2	**Stored biogenic carbon**		**Specified carbon source?**	**Zero burden approach?**
	Value (kg CO_2-eq·kg CO_2^{-1})	Value	Unit		
Aghajani et al.[124]	0	-	-	No	Yes
Blengini et al.[65]	1	-1	Unspecified	No	Yes
Chang et al.[96]	0	-	-	No	Yes
Manfredi et al.[62]	0	-44/12	kg CO_2-eq·kg C^{-1}	Yes	Yes
Minoglou et al.[125]	0	-	-	No	Yes
Tabata[98]	0	-	-	Yes	Yes
Turner et al.[122]	0	0 or -44/12	kg CO_2-eq·kg C^{-1}	Yes	Yes
Vergara et al.[121]	0	-1	Unspecified	Yes	No
	1	0	Unspecified	Yes	Yes

Accounting for uncertainty

Models aiming at describing complex systems carry a level of uncertainty whose effect on the outcome might be hard to predict without the right methodology. There are plenty of sources of uncertainty within an IWMS, such as waste composition, the efficiency of the treatment processes, the substitution ratio of virgin materials or the effect that the seasonal changes in weather may have on the waste degradation rate. For a detailed compilation of uncertainty sources, the reader should refer to Clavreul et al.[126] However, the paramount variable with which uncertainty is associated, regardless of the complexity of the model, is waste composition.

As Laurent et al.[127] pinpointed in their review, LCA studies do not usually account for waste composition very accurately. This asseveration could be further extended to waste management models in general, even though waste composition will determine the results of the subsequent analysis, simulation or optimization, given that the available treatment options and the type and amount of emissions resulting from the different waste treatment alternatives strongly depend on the elemental composition of waste. This is the reason coupling MFA with other analysis tools is the precursor to identifying the optimal configuration of an IWMS. Nevertheless, adequately characterizing the waste composition is a difficult task because of the heterogeneity of the material flows, and it might require complex statistical analysis. Thus, representative data of the average waste composition inevitably brings uncertainty into the model.

The elements that are excluded from the analysis without a clear justification also represent a source of uncertainty. For instance, the environmental impacts related to capital goods might have a significant influence on the results of an LCA,[128] but they are often not modeled.[127,129,130]

Stochastic modeling, which relies on the propagation of probability distributions, is the most frequently deployed methodology to consider the effect of uncertainties on the LCA results, although scenario analysis is more commonly applied for the LCA of waste management.[126] Sensitivity analysis to investigate the effects of a change on an assumption or the value of a parameter are routinely performed in many of the reviewed studies.[50,53,60-62,65,80,81,115,121,122,129,131-139] Massarutto et al.[69] also carried out a sensitivity analysis in their LCC analysis. Notwithstanding only three of the above-mentioned studies[50,53,131] analyzed the impact that different waste compositions would have on the results.

Hanandeh and El-Zein[140] considered the uncertainty related to the input waste composition, among other parameters. Comparing the results of the stochastic model of an IMWS with those of a deterministic model, they found that when uncertainty is taken into account, the environmental burdens of one of the studied impact categories became environmental credits, proving that the uncertainty of the data in their case study was definitely not negligible. However, Clavreul et al.[126] claim that probability distributions, which are oftentimes dependent on incomplete information, should be applied cautiously. Instead, they proposed a systematic sequential approach to quantify uncertainty in LCA models of waste management systems that comprises a number of complementary methodologies for uncertainty analysis.

Regarding the quantification of uncertainty in the models aiming at optimizing IWMSs, two methodologies can be differentiated in the reviewed literature:

- After the initial optimization of the objective functions a sensitivity analysis is performed to check the effect of a change in the input parameters or the assumptions made on the optimal solution. Tabata et al.,[98,141] and ThiKimOanh et al.[142] apply this methodology.
- A methodology to quantify uncertainty is embedded in the model or the optimization technique. Table 3.1.3 compiles the modeling and optimization methodologies applied for that purpose in the reviewed studies.

As can be seen in Table 3.1.3, some studies apply a combination of techniques. Interval programming, in which uncertainties are expressed as interval values, is the most common programming technique to quantify uncertainty. Stochastic and fuzzy programming are also popular; the difference between them is that in stochastic programming uncertainty is modeled through discrete or continuous probability functions, whereas fuzzy programming considers random parameters as fuzzy numbers and constraints are treated as fuzzy sets.[143]

Finally, an approach to quantify uncertainty within MCDM models was proposed by Pires et al.[144] They developed a MCDM framework that integrates an interval-valued fuzzy method with the analytic hierarchy process (AHP) and the technique for order performance by similarity to ideal solution (TOPSIS) in order to help decision-makers prioritize waste management scenarios.

The extensive amount of methodologies developed to account for uncertainty makes it hard for the non-experts to choose the most appropriate one for the analysis of their IWMS. Two trends have been observed in the literature: the performance of sensitivity analysis and the combination of several methodologies. The former risks not capturing the complexity of the model, while the latter may become a time-consuming process that considerably increases the researchers' effort.

In any case, a meaningful uncertainty analysis must be based on the correct identification of the parameters and assumptions that will bring uncertainty into the model, which are not always clearly listed in the reviewed studies.

Table 3.1.3. Methodologies to quantify the effects of uncertainty in the reviewed optimization models

	Fuzzy programming	Stochastic programming	Interval programming	Factorial design	Minimax regret analysis
Cui et al. [145]			x		x
Chang et al.[100]	x				
Dai et al.[146]			x		
Li and Chen[147]	x	x	x		
Srivastava et al.[148]	x				
Wang et al. [149]	x	x	x		
Zhai et al.[150]			x	x	
Zhou et al.[151]		x			
Zhu and Huang[152]		x			

Dynamic modeling

Most of the reviewed models, with the exception of multi-period optimization models,[97,103,141,145-148,152-154] describe static IWMSs that do not account for changes in the system variables throughout time. Oppositely, multi-period optimization models assume that the constraints and the parameters remain constant within a given time period, although they may differ between different stages. Hence, in spite of being time dependent, the outputs of these models are not a function of time, but a function of the time period. In fact, models introducing time series have been classified as *quasi-dynamic*,[155] under the argument that the results of one period do not determine the results of the next period. The implementation of dynamic models whose outputs are a function of time would bring a higher degree of complexity into the analysis; for instance, modeling the behavior of markets throughout time would add realism to an LCA, but because of the large data requirements, it is not usually considered a feasible option.[155]

Thus, the definition of time stages appears to be the most straightforward and practical route to account for the time-dependent changes in the system, such as the need to manage obsolete goods after they have provided the expected service. The shorter the established time periods, the more reliable the model will be. The time periods should be established so that the seasonal variations in waste composition are accounted for. Of the reviewed studies, only Levis et al.[156] took into account the changes in waste composition in the studied time period. If the study aims at quantifying the environmental impacts and the consumption of natural resources of the system, successive LCAs should be performed for each time period in which the input waste composition varies. Accordingly, different functional units referring to each specific time period should be defined.

The seasonal changes in waste composition (proven for example by Castrillón et al.[157] pose a challenge to the design of CIWMSs, given that they must be flexible enough to adjust to the changes in the feed composition. Furthermore, since manufacturers cannot count on a steady supply of secondary materials, the fluctuations in waste composition hamper the shift to a circular economy.

It is important not to confuse the duration of the supply of goods and services provided by the system, which is identified by the functional unit, with the time horizon of the LCA,[114] which is the time length during which the flows that connect the IWMS with the environment are accounted for. Additionally, the selected time horizon determines the value of the characterization factors used to calculate the contribution of the different substances exiting the system to each of the impact categories studied on the LCA.[158] Thus, the time horizon must be long enough to include all the relevant emissions to the environment. This guideline is of particular interest for modeling landfills, since their emissions may prevail for a long time in the order of thousands of years.[159]

For the defined time period in which a CIWMS is analyzed, certain waste fractions might travel within the system for a number of times; depending on the time at which the system is being described, some materials may be part of the waste or the products. In fact, the products into which a material is transformed might even be different if they undergo an open-loop recycling process. A methodology to calculate the average number of times a material is used was proposed by Yamada et al.[160]

The disparities in the material flows within a given time period can only be solved by assuming that the model concerning each time period is a steady-state model; i.e, that the incoming natural resources and the flows of waste and products within the system are constant and homogeneously distributed along the studied time period. Following this methodology, materials should be counted as both waste and products as many times as cycles they describe within the system in the defined time period.

APPLICATION OF THE CRADLE-TO-CRADLE APPROACH

The boundaries of a CIWMS do not allow the implementation of the traditional linear cradle-to-grave LCA; thus, a cradle-to-cradle approach must be applied. In this section the adjustments to the LCA scope that this new perspective requires will be discussed, focusing on the modeling framework, the multi-functionality problem and the definition of the functional unit, all of which are intrinsically related to one another and will be determined by the goal and scope definition.

Goal and scope definition

The goal of the LCA of a given CIWMS might differ among studies, which makes it hard, if not impossible, to compare their results. The proposed methodology discussed in this section will be coherent with this goal: to identify possible improvements in the design of a CIWMS wherein waste prevention activities are implemented, so that its environmental impacts and its consumption of natural resources can be minimized. Hence, the analysis is intended to assist the decision-makers in the design of a CIWMS.

Multi-functionality problem

The LCA practitioner might come across a multi-functionality problem: how to allocate the environmental impacts between all the functions that the system supplies if the further subdivision of the subsystems that configure the CIWMS cannot be applied to avoid allocation, because of the interconnection between them. To deal with this multi-functionality problem, two strategies, which depend on the selected modeling approach, can be applied:[23, 161] system expansion or allocation. According to ISO 14044,[23] system expansion should be deployed wherever possible in order to avoid partitioning the environmental burdens.

Most studies analyzing IWMSs apply the direct substitution (also called avoided burden) method; [50,53,58,61-63,65,80,81,106,111,115,121,122,129-132,134,139,162-174] they consider that the primary aim of their system is to treat waste, and they expand the system boundaries to include within the system the other products and services supplied, like materials and energy, and subtract their environmental impacts from those of the original system. However, a CIWMS does not operate under the assumption that waste needs to be treated in order to minimize its negative impacts, but valorized, so that the consumption of natural resources is reduced.

Functions of a CIWMS

According to the system boundaries set in Figure 3.1.3, the functions fulfilled by a CIWMS are twofold:

- To supply the services that society demands, regardless of the origin of the raw materials.
- To exploit the maximum amount of the generated waste, by either producing new products from it or recovering its energy, with the ultimate goal of minimizing the consumption of natural resources.

The second function is a consequence of the first one, and the first one can be partially achieved due to the accomplishment of the second function. However, if waste upgrading and energy recovery processes were not implemented, the supply of the services demanded by society could still meet the demand, relying solely on the extraction of natural resources. Thus, it can be agreed that the primary function of a CIWMS is waste exploitation.

According to the definition of the system functions, it is not necessary to disaggregate any of them by the type of services and products provided in order to solve the multi-functionality problem. This way, the uncertainty brought into the model by the choice of the allocation procedure is reduced. Moreover, the problem of allocation in open-loop recycling, which is a recurrent discussion in the LCA literature,[26,159,160,175,176] is avoided.

System expansion approach

If the LCA practitioners are interested in analyzing the overall environmental impacts of the whole system, the system expansion approach must be followed. The studied CIWMS should be compared to a functionally equivalent system whose functions are provided by alternative subsystems;[159] for instance, a linear IWMS that depends exclusively on virgin raw materials.The environmental benefits of the complete CIWMS could be estimated as the difference in the environmental impacts of the linear and circular IWMSs.

If on the contrary, the study aims at investigating the environmental impacts derived from the primary function of the CIWMS, the direct substitution or avoided burden approach could be applied by expanding the system boundaries to include alternative subsystems responsible for the secondary function, based entirely on virgin raw materials. Their environmental impacts should be subsequently calculated and subtracted from the environmental impacts of the studied CIWMS. Accordingly, the resulting environmental impacts are assumed to be due to the primary function of the system. This might result in overall negative environmental impacts and, as a consequence, the system could be mistaken for a sink of environmental burdens.

If system expansion is applied, a choice between marginal and average data must be made to model the system functions. Marginal data is used to model systems whose outputs change in response to decisions regarding the life cycle of the system under study, for example a decrease in the demand for the electricity produced from natural gas as a consequence of the supply of electricity from waste-to-energy processes. Average data, on the other hand, represents the mean data in a region; the average electricity mix refers to the grid mix of that region, and it does not reflect any changes in fuel consumption because of the changes in the electricity demand. Although average data might lack accuracy, it is more appropriate if the effects that the decisions taken have on the surrounding systems are not certain. The selection of the data is closely related to the LCA modeling framework applied. Whereas "attributional LCA focuses on describing environmentally relevant physical flows to and from a life cycle, consequential LCA aims at describing how the environmentally relevant physical flows to and from the life cycle will change in response to possible decisions".[161]

Allocation approach

Heijungs and Guinée[177] are firm advocates of allocation procedures because the assumptions on which the direct substitution approach is based are likely to introduce considerable uncertainty into the model. Whereas they recognize that the allocation approach is subject to essentially arbitrary allocation factors, they argue that it is extremely hard to predict what system would be affected if the secondary function of the studied system was meant to replace one of the functions of another system, and up to what extent the environmental impacts caused by the other system would be avoided. Although the selection of a 100% substitution ratio is common, several authors suggest that a complete displacement is unlikely.[31,178-180]

In addition to that, if the substituted function was produced in a multi-functional system, the system boundaries would have to be further expanded until mono-functional systems were found, significantly increasing the complexity and the uncertainty of the system. Ekvall and Finnveden[176] also acknowledged the importance of the uncertainty caused by system expansion;

they stated that system expansion is an adequate procedure to solve the multi-functionality problem as long as data for the competing production of the secondary function is available, and the data uncertainties are not too large, which agrees with the guidelines of ISO 14044.[23]

This argument can be easily extrapolated to the case of a CIWMS aiming at valorizing MSW. The resources transformation subsystem, responsible for the secondary function of a CIWMS, comprises many production subsystems; modeling the alternative processes relying on virgin raw materials would bring multiple sources of uncertainty into the model, not to mention that it would be an extremely time-consuming task.

If an allocation procedure is selected to solve the multi-functionality problem, it must be borne in mind that except when physical causal relationships are deployed as a basis for allocation, the property according to which the allocation is performed depends entirely on the choice of the LCA practitioner.

The chemical composition of the flows within a CIWMS, determined by the MFA, is a valid causal criterion to allocate the input-specific environmental impacts. However, given that the composition of the recycled materials should be, a priori, identical to the composition of the virgin materials, this criterion could only be applied in the cases wherein either the recycled materials carry pollutants accumulated in the recycling process, or certain materials cannot be recycled and thus the environmental impacts derived from the processing of those materials are due to the incoming virgin materials into the system. Furthermore, the environmental impacts caused by the process specific emissions, such as dioxins and furans produced in the incineration processes,[181] which are dependent on the operating conditions and the applied technologies, cannot be allocated according to the chemical composition of the input flows.

Hence, a different allocation factor that enables partitioning all the environmental impacts between the system functions must be defined. There are basically two types of approaches to perform the allocation of environmental impacts in the cases wherein causal relationships cannot be found, those relying on a physical parameter, such as mass or volume, and those that are based on socioeconomic criteria. Even though both approaches are internally consistent as long as the selected physical property or socioeconomic indicator is also applied to quantify the performance of the system and used to calculate the functional unit, different results will be obtained for different allocation factors, and they might show opposite trends. Therefore, the choice of the allocation factor should never be made based on an arbitrary decision, it should respond to the goal and scope of the LCA instead.[182]

One of the reasons for not including socioeconomic parameters in the LCA is that if more than one of the sustainability dimensions (economy, environment and society) are studied jointly, some of the trends in the results might be overlooked. For instance, the objective of the study of the carbon footprint of a CIWMS wherein the functional unit is defined as the revenues generated in a given time period, could be to detect what changes in the configuration of the CIWMS would result in a minimization of the ratio kg CO_2-eq·€$^{-1}$. Expressing the results as a ratio between those two variables might make it harder to identify if only the environmental impacts, only the economic revenues or both the environmental impacts and the economic revenues are improved as a consequence of a change in the technical parameters of the system.

Moreover, since the goal of the LCA was defined at the beginning of this section from a technical perspective, making no reference to economic criteria, a physical parameter is more appropriate to allocate the environmental impacts. **T**he different material fractions emerging from the materials sorting subsystem will be transformed into a variety of goods and services, which hinders the selection of a single allocation factor based on a physical property that enables assessing the multiple functions of the goods and services delivered. Nonetheless, the mass of waste before it has been transformed into products or supplies any services could be viewed as an indicator of its potential. Hence, mass seems to be the most appropriate physical parameter to perform the allocation of the environmental impacts of a CIWMS.

In the context of a CIWMS, MSW is a substitute for natural resources; in particular, for raw materials. If the amount of energy, materials and products derived from waste that enter SS 1 rises, the incoming raw materials to subsystem 0 decrease in order to maintain the functions delivered by the CIWMS constant. Therefore, the allocation factor of the environmental impacts to the primary function of the system (*AF*) could be defined as the ratio between the mass of the MSW that is valorized in subsystems 6 and 7 ($MSW_{6,7}$), and the mass of raw materials *(RM)* and the valorized MSW, as shown in Equation 3.1.1.

$$AF=\frac{MSW_{6,7}}{RM+MSW_{6,7}} \qquad \text{(Equation 3.1.1)}$$

Summary of approaches to solve the multi-functionality problem

The LCA practitioner should ponder the disadvantages of each approach and apply the one that fits the best the goal of the study and the data availability. Table 3.1.4 sums up the main disadvantages of the application of the different methodological approaches to the LCA of a CIWMS.

Table 3.1.4. Summary of the drawbacks of alternative methodological approaches

<table>
<tr><th colspan="3"></th><th>Attributional</th><th>Consequential</th></tr>
<tr><td rowspan="2">Allocation</td><td colspan="2">By mass</td><td>a</td><td rowspan="2">Not applicable</td></tr>
<tr><td colspan="2">By economic value</td><td>a, b</td></tr>
<tr><td rowspan="4">System expansion</td><td rowspan="2">Average data</td><td>Comparison</td><td>c, e</td><td rowspan="2">Not applicable</td></tr>
<tr><td>Substitution</td><td>d, e</td></tr>
<tr><td rowspan="2">Marginal data</td><td>Comparison</td><td rowspan="2">Not applicable</td><td>c</td></tr>
<tr><td>Substitution</td><td>d</td></tr>
</table>

a. Consequences on the exported functions of alternative systems not considered
b. Hard to separately identify the response of revenues and environmental impacts to changes in the IWMS
c. Environmental impacts of the overall system; specific environmental impacts of the primary function not known
d. Negative results not coherent with waste prevention activities
e. Data uncertainty modeling alternative processes

Functional unit

Regarding the functional unit, it must describe the performance of the CIWMS in terms of the fulfillment of the primary function of the system; its aim is to quantify the performance of a system so that it can be used as a reference unit.[183]

Two thirds of the reviewed LCA studies deployed a round functional unit (1 metric ton or 1,000 metric tons of MSW), which, as highlighted by Laurent et al.,[127] simply quantifies a waste flow, without describing the performance of the IWMS. On the other hand, the functional unit of several of the reviewed studies was the incoming amount of waste into the system. Notwithstanding, the shift in the perspective of the analysis from waste (in a typical linear IWMS) to resource (in the defined CIWMS) should be reflected on the functional unit. Therefore, since the ultimate goal of a CIWMS is to reduce the extraction of raw materials, the mass of the incoming raw materials into the system could be accounted for in the definition of the functional unit of a CIWMS.

Furthermore, if waste prevention activities are considered one of the targets of a CIWMS, the amount of raw materials prevented as a consequence of the waste prevention activities should also be taken into account in the definition of the functional unit, so that scenarios with and without waste prevention activities can be compared on the same basis.

Thus, the functional unit of a CIWMS could be defined as the sum of the incoming raw materials into the system in the selected time period and in a given region plus the amount of raw materials that would have been consumed if waste prevention policies had not been implemented in that time period in that geographic area.

These recommendations are provided for a generic CIWMS that manages the variety of materials present in MSW. The discussion would be different if the system under study aimed at valorizing a specific type of waste and sending it back to the subsystem where it was generated. In this scenario, the selected functional unit could be a parameter different from the mass of the raw materials that reflects the precise primary function of the system.

Taking a CIWMS that focuses on the management of food waste as an example, its functions are to provide food for the population of a given region, and to valorize the generated organic waste into a fertilizer that is looped back into the food production subsystem. One parameter that could quantify the primary system function (waste valorization into a fertilizer) better than the incoming mass of raw materials into the system would be the area of land that is fertilized.

CONCLUSIONS

Based on the insights gained in the literature review, it was concluded that some of the shortcomings that applying the current methodological approaches to a CIWMS would entail could be solved by expanding the boundaries of a traditional linear IWMS to include upstream subsystems that link the transformation of raw materials into MSW with the waste treatment subsystems. This approach is also helpful for the analysis of waste prevention activities and the quantification of the biogenic carbon present in waste.

Waste composition will determine the functions fulfilled by the CIWMS. A CIWMS managing mixed MSW could deliver materials, energy, nutrients and even chemicals. Because of the wide range of technologies that each waste fraction can be subjected to, mathematical programming and MFA are essential to the design of CIWMSs. However, these techniques must be combined with system assessment tools, such as LCC and LCA.

Unarguably, the benefits derived from the implementation of CIWMs are due to the reduction in the consumption of natural resources. However, the economic and environmental benefits of CIWMSs are not self-evident and need to be proven by an in-depth analysis.

One of the challenges of performing the LCA of a given CIWMS lies on the multiplicity of materials that the system can handle, which translates into the great variety of services supplied and makes it hard to select the functional unit, which should reflect the shift in the perspective of the analysis from waste to resource.

Nonetheless, the main difficulty that will arise from the recommended approach will probably not stem from the integration of different methodologies, but from the upstream subsystems; considering their large size, their detailed analysis will increase the complexity of the model and the researchers' efforts needed in the modeling phase.

Table 3.1.1. Reviewed studies and applied methodologies

Reference	Methodology
Abeliotis et al.[162]	LCA
Aghajani et al.[124]	MCDM
Akbarpour et al.[184]	Optimization
Allesch and Brunner[185]	Review
Antonopoulos et al.[164]	LCA
Arena and Di Gregorio[70]	MFA and SFA
Belboom et al.[58]	LCA
Blengini et al.[65]	LCA
Boesch et al.[80]	LCA
Bovea et al.[132]	LCA
Chang et al.[21]	Review
Chang et al.[96]	LCA and optimization
Chang et al.[100]	Optimization
Chi et al.[129]	LCA
Consonni et al.[66]	Review
Consonni and Viganò[73]	Material and energy analysis
Cui et al.[145]	Optimization
Dai et al.[146]	Optimization
Eriksson and Bisaillon[83]	LCA
Eriksson et al.[166]	LCA and financial cost calculation
Erses Yay[64]	LCA
Falzon et al.[186]	LCA
Fernández-Nava et al.[187]	LCA
Fiorentino et al.[61]	LCA
Ghiani et al.[188]	Review
Giugliano et al.[115]	LCA
Herva et al.[107]	EFA, MFA and Ecological footprint
Ionescu et al.[189]	Environmental indicators
Jovanovic et al.[190]	LCA and MCDM
Juul et al.[82]	Review
Karmperis et al.[94]	Review
Koci and Trecakova[135]	LCA
Koroneos and Nanaki[136]	LCA
Laurent et al.[127, 191]	Review
Levis et al.[97, 156]	LCA and optimization
Martinez-Sanchez et al.[95]	LCA and optimization
Li and Chen[147]	Optimization
Massarutto et al.[69]	LCC
Menikpura et al.[106]	LCA, economic and social assessments
Menikpura et al.[168]	LCA

Reference	Methodology
Mirdar-Harijani et al.[99]	Optimization and social LCA
Münster et al. [84]	LCA and optimization
Ng et al.[192]	Optimization
Niziolek et al.[52]	Optimization
Pandyaswargo et al.[170]	LCA
Pires et al.[171]	LCA
Pires et al.[93]	Review
Pressley et al.[53]	LCA
Rada et al.[172]	LCA
Rigamonti et al.[173]	LCA
Rigamonti et al.[193]	Materials recovery, energy recovery and costs indicators
Sadhukhan et al.[51]	Techno-economic analysis
Santibáñez-Aguilar et al.[101]	Optimization
Santibáñez-Aguilar et al.[102]	Optimization
Satchatippavarn et al.[194]	Optimization
Song et al.[138]	LCA
Srivastava et al.[148]	Optimization
Srivastava et al.[103]	Optimization
Suwan and Gheewala[130]	LCA
Tabata et al.[98]	LCA and optimization
Tan et al.[141]	Optimization
ThiKimOanh et al.[142]	Optimization
Tonini and Astrup[50]	LCA
Tonini et al.[131]	LCA
Tonini et al.[108]	MFA, SFA, EFA, optimization
Tulokhonova and Ulanova[63]	LCA, economic and social assessments
Tunesi[174]	LCA
Vadenbo et al.[104, 105]	MFA, LCA, optimization
Wang et al.[139]	LCA
Wang et al.[149]	Optimization
Zaccariello et al.[195]	MFA and efficiency indicators
Zhou et al.[151]	Optimization
Zhu and Huang[152]	Optimization

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CHAPTER 3.2

RESOURCE USE AND CARBON EMISSIONS

"Remember that all models are wrong; the practical question is how wrong they have to be to not be useful."

George Box, English mathematician (1919-2013)

Chapter 3.2 is constituted by this paper:

Cobo, S.; Dominguez-Ramos, A.; Irabien, A. Minimization of resource consumption and carbon footprint of a circular organic waste valorization system. *ACS Sustainable Chem. Eng.* **2018***, 6*, 3493-3501; DOI: 10.1021/acssuschemeng.7b03767.

"The transition to a circular economy could relieve the pressure on the ecosystems to meet the demand for natural resources. Thus, the implementation of systems that strengthen the connection between waste management and the transformation of raw materials, hereafter referred to as Circular Integrated Waste Management Systems (CIWMSs), should be promoted.[1] CIWMSs provide a solid framework to assess the consequences of the recirculation of the waste components.

The application of the concept of CIWMSs to the management of organic waste, also known as bio-waste, is particularly challenging because of the diversity of materials that it may contain and its high moisture content. Nevertheless, due to its carbon (C) rich composition and the presence of nutrients such as nitrogen (N) and phosphorus (P), energy and nutrients can be produced from organic waste. State-of-the-art research focuses on the production of chemicals and fuels from organic waste.[2-4]

Closing the loop of nutrients to a certain extent would help us secure the food supply. N and P are essential to the metabolism of plants, and by extension, to agriculture and food production systems. Paradoxically, human tampering with the N and P biogeochemical cycles, mostly due to the inefficient production and use of fertilizers, leads to eutrophication problems that affect the aquatic food chains,[5] whereas the remaining accessible reserves of clean phosphate rock could run out as soon as 50 years from now.[6] Although N is an abundant element in the atmosphere, the synthetic production of N-based fertilizers is an energy intensive process.[7] Hence, the substitution of the N recovered from waste for N-fertilizers could potentially contribute to climate change mitigation.

Despite the benefits that a circular economy of nutrients offers, without policies to support the circularity of resources, this is not likely to become the priority of the stakeholders involved in waste management. One of the local resources that has more influence on the configuration of integrated waste management systems is land. Moreover, the ecosystem around the area that has been used as a landfill is severely degraded and the site has very limited applications. Although landfills can never be completely avoided,[8] a well-designed CIWMS should minimize their use.

This discussion is of interest for the region of Cantabria, located in the northern coast of Spain, since the government is studying the possibility of expanding the existing landfill to guarantee its lifespan. The municipal organic waste generated in Cantabria is sorted out from the inorganic fraction of mixed household waste and composted at a mechanical-biological treatment facility. The agricultural application of the resulting bio-stabilized material entails certain environmental risks associated with the transfer of heavy metals and organic pollutants to the soil.[9] After the application of Directive 2008/98/EC,[10] which was transposed into the Spanish Law 22/2011 about waste and polluted soil,[11] a distinction between the bio-stabilized material and the compost generated from the source-separated organic waste is made; the former cannot be applied to land. However, the Cantabrian waste managers were granted an authorization to continue with this practice.[12] Its expiration in early 2018 poses the unanswered question of how to manage the organic waste generated in Cantabria.

The objective of this work is twofold: i) to propose a methodological framework to address some of the sustainability challenges related to the management of organic waste, and ii) to assist decision-makers in selecting the optimal configuration of a CIWMS that aims at valorizing the organic waste generated yearly in Cantabrian households. The optimal configuration of the system is defined as the combination of nutrient and energy recovery technologies that minimize these three objective functions: climate change impacts, land use and consumption of raw materials.

The potential of systems engineering to establish a connection between resource and waste is recognized in the literature.[13-15] However, most of the studies that seek to optimize waste management systems only consider environmental criteria.[16-20] The novelty of this research is that the problem is also approached from the perspective of the minimization of the consumption of natural resources. To the best of the authors' knowledge, waste management has never been analyzed from the viewpoint of a CIWMS that includes within its boundaries the upstream processes responsible for the delivery of waste and the transformation of the recovered waste components.

The chapter is structured as follows. First, the system under study is described. Then, the methodological approach is defined, and the hypothesis regarding the life cycle model and the problem formulation are provided. Finally, the results are presented and discussed.

SYSTEM DESCRIPTION

The superstructure shown in Figure 3.2.1 accounts for the alternative technologies to handle organic waste within the studied CIWMS. The unit processes whose input flow is a decision variable have been shaded in green. The solution to the optimization problem will determine the flows of organic waste that must be sent to each unit process to achieve optimal results. The CIWMS described in Figure 3.2.1 also includes the agricultural application of the products recovered from organic waste, and the remaining Cantabrian food production and consumption subsystem, responsible for the generation of organic waste. The dotted line in Figure 3.2.1 represents the boundary that separates the CIWMS from the environment. They are connected through the consumption of natural resources and the emission of environmental burdens of the system, which have not been shown in Figure 3.2.1 because of their large number.

Over half of the organic waste generated in Cantabria is food waste (see waste composition in Appendix A of the Supporting Information). It ends up in the Cantabrian bins mixed with other organic materials (yard waste and wood) and inorganic residues. The CIWMS comprises two waste collection systems: commingled waste and source separated organic waste (SS-OW).

The organic waste recovered from the mixed waste stream (mix-OW) is separated from the inorganic materials via trommel screen. Ferrous and non-ferrours metals are previously sorted from the mixed waste stream with magnetic and Eddy current separators respectively. The

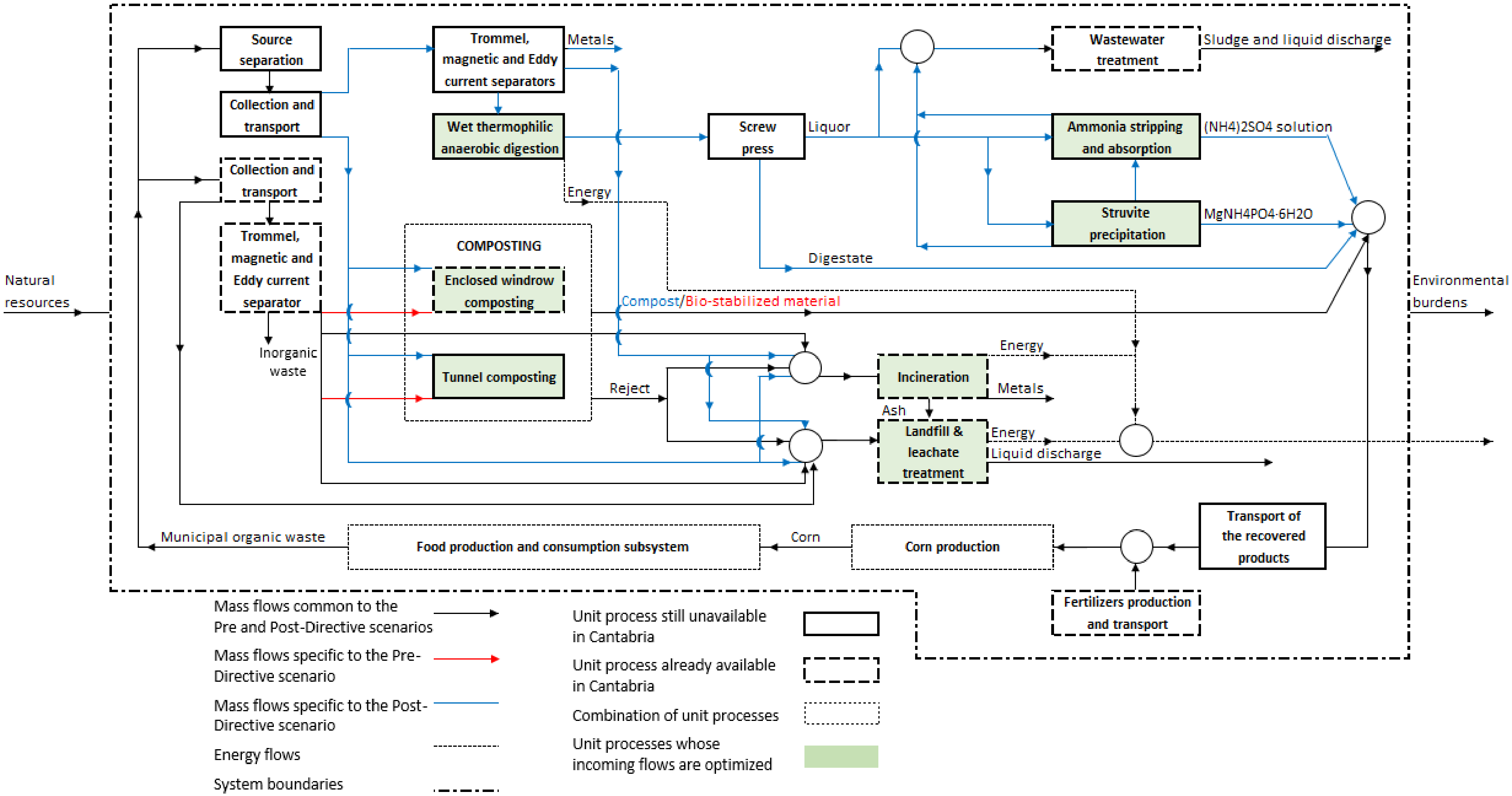

Figure 3.2.1. System boundaries and superstructure

processing of the metals and the rest of the inorganic materials is outside the scope of this study. The SS-OW does not require any pretreatment, except for the fraction that is subjected to anaerobic digestion,[21-25] which requires a trommel screen to remove the inorganic materials and avoid the transfer of toxic elements from the digestate to the soil. The composting technologies do not require any specific pretreatment, because the rejects are screened after the final curing or maturation phase.

Two types of composting technologies were studied: enclosed windrow and tunnel composting. Both technologies count with a biofilter to treat the gases and a turner that agitates the feedstock to ensure its aerobic degradation.

The modeled anaerobic digestion process was based on a wet one-stage thermophilic anaerobic digestor. The generated biogas is combusted to produce electricity, and the digestate is dewatered through a screw press. The nutrients in the resulting liquor can be recovered via ammonia stripping and absorption (as ammonium sulfate) or struvite precipitation. Alternatively, the liquor may be sent to an existing wastewater treatment plant for sewage water, which can also receive the residual liquid from the above-mentioned unit processes.

The organic waste can also be incinerated or disposed of in a non-hazardous landfill, along with the rejects of the composting processes and the organic waste rejected at the pretreatment stage of the anaerobic digestion. Incineration is modeled as a grate furnace with wet flue gas cleaning, SNCR and activated carbon to treat the flue gas. The released energy is sold as electricity. The fly ash undergoes a solidification/stabilization process with cement and water prior to its disposal in a mono-landfill, whereas the bottom ash is disposed of in the non-hazardous landfill after the removal of metals with magnetic and Eddy current separators. The landfill has systems for leachate collection and treatment and biogas combustion and treatment for power generation.

The products generated from the organic waste (compost, digestate, struvite and ammonium sulfate) are applied to land to grow corn. This cereal was selected because it is the main fodder crop in Cantabria.[26] The nutrients recovered from the organic waste are not enough to fertilize the land available in Cantabria for corn production. Hence, the use of industrial fertilizers is imperative. However, as the circularity of nutrients increases, the need for industrial fertilizers decreases.

The produced corn enters the food production and consumption subsystem, which accounts for all the food commodities. It is mainly used as forage for livestock, but it may also be processed by the food industry or directly sold to consumers.

METHODS

Once the superstructure of the system and its boundaries were established, a mass balance model was developed in GAMS 24.7.1. Figure 3.2.2 provides an overview of the sequence of methodological steps taken.

The Life Cycle Assessment (LCA) methodology was followed to account for the consumption of natural resources and the emission of environmental burdens of the system. An individual LCA was carried out for each unit process, and the results were exported to GAMS as model parameters. The EASETECH (Environmental Assessment System for Environmental Technologies) 2.3.6 software[27] enabled i) obtaining LCA results for the unit processes concerning the treatment of solid organic waste and the land application subsystem, which are dependent on the waste composition, and ii) performing a material flow analysis (MFA) of the system. Appendix B compiles the parameters and assumptions made, including the data taken from the literature to model the trommel separation,[12,28,29] anaerobic digestion,[30,31] struvite precipitation,[32,33] ammonia stripping and absorption[34] and transport[35] unit processes.

The DNDC (Denitrification-Decomposition) software models the C and N biogeochemical cycles in agricultural ecosystems.[36] DNDC 9.5 was used to predict corn yield, C sequestration, nitrate leaching losses and emissions of C and N gases associated with corn production and the application of the fertilizing products to land. These data were subsequently introduced in EASETECH, to be translated into environmental impacts. More information about the modeling procedure for these subsystems can be found in Appendix C, which includes the DNDC input parameters taken from the literature.[37-40]

Modeling in detail the Cantabrian food production and consumption subsystem is outside the scope of this work. It was described with the data provided by Ivanova et al.[41] in their study on the environmental footprints of European regions.[42]

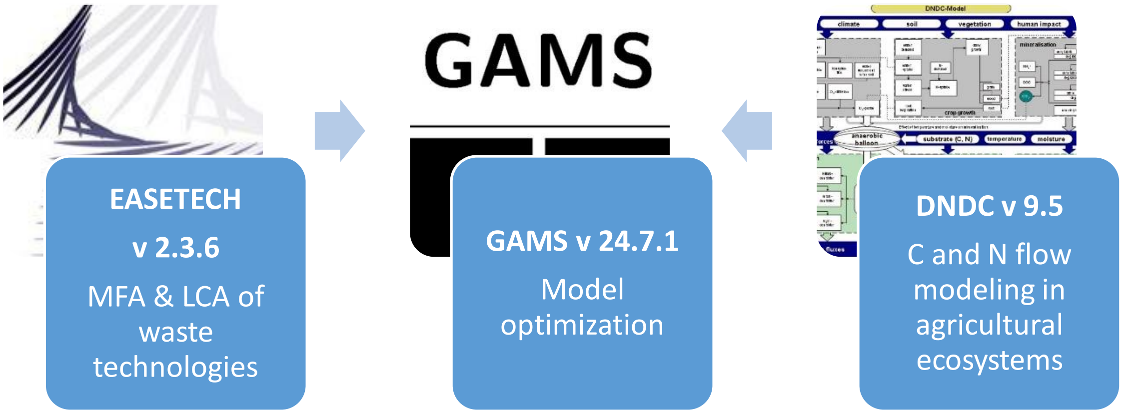

Figure 3.2.2. Simplified methodological steps

LIFE CYCLE MODEL

The goal of a CIWMS is not waste treatment, but waste valorization through the recirculation of the waste components to the upstream subsystems. Thus, the primary function of the studied system is land fertilization (which is achieved by means of the combined application of industrial fertilizers and the products obtained from the valorization of organic waste), whereas the secondary system function is energy generation. The selected functional unit to perform the LCA of the system is the area available to grow corn in Cantabria (4810 ha).[43]

The direct substitution method is applied by expanding the system boundaries to include the generation of electricity from the Spanish grid mix. The Spanish legislation prioritizes electricity from renewable resources over electricity derived from fossil fuels. Although the biogas produced at landfills and anaerobic digestion facilities is considered a renewable energy source, the electricity generated from waste incineration does not have priority access to the grid.[44] Nonetheless, foreseeing the consequences of connecting another power source to the grid is outside the scope of the study, whose modeling framework is based on an attributional approach. Hence, a 100% substitution ratio was assumed.

The model applied to characterize the impact of each emission was the hierarchical 100-year perspective of ReCiPe 1.11. The results of the global warming impact category strongly rely on the hypothesis that only the biogenic C present in animal and vegetable food waste is considered neutral. Neutrality implies that the CO_2 that is withdrawn from the atmosphere during photosynthesis is accounted for as negative CO_2 in the life cycle inventory. Since the upstream processes concerning the production of other materials present in the organic waste, such as paper, were not modeled, it is not correct to consider the environmental benefits associated with the life cycle stage involved in the capture of CO_2 by biomass, while the environmental impacts of the other life cycle stages of these materials are not quantified.

One of the main limitations of the proposed model is that the life cycle impacts related to capital goods were not considered. The study performed by Brogaard and Christensen[45] concluded that, although capital goods should always be included in the LCA modeling of waste management, their contribution to the results of the global warming impact category may be negligible.

PROBLEM FORMULATION

A Mixed Integer Linear Programming problem was formulated for the optimization of the material flows that enter each unit process in Figure 3.2.1, according to the following indicators that were considered as objective functions to be minimized:

- The global warming impacts of the system (GW).
- The consumption of non-renewable raw materials required for the operation of the system (NR-RM). This definition excludes the raw materials used for energy production, such as coal.
- The landfill area where household organic waste and the rejects and ashes generated from the management of organic waste are disposed of (LFA).

For the set *i* of indicators and the set *j* of unit processes, the objective functions (OF_i) were calculated multiplying the amount of waste that each unit process handles (W_j) by the indicators (S_{ij}) related to the treatment of 1 metric ton of waste by each unit process, as shown in Equation 3.2.1.

$$OF_i = \sum_{j=1}^{n} W_j \cdot S_{ij} \qquad \text{(Equation 3.2.1)}$$

The problem is subject to these restrictions:

- The maximum amount of biodegradable waste sent to landfill. Directive 1999/31/EC[46] establishes that biodegradable municipal waste going to landfills must be reduced to 35% of the total amount produced in 1995.

- The minimum amount of organic waste recycled. One of the objectives set by the Cantabrian waste management plan is to recycle 50% of organic waste before 2020.[12]
- A waste stream of a given composition cannot be split between tunnel and windrow composting.
- SS-OW and mix-OW cannot be mixed in the composting process.

The Ɛ-constraint method was applied for the multi-objective optimization of the problem.[47]

PERFORMANCE INDICATORS

The fraction of the N present in waste that is recovered and assimilated by corn could be an appropriate indicator to measure the circularity of N within the system. However, its value does not only rely on the efficiency of the technological system, but also on the ability of plants to capture nutrients from the soil. Thus, a circularity indicator based solely on parameters under the control of the decision-makers was developed: the fraction of N that is recovered from waste and applied to land with respect to the N present in the collected waste. It is hereafter referred to as N recovery, and it is expressed as kg of recovered N per kg of N in waste. This indicator was not selected as an objective function because the consequences of increasing the circularity of N cannot be foreseen *a priori;* it might not lead to a minimization of the environmental impacts and the consumption of natural resources.

Another indicator was developed to compare the circularity of N to the wasted N within the system: the efficiency of the corn N uptake (η). It was defined as the fraction of N that is absorbed by corn with respect to the available N for corn production within the system, which is the sum of the N that comes into the system via fertilizers intended for corn production, and the N present in the collected waste. The amount of available N that is not taken up by corn ($1 - \eta$ times the available N) is lost throughout the system. These losses can be stored in soil or released to the environment as gas emissions or leachate. The fractions of N that end up in each sink depend on the type of product that is applied to soil, as shown in Appendix C. The N losses of the food production and consumption subsystem, which are not quantified, may also come into other systems as sewage sludge or industrial waste.

DEFINITION OF SCENARIOS

The collection of SS-OW requires the active participation of citizens, which is the reason it is hard to estimate the extension of its implantation. It is assumed that the composition of the SS-OW is 98% organic matter and 2% impurities.[48] Different source separation rates (SSRs) were assessed: 20%, 50% and 80%. For each studied SSR, a pre-Directive and a post-Directive scenario (before and after the expiration of the Cantabrian authorization to apply to land the compost produced from the mix-OW) were analyzed.

The expiration of this authorization implies that only SS-OW can be recycled. Thus, the recycling objective of 50% of the organic waste will not be achieved unless at least a 50% SSR is implemented in the post-Directive scenarios. Consequently, only two of the six studied scenarios comply with the legislation and all the restrictions of the model: the post-Directive scenarios with 50% and 80% SSRs.

RESULTS AND DISCUSSION

The GW and the consumption of NR-RM of the food production and consumption subsystem (excluding corn production) are assumed to be constant (1.56 million metric tons of CO_2-eq and 0.69 million metric tons of NR-RM, according to the references)[41,42] regardless of the value of the optimized variables. Their values are three orders of magnitude larger than those of the remaining system. Hence, the results presented in this section do not include the values associated with the food production and consumption subsystem.

Figure 3.2.3 shows the normalized values of the objective functions obtained as a result of the three single-objective optimizations performed for each scenario. The minimal values of each objective function are obtained for the highest SSR, because the flows of waste that the system manages are smaller compared to those of lower SSRs, on account of the fewer inorganic materials that the waste streams contain.

The minimal GW is achieved at the expense of maximizing the consumption of NR-RM and the LFA. In the pre-Directive scenarios the minimization of the consumption of NR-RM requires an increase in the LFA and vice versa, whereas in the post-Directive scenarios the model responds similarly to the minimization of the NR-RM and the LFA. These results demonstrate that it is pertinent to use the multi-objective optimization technique to solve the problem.

Figure 3.2.4 shows the combination of technologies required for the minimization of the objective functions in all the scenarios, as well as the flows of solid organic waste processed by each of them. The flows of processed organic waste are lower for scenarios with low SSRs because part of the organic waste present in the mixed waste ends up in the inorganic waste stream after the trommel separation required for the pretreatment of mixed waste.

The ranking of the unit processes according to their GW agrees with the results found in the literature for organic waste,[49-52] although the specific values of their carbon footprints differ among publications, given that they are highly dependent on the assumptions made and the waste composition.[53] Regarding the management of the liquid digestate, the ammonia stripping and absorption unit was selected as the best alternative to minimize the GW of the system.

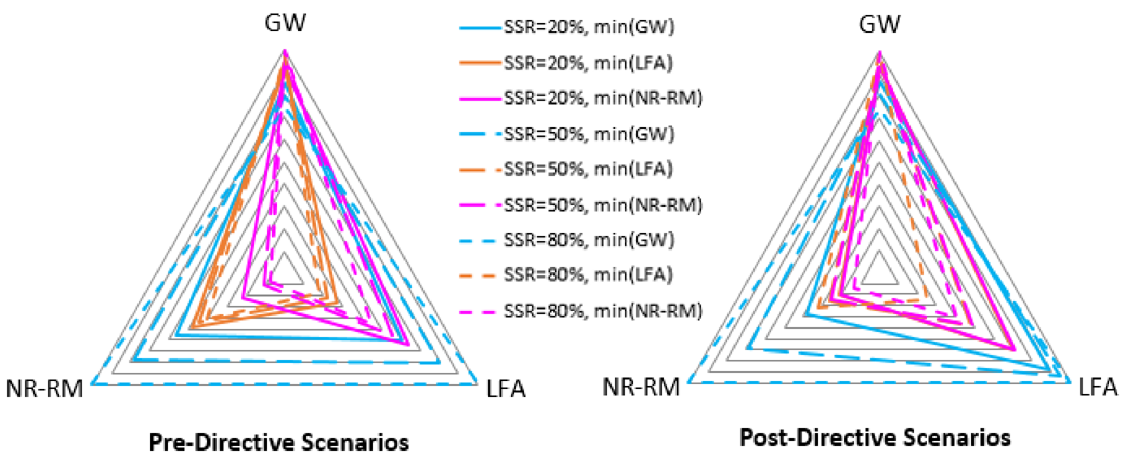

Figure 3.2.3. Normalized results for the minimization of the objective functions

As Figure 3.2.4 shows, the shift from pre-Directive to post-Directive scenarios is mostly reflected on the fact that, since the mix-OW cannot be composted, it is incinerated instead. Figure 3.2.4 also depicts the performance indicators of the studied scenarios. Since the production of fertilizers is very energy intensive, the system configuration that minimizes its GW achieves the highest N recovery rates, which leads to a decrease in the reliance on industrial fertilizers.

The scenarios with the lowest N recovery rates, which rely on incineration to a greater extent, minimize the consumption of NR-RM because of the consumption of NR-RM that is avoided as a result of the electricity from the grid mix that is assumed to be displaced.

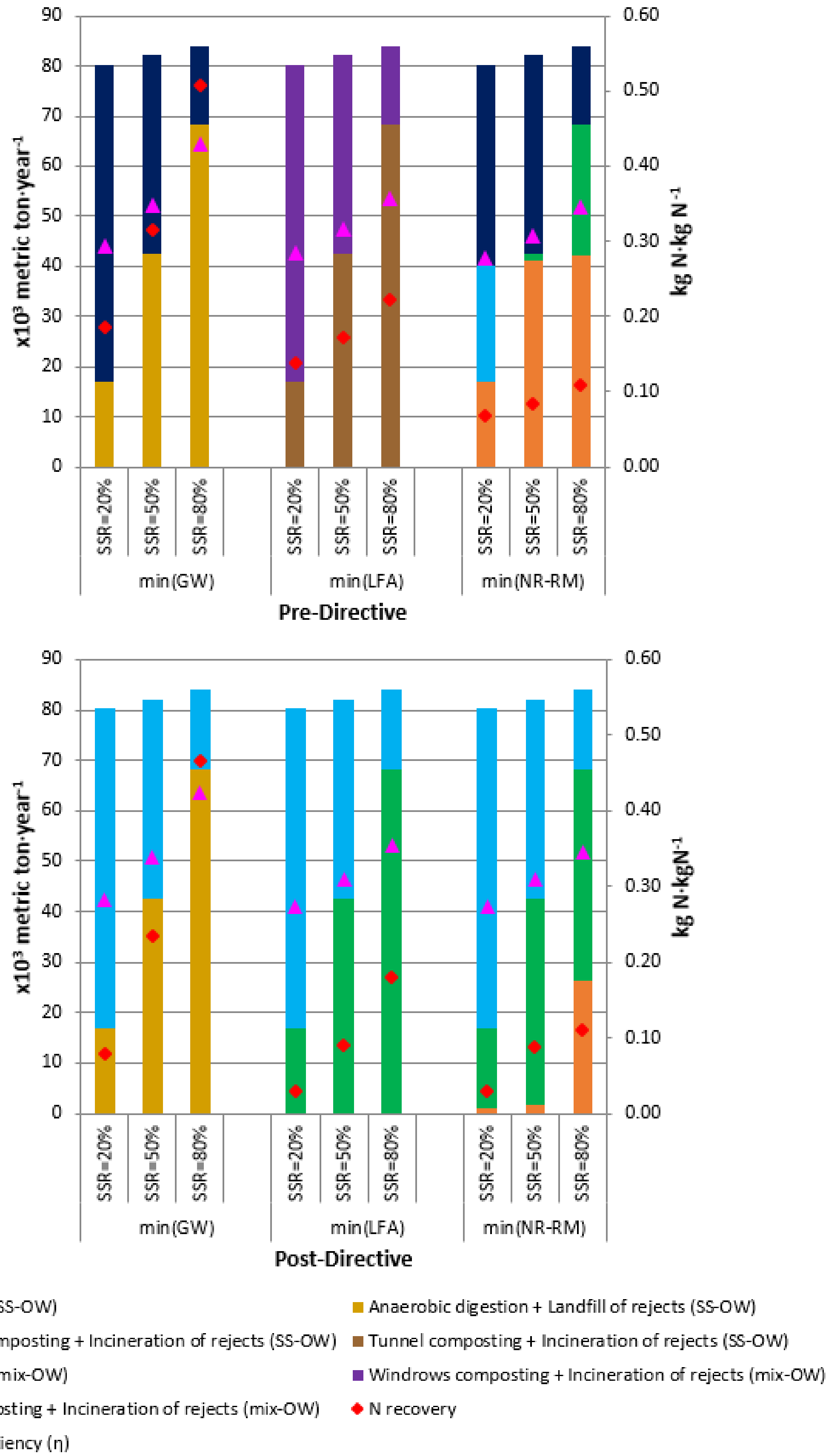

Figure 3.2.4. Mass flows of organic waste to each unit process and performance indicators

The N recovery and the η increase as the SSR increases, although these parameters are not directly proportional. As the simplified N flow analysis illustrated in Figure 3.2.5 proves, the scenario with the highest N recovery is not necessarily the scenario with the highest η; i.e., the N losses throughout the system may be larger for the scenario with the highest N recovery. This happens because, as noted by Yoshida et al.[54] it is easier for crops to absorb N from fertilizers than from the products derived from organic waste.

The Pareto optimal solutions for each studied scenario are shown in Figure 3.2.6. Each Pareto point corresponds to a given system configuration. The system configurations corresponding to the points with the minimal values of the objective functions are those depicted in Figure 3.2.4.

It can be seen in Figure 3.2.6 that, as the SSRs increase, the range of values of the objective functions increases too; i.e., the minimal values of the objective functions decrease as the SSRs increase, but this improvement is accomplished increasing the values of the other objective functions associated to those Pareto points.

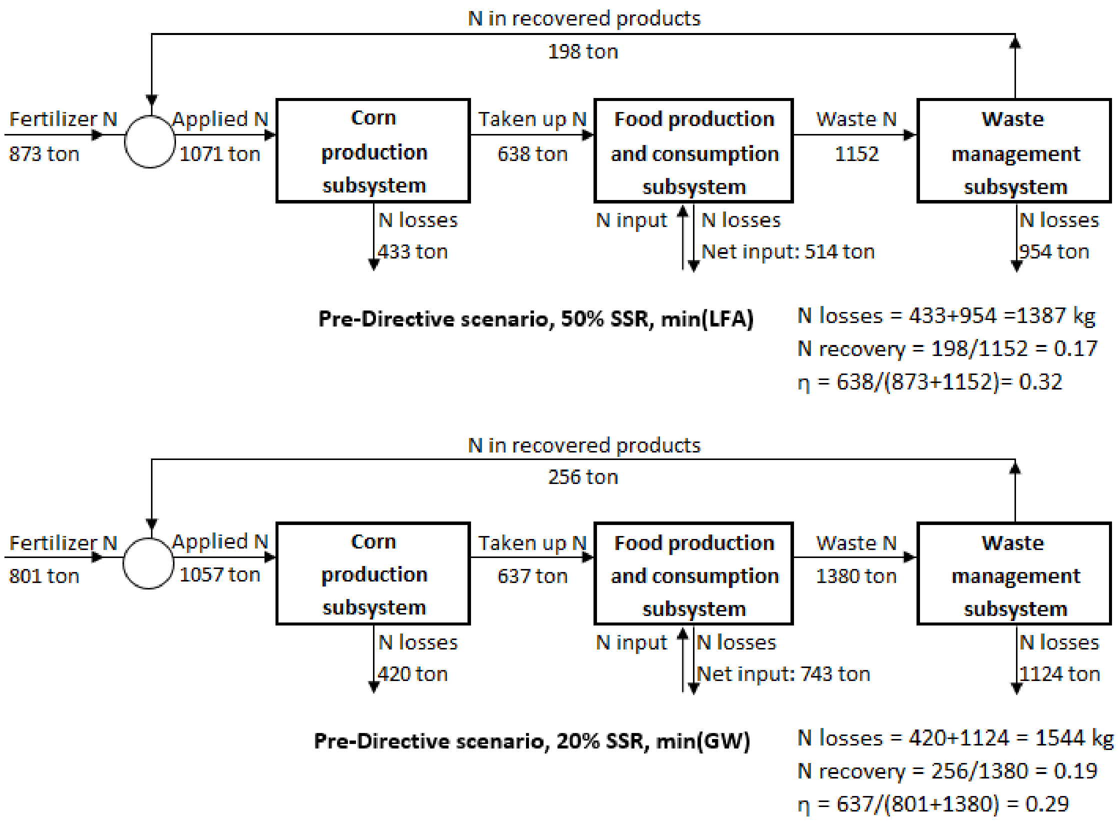

Figure 3.2.5. N flow analysis of different scenarios (metric ton)

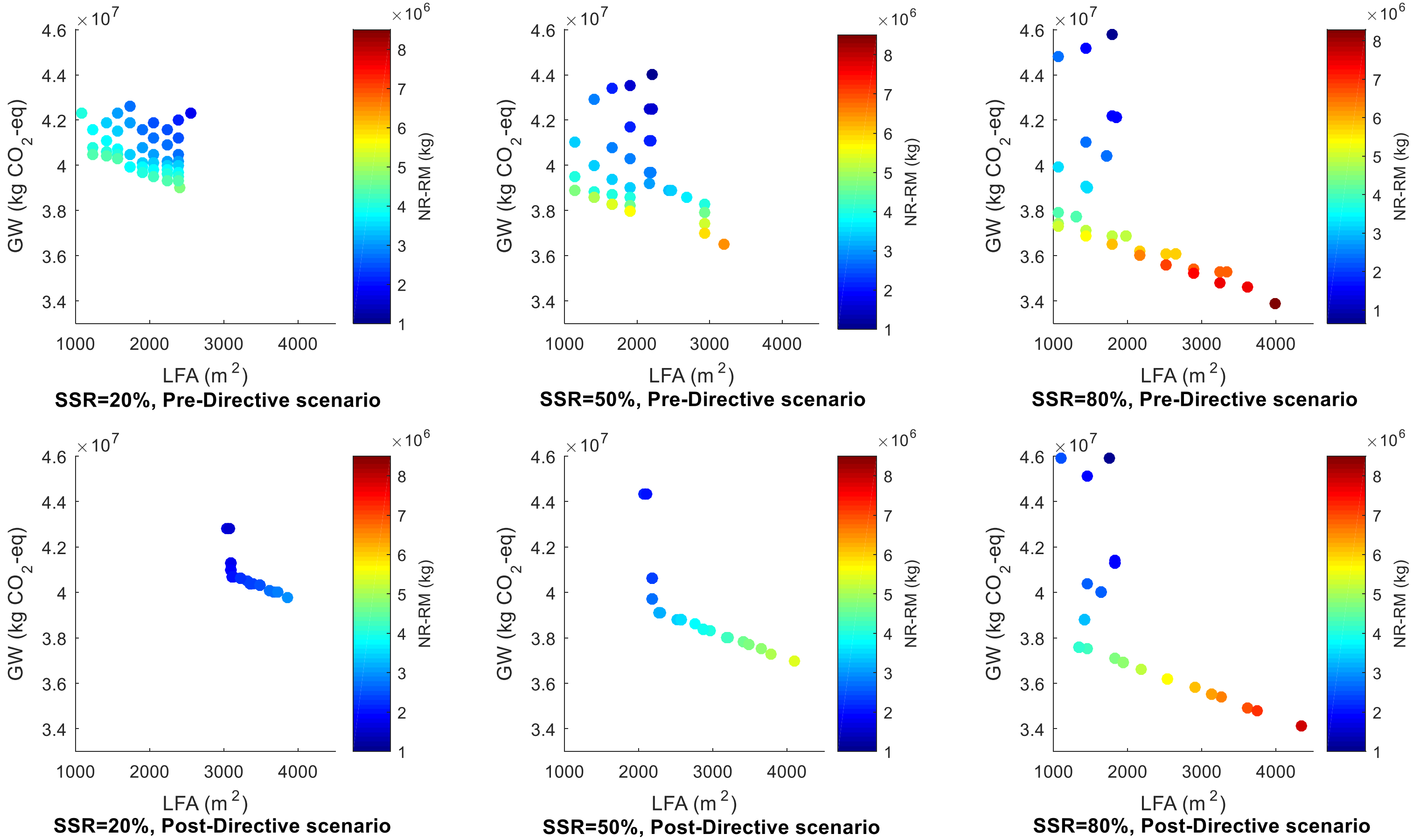

Figure 3.2.6. Pareto optimal solutions

The consumption of NR-RM is lower in the post-Directive scenarios because the fraction of organic waste that is incinerated is larger than in the pre-Directive scenarios, and thus, the avoided consumption of NR-RM, is also larger.

The worse performance of the post-Directive scenarios in terms of the values of the GW and the LFA can be attributed to the fewer possible system configurations available in comparison to the pre-Directive scenarios, because of the additional restrictions of the model.

Research relevance and shortcomings

This research demonstrates that the proposed methodological approach provides a valuable framework for the consideration of circularity and sustainability criteria in the design of CIWMSs. Furthermore, it provides a basis to further investigate the consequences of nutrient looping.

Multiple optimal system configurations for the management of organic waste in Cantabria were presented; it is up to the regional decision-makers to weigh the importance of the identified objective functions and select the desired range of operation values. Although the retrofit of the existing Cantabrian facilities is essential to abide by the current legislation, it is imperative that future work includes an economic evaluation and an assessment of the uncertainty of the results. Furthermore, other waste fractions should be integrated within the developed model so that the restrictions related to the capacity of the unit processes that are not exclusive of organic waste can be taken into account. Alternative system configurations that contemplate new applications for bio-stabilized materials are also worth exploring.

Beyond the applicability of the results to solve a real problem, the interest of the research resides in the conclusions about the connection between the circularity of resources and other sustainability aspects that can be drawn. The complete circularity of the nutrient flows within any CIWMS is infeasible, because it does not only depend on the efficiency of the recovery technologies, but also on the ability of plants to capture nutrients. Since crops absorb N from fertilizers more efficiently than from the products recovered from organic waste, a system configuration with a high N circularity might have larger N losses (and consequently, higher eutrophication impacts) than a system that consumes more industrial fertilizers.

Moreover, in this case study the minimization of the consumption of the NR-RM leads to the system configuration with the lowest N recovery rates. Hence, this work proves that closing the material loops to a greater extent does not necessarily go hand in hand with a decrease in the overall consumption of resources or the emission of environmental burdens; such claims must be supported by a thorough analysis."

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CHAPTER 3.3

CIRCULARITY AND ENVIRONMENTAL ASSESSMENT

"If you cannot measure it, you cannot improve it."

Lord Kelvin, British physicist and mathematician (1824-1907)

This subsection contains the following paper:

Cobo, S.; Dominguez-Ramos, A.; Irabien, A. Trade-offs between nutrient circularity and environmental impacts in the management of organic waste. *Environ. Sci. Technol.* **2018***, 52*(19), 10923-10933; DOI: 10.1021/acs.est.8b01590.

"In the context of a boom of initiatives promoting a circular economy within the European Union,[1-3] it is the responsibility of researchers to provide policy-makers with the data and tools needed to make informed decisions. Measuring the circularity of resources is key to assessing the performance of a circular economy.

Several approaches have been presented to tackle this challenge. One study defined a global circularity indicator as the share of material inputs into the global economy that are cycled, subsequently estimating that the global economy was 9.1% circular in 2015.[4] Although this indicator provides insight into the global materials metabolism, policy implications cannot be directly derived from it. Instead, an indicator that can be applied to systems design and operation is of more interest to the policy makers.

Some authors suggest that circularity indicators should capture how the differences between the physico-chemical properties of the recovered waste components and the primary resources they displace affect their substitution ratio.[4-7] Accordingly, Moriguchi[5] pointed out that the reduction in the requirement for primary resources could be a good indicator of circularity.

However, this does not necessarily entail that more waste components are being recovered; it could be the consequence of an increase in the eco-efficiency of the system.

Haupt et al.[6] suggested that open-loop and closed-loop recycling rates that reflect the efficiency of the recycling processes and the type of application of the recycled components in their next life cycle stage should be used as performance indicators for a circular economy.
The duration of material retention within a system has also been recommended as an indicator of circularity.[7] Following this line of thinking, the Ellen MacArthur Foundation proposed the lifetime of a product as one of the parameters used to calculate its circularity indicator.[8] Although this indicator is useful for companies, it does not provide information about the circularity of the components of the product, since it does not consider their entire life cycle.

The described indicators do not correlate with the quality of the recovered components and they do not reveal how much of the recovered components are consumed again; i.e., to what extent the loop is closed.

The methodology proposed by Cobo et al.,[9] which enables us to track waste components within a Circular Integrated Waste Management System (CIWMS), might contribute to overcome these limitations, since CIWMSs encompass not only waste management, but also the processing and consumption of the components recovered from waste and the external raw materials that eventually become waste.

This framework is applied to the study of the management of municipal organic waste in the region of Cantabria, in the north of Spain. The organic waste generated in Cantabria ($83.5 \cdot 10^3$ metric ton in 2014) is collected with other discarded household inorganic materials. The organic waste that is sorted out at the regional mechanical-biological treatment facility is subjected to a windrow composting process. Nonetheless, Directive 2008/98/EC[10] does not allow the land application of the bio-stabilized material derived from the composting of the organic waste separated from the mixed waste stream (mix-OW); only the organic waste that has been source separated (SS-OW) can be recycled. The expiration of the regional authorization that permitted the sale of the bio-stabilized material as compost until 2018[11] makes it impossible for the current waste management system to comply with the legal restraints. The need to retrofit the system represents an opportunity to implement new circularity practices. The interest of recycling lies in the nutrients it contains.

This study focuses on three essential nutrients to soil amendment: carbon (C), nitrogen (N) and phosphorus (P). Enhancing the circularity of these nutrients within a CIWMS *a priori* seems to be a strategy that will contribute to closing their natural biogeochemical cycles by avoiding the accumulation of nutrients in one of the Earth's subsystems (atmosphere, hydrosphere, biosphere or lithosphere) at a rate faster than the ecosystems can sustain. Thus, the relevance that a circular economy of nutrients might have to global sustainability challenges should not be underestimated. On the one hand, the forthcoming peak P production, due to the depletion of the global rock phosphate reserves, threatens future food security;[12] on the other, the anthropogenic interference with the C and N biogeochemical cycles to meet the energy and food demands has already caused the transgression of the estimated climate change and N cycle planetary boundaries within which humanity is expected to operate safely.[13]

Since the nutrient cycles interact with each other,[14] promoting the circularity of one nutrient might have consequences on the biogeochemical cycles of the others. For instance, increasing Soil Organic Carbon (SOC) stocks may exacerbate N_2O emissions,[15] and an increased availability of reactive N may lead to C sequestration because of biomass growth.[16] Another counter-effect related to the land application of the products recovered from organic waste is the accumulation of surplus P in agricultural soils, because the N:P ratio in organic fertilizers is lower than the N:P ratio required by crops.[17-19]

The circularity of C, N and P within a CIWMS and the main impacts associated with the emissions of these elements to the environment – global warming (GW), marine eutrophication (MEU) and freshwater eutrophication (FWE) – must be jointly analyzed. Although the recovery of nutrients is a subject that is drawing the attention of the scientific community,[20-24] the trade-offs between these indicators have not been systematically explored in the literature yet. Therefore, the objectives of this chapter are the following:

- To propose a circularity indicator that can be applied to any non-renewable resource and accounts for the extended service of the components recovered from waste.
- To optimize the organic waste management system in the region of Cantabria, setting as objective functions the maximization of the circularity indicators of C, N and P, and the minimization of the GW, MEU and FWE impacts.

METHODOLOGY

Material Flow Analysis (MFA), Life Cycle Assessment (LCA) and multi-objective optimization were applied to determine the optimal configuration of the Cantabrian CIWMS aiming at nutrient recovery from OW. A superstructure comprising the combinations of unit processes that could emerge as a result of the optimization was proposed, as shown in Figure 3.3.1. The unit processes that already belong to the Cantabrian waste management system are represented with a discontinuous line.

Superstructure description

The products recovered from organic waste were assumed to be applied to land to grow corn, the main fodder crop grown in Cantabria.[25] The superstructure comprises a set *j* of unit processes for the management of organic waste and a set *k* of corn production unit processes. The unit processes that can handle the solid organic waste are wet thermophilic anaerobic digestion, windrow composting inside an enclosed building, composting inside a tunnel reactor, incineration and landfill. The ammonia stripping and absorption and the struvite precipitation unit processes recover nutrients from the liquid digestate produced in the anaerobic digestion, which only processes SS-OW after it has been pretreated.[26-30] The remaining liquor is sent to a wastewater treatment plant. Incineration and landfill can also handle the rejects generated by the other unit processes. It is assumed that all the waste processing units are in the same facility. A detailed description of these unit processes can be found in Cobo et al.[31]

The nutrient uptake efficiencies of corn (shown in Appendix C of the Supporting Information) differ for each type of applied product (bio-stabilized material, compost, digestate, struvite and ammonium sulfate). P is in excess with respect to the amount of N required by corn in all the recovered products except for ammonium sulfate. Consequently, the nutrient flows were modeled so that the optimal approach to corn production can be either based on one of these strategies or on a combination of them:

S1) Application of the amount of recovered product needed to cover the corn N requirements. Unless ammonium sulfate is recovered, excess P is applied to soil, leading to FWE.

S2) Application of the amount of recovered product needed to cover the corn P requirements. The N requirements are fulfilled with an industrial fertilizer (ammonium nitrate).

S3) Application of industrial N and P fertilizers (ammonium nitrate and diammonium phosphate).

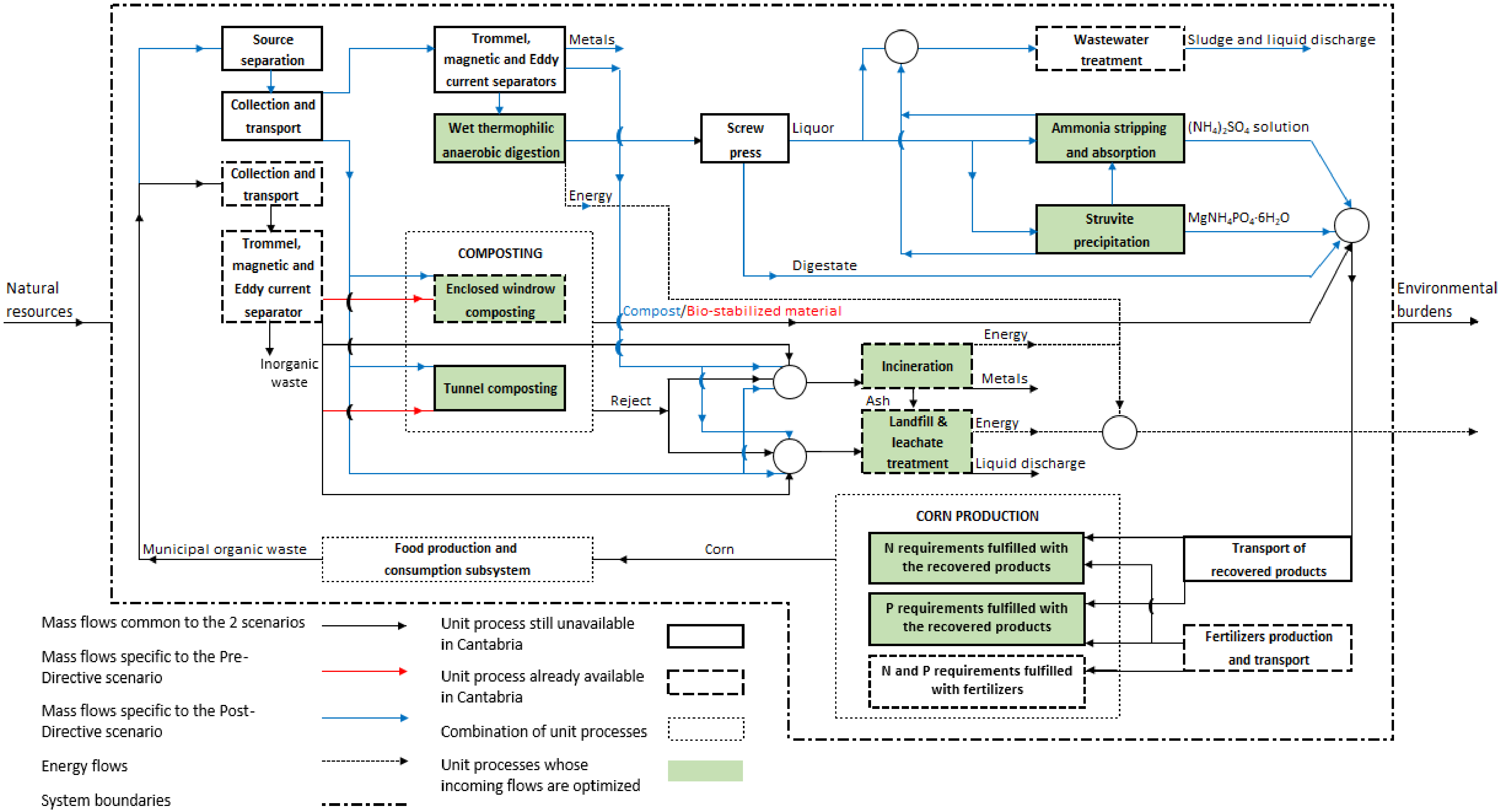

Figure 3.3.1. System boundaries and superstructure

The alternative combinations of the corn production unit processes that can arise from the application of these strategies are shown in Figure 3.3.2. The N and P requirements of corn are defined as the amounts of these nutrients that yield the maximum average annual crop production that can be achieved in a 100-year timeframe with industrial N and P fertilizers. Assuming an 80% collection rate of the produced corn grain, it corresponds to a net production of 7.11 metric tons of corn grain per ha per year.

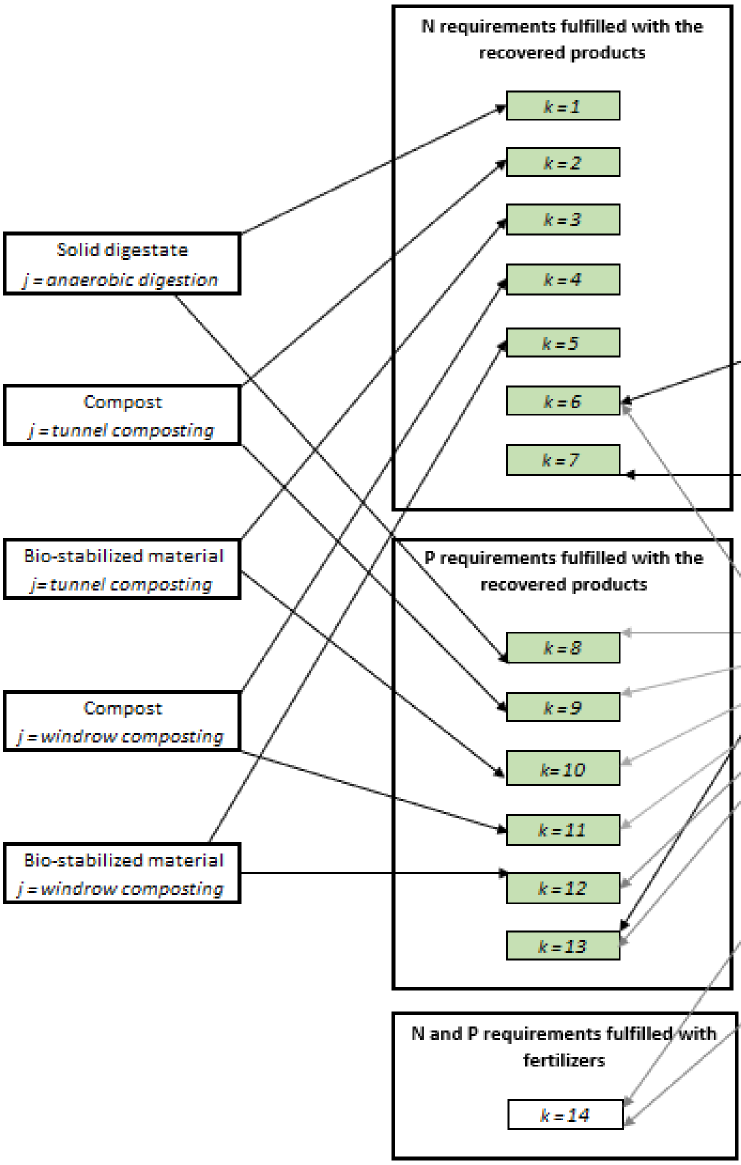

Figure 3.3.2. Possible combinations of inputs to the corn production subsystem

Data flow

A modular LCA approach, where the LCA of the individual unit processes of the system is carried out,[32,33] was performed. The unit processes concerning the management of solid organic waste were modeled with EASETECH 2.3.6,[34] which provided their environmental impacts. The nitrate and phosphate leachate, the emissions of CO_2, N_2O and NO, the amount of Dissolved Organic Carbon (DOC) consumed by soil microorganisms, the flows of N and P taken up by corn and the amount of nutrients stored in soil per hectare of cultivated corn were calculated with DNDC 9.5.[35] These results were transferred to EASETECH 2.3.6, where the environmental impacts associated with the land application of the recovered products and corn production were calculated.

The results obtained with DNDC and EASETECH were exported as parameters to GAMS (General Algebraic Modeling System) 24.7.1, where the problem was formulated. Figure 3.3.3 clarifies the data flows derived from the application of this methodology.

The data required to characterize the unit processes that integrate the system are compiled in the Supporting Information: waste composition (Appendix A), waste management unit processes (Appendix B) and corn production subsystem (Appendix C).

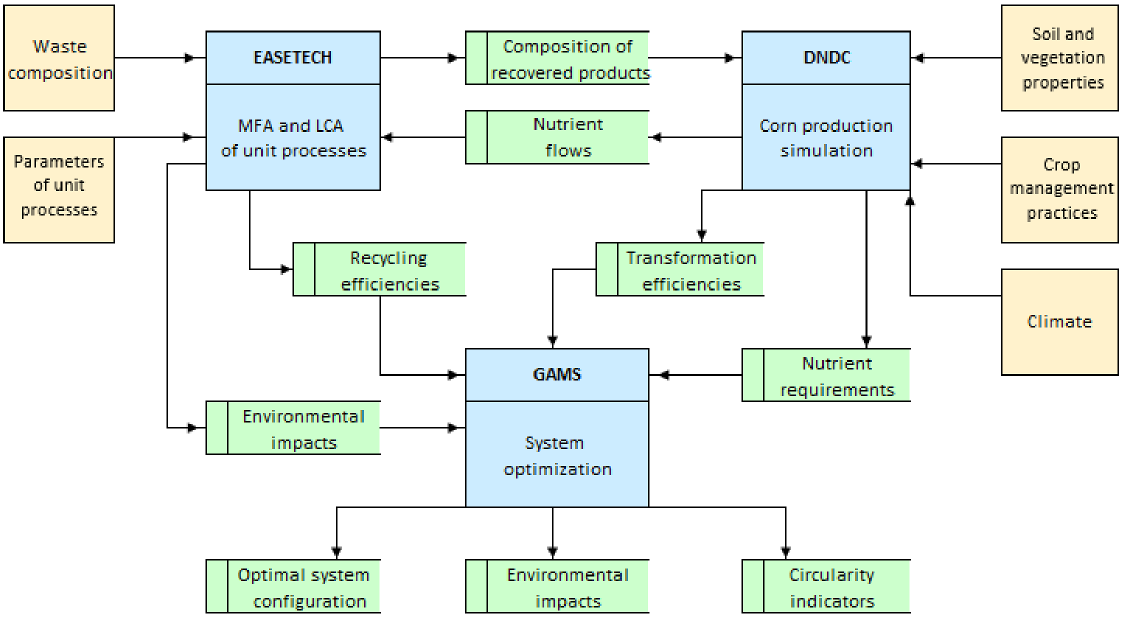

Figure 3.3.3. Data flow diagram

DEFINITION OF THE CIRCULARITY INDICATORS

Figure 3.3.4 illustrates the flows of the component *i* of a given waste stream within a CIWMS. The circularity indicator of component *i* (CI_i) is defined as the amount of component *i* that extends its lifetime by providing a service in the upstream processes with respect to the amount of that component present in the collected waste. Equation 3.3.1 shows how the CI_i is calculated for a set of *n* recycling and preparation for reuse processes and *m* production processes that valorize this component.

$$CI_i = \frac{\sum_{k=1}^{m}\sum_{j=1}^{n} R_{ijk} \cdot \eta_{r_{ij}} \cdot \eta_{p_{ik}}}{W_i} \qquad \text{(Equation 3.3.1)}$$

The variables needed for the calculation of CI_i are these:

- W_i. Amount of component *i* present in the waste stream (kg).
- R_{ijk}. Amount of component *i* that enters the recycling or preparation for reuse process *j*. The subsequently recovered component *i* enters the production process *k* (kg).
- $\eta_{r_{ij}}$. Efficiency of the recycling or preparation for reuse process *j* for component *i* (kg of component *i* recovered per kg of component *i* that enters process *j*).
- $\eta_{p_{ik}}$. Efficiency of the production process *k* at transforming or incorporating the recovered component *i* into a product that will deliver a service in the consumption subsystem (kg of component *i* transformed per kg of component *i* that enters process *k*).

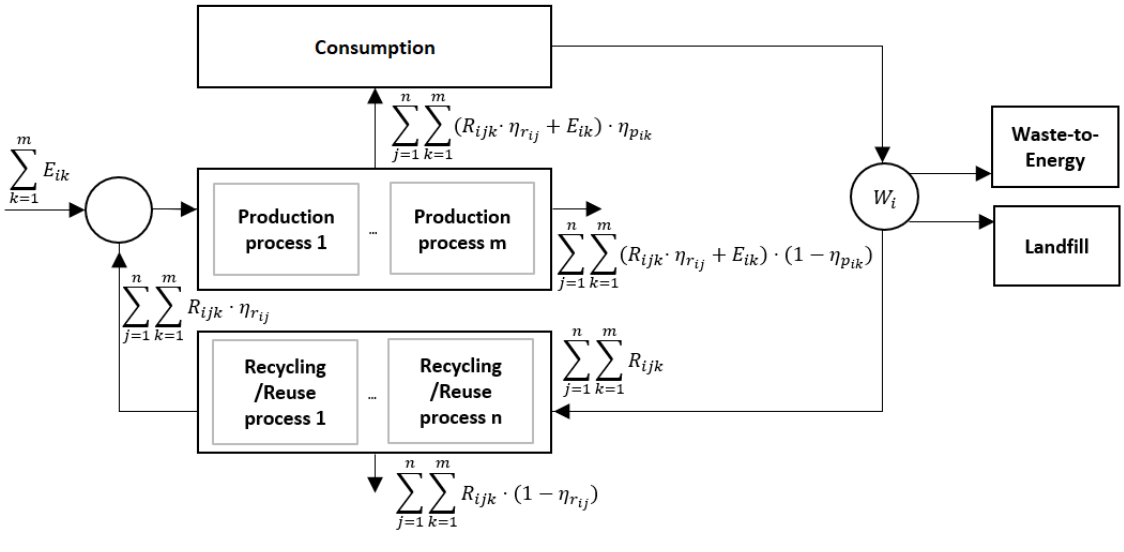

Figure 3.3.4. Simplified CIWMS

CI_i is dimensionless, its value can range between 0 and 1. A value of 1 implies that the total amount of component *i* that was discarded is recovered and reprocessed to enter the consumption subsystem, indicating that there are not any losses of component *i* in the recycling, preparation for reuse and upstream processes. If $CI_i = 0$, component *i* is not recovered at all, but incinerated or landfilled instead.

The proposed indicator indirectly accounts for the quality of the recovered components by quantifying how much of the recovered component is consumed. This indicator does not account *per se* for the degradation of the waste components after successive cycles, but if the selected time horizon of the study is wide enough, a dynamic analysis should show how for a sustained service demand, the external supply of component *i* ($\sum_{k=1}^{m} E_{ik}$) must increase due to the degradation of the recovered component.

Nutrient circularity indicators

The circularity indicators of N and P (CI_N and CI_P) were defined as the amount of nutrient *i* that is recycled, applied to land and taken up by corn with respect to the amount of nutrient *i* present in the collected OW.

The same definition cannot be applied to the C circularity indicator (CI_C), since the C captured by vegetation in the photosynthesis process does not come from the soil but from the atmosphere.

Besides improving the water-holding capacity of soil and its ability to retain cations in a plant available form, contributing to C sequestration and promoting the formation of soil structure,[36,37] the purpose of applying a source of C to land is to feed the soil microorganisms. When these microorganisms decompose the SOC, the decomposed C is partially lost as CO_2, and DOC is produced as an intermediate that can be consumed by the soil microorganisms.[38] These microbes are also responsible for the N fixation, ammonification and nitrification processes that release N compounds that plants can assimilate; they are essential for crop production.

Consequently, a different definition was proposed for CI_C. It was defined as the ratio between the mass of DOC that is recycled, applied to land and consumed by microbes with respect to the amount of C present in the collected waste.

The values of $\eta_{r_{ij}}$ and $\eta_{p_{ik}}$ required for the calculation of the circularity indicators are compiled in Appendix D of the Supporting Information.

PROBLEM FORMULATION

A single-period Mixed Integer Linear Programming problem was formulated for the optimization of the decision variables; i.e., the incoming material flows (waste and recovered products) to the green shaded unit processes in Figure 3.3.1. The problem was optimized according to these objective functions, where x and y represent the continuous and binary variables respectively: the circularity indicators of the studied nutrients, which must be maximized ($CI_C(x,y)$, $CI_N(x,y)$, and $CI_P(x,y)$), and the selected environmental impacts of the system to be minimized ($GW(x,y)$, $MEU(x,y)$ and $FWE(x,y)$).

After verifying the trade-offs between the objective functions, a multi-objective problem was formulated as follows:

$$\min U(x,y) = \\ = \{GW(x,y), MEU(x,y), -CI_N(x,y), -CI_P(x,y)\} \;\; s.t. \begin{cases} h_a(x,y) = 0 & a = 1,2,\dots,v \\ g_b(x,y) \le 0 & b = 1,2,\dots,z \\ x \in \Re^n & \\ y \in \{0,1\}^m & \end{cases}$$

The equations that describe the behavior of the system ($h(x,y) = 0$) are based on the mass balances of the unit processes. The problem is subject to these restrictions ($g(x,y) \le 0$):

- The area fertilized with the recovered products cannot exceed the available area to grow corn in Cantabria (4810 ha).[39]
- The amount of biodegradable waste sent to landfill must be lower than 35% of the domestic waste generated in 1995 (170,168 metric ton),[11] as established by Directive 1999/31/EC.[40]
- Windrow and tunnel composting cannot accept waste streams with the same composition.
- SS-OW and mix-OW cannot be mixed in any composting processes.

The GAMS model comprises a total of 844 equations, 19 inequations, 817 continuous variables and 28 discrete variables. The main input parameters to the models are the source separation rate (SSR), the total area available for corn production and the amount of organic waste generated yearly in Cantabria.

Different waste collection systems for SS-OW and commingled waste were modeled. It was considered that the composition of SS-OW is 98% organic waste and 2% impurities, which is consistent with documented source separation experiences.[41] Two scenarios (neglecting and considering the current legislative framework) were analyzed:

- Pre-Directive scenario. Mix-OW can be recycled. The SSR is 0% and no recycling target is set. The red arrows in Figure 3.3.1 represent the flows of mix-OW that can only be composted in this scenario.
- Post-Directive scenario. Mix-OW cannot be recycled. To comply with the 50% organic waste recycling target established by the Cantabrian waste management plan[11] for 2020, a 50% SSR is set, and an additional restriction is added to the model to ensure that 50% of the collected organic waste is composted or anaerobically digested. The blue arrows in Figure 3.3.1 represent the flows of SS-OW that are specific to this scenario.

The multi-objective optimization problem was solved with the CPLEX solver and the Ɛ-constraint method.[42]

MODELING APPROACH AND ASSUMPTIONS

The boundary that separates the studied CIWMS from the ecosphere (which provides the natural resources consumed by the system and a sink for the generated environmental burdens) and the rest of the technosphere is depicted in Figure 3.3.1.

Although crops are managed by farmers under controlled conditions in the technosphere, they produce natural biotic resources. Hence, the boundary between technosphere and ecosphere is difficult to identify for agricultural soils.[43] One of the strategies recommended by Notarnicola et al.[44] to overcome the limitations of considering agricultural soils as part of the technosphere, is to include the impacts of crop production on soil. In this study the land application of the recovered products and the production of corn were modeled as a UP. Although the system was optimized for 1 year of operation, the selected 100-year time horizon enabled us to account for the loss of soil quality due to soil nutrient depletion caused by the production of consecutive annual crops. The average annual corn production and emission rates in that timeframe were considered.

Corn enters the food production and consumption subsystem, which comprises the upstream processes that transform corn and the other food commodities consumed in Cantabria into OW. It composes the background subsystem of the CIWMS because its configuration does not affect the results of the study;[45] only the flows and the composition of its inputs and outputs (corn and waste) that connect it to other unit processes are calculated.

According to Cobo et al.,[9] the primary function of CIWMSs is to recover waste components so that their service life in the upstream processes can be extended. In this case study the elements recovered from organic waste are used for land fertilization and soil conditioning. Since the studied CIWMS encompasses the entire corn production of the region, the functional unit selected to perform the LCA of the system is defined as the area available to grow corn in Cantabria (4810 ha).[39]

An attributional LCA approach was applied. The electricity generated at incineration, anaerobic digestion and landfill is considered the secondary system function. The direct substitution method was applied by expanding the system boundaries to include the generation of electricity from the Spanish grid mix. A 100% substitution ratio was assumed.

The characterization factors of each emission were calculated with the hierarchical 100-year perspective of the ReCiPe 1.11 method. The assumptions made by the DNDC model about the distribution of nutrients in the environment can be found in Li et al.[46] Following the rationale explained by Cobo et al.,[9,31] only the biogenic C present in animal and vegetable food waste (which can i) leach into the water, ii) be emitted to the atmosphere, or iii) be stored either in the landfill or the soil as a result of the land application of the recovered products, as shown in Appendix C of the Supporting Information) was considered neutral. The CO_2 derived from the decomposition of SOC was also quantified as fossil C.

Regarding the limitations of the model, the environmental impacts related to capital goods were excluded from the analysis. Moreover, this work assumes that all the P is in mineral form and accessible for plants. Studies have shown that most of the P in the products recovered from organic waste is in mineral form, but not all of it.[47-50]

On the contrary, the mineralization of organic N is quantified by the DNDC biogeochemical model. The organic/inorganic N ratio was assumed to be 93/7 for the compost and bio-stabilized material,[50] and 62.96/37.04 for the solid digestate.[51]

The DNDC model assumes a 60% microbial efficiency to calculate the amount of C incorporated into microbial biomass in amended soils, defined as the ratio of C assimilated into microbial biomass to residue C released by decomposition.[46]

RESULTS AND DISCUSSION

The results of the problem optimization determine the system configuration; i.e., the unit processes that the system comprises and their incoming flows of waste and recovered products. The values of the objective functions and the decision variables that optimize each objective function for the two studied scenarios are compiled in Figure 3.3.5. Figure 3.3.5A shows the optimal flows of organic waste and liquid digestate handled by the *j* unit processes. The optimal flows of the recovered products into the k corn production unit processes (Figure 3.3.5B) are shown along with the area fertilized with the recovered products. The contribution of the unit processes to the environmental impacts of the optimal system configurations of each scenario are depicted in Figure 3.3.6.

The flows of organic waste shown in Figure 3.3.5A are lower in the Pre-Directive scenario because part of the organic waste present in the mixed waste ends up in the inorganic waste stream after the trommel separation required for the pretreatment of mixed waste.

There are several system configurations that lead to the maximization of a given circularity indicator, because the unit processes that manage the rejects do not affect the corn production subsystem, and thus they do not contribute to closing the nutrient loops. By analogy, in the Post-Directive scenario where mix-OW cannot be recycled, the selection of any unit process for its management will result in the same circularity indicators. This is the reason the maximization of the circularity indicators in Figure 3.3.5A only shows the unit processes that contribute to recirculate nutrients.

The amount of P present in the mix-OW collected in the Pre-Directive scenario is more than enough to cover the P requirements of the corn produced in Cantabria under the hypothesis of this work. However, the N present in organic waste cannot fertilize all the land available for corn production in any of the studied scenarios. Consequently, strategies S1 and S2 must be combined in the Pre-Directive scenario to maximize CI_C and CI_N. As Figure 3.3.5B shows, more area is fertilized with the recovered products in the Pre-Directive scenario because of the higher amount of organic waste that can be recycled, which makes farmers less dependent on industrial fertilizers (strategy S3). Oppositely, the optimization of all the objective functions are partially based on strategy S3 in the Post-Directive scenario.

The optimization of some objective functions provides duplicate or very similar results (FWE and CI_P on the one hand, CI_C and different circularity indicators in each scenario on the other). To avoid redundant results, FWE and CI_C were not considered in the next part of the study, focused on a multi-objective optimization of the other four objective functions.

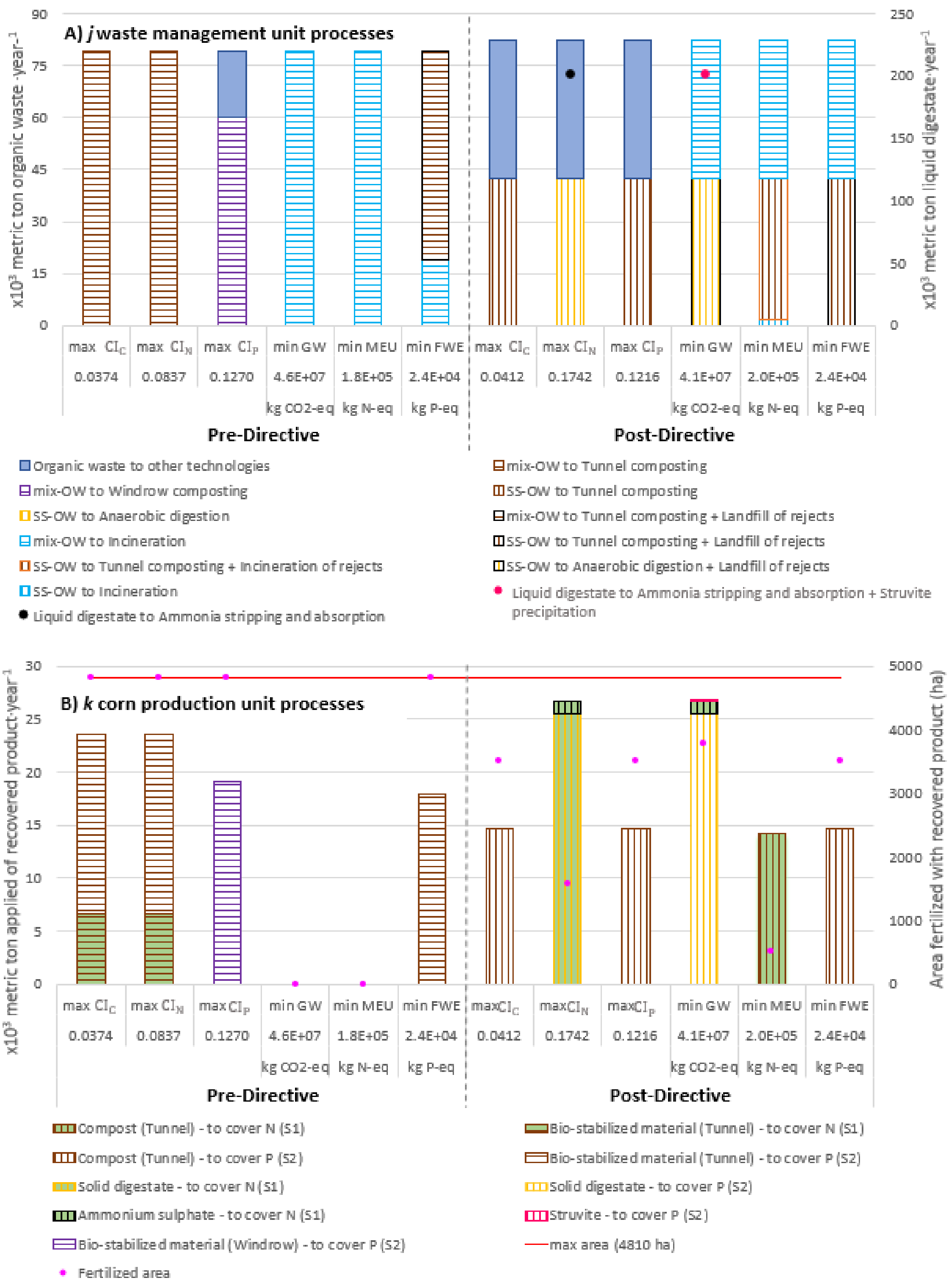

Figure 3.3.5. Values of the objective functions and decision variables for the optimization of the Pre-Directive and Post-Directive scenarios

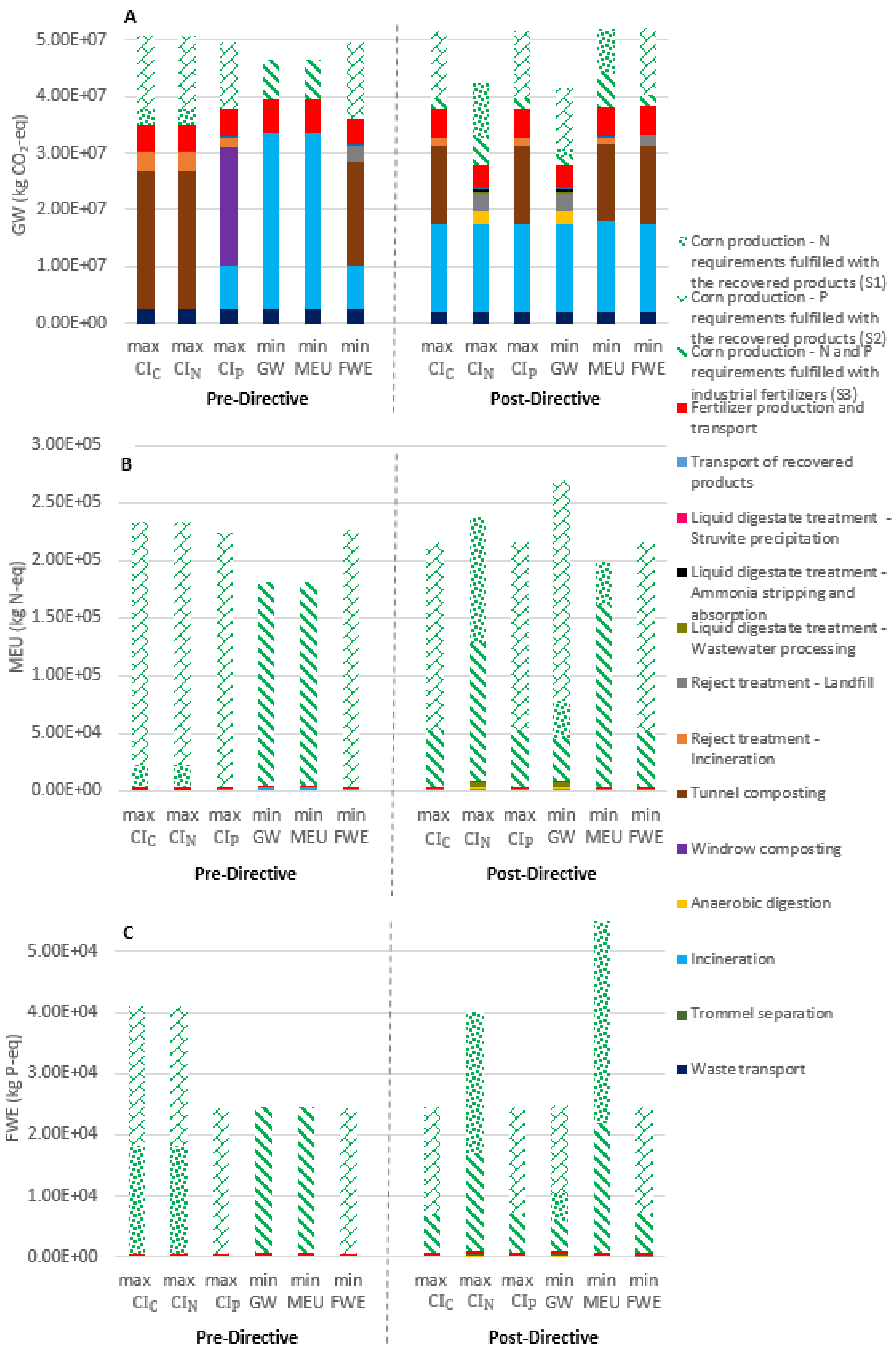

Figure 3.3.6. Contribution of the unit processes to the environmental impacts in the Pre-Directive and Post-Directive scenarios

Figure 3.3.7 shows the Pareto fronts of the two scenarios, where each point is better than the others in at least one of the values of the objective functions. GW and MEU are normalized with respect to the maximum value of the two scenarios.

As the results of the DNDC simulations show, if industrial fertilizers, ammonium sulfate or struvite (inorganic fertilizers) are exclusively applied to soil, the corn Nitrogen Use Efficiency (defined as the fraction of N input harvested as product)[52] decays over time because of the depletion of SOC. The opposite occurs when bio-stabilized material, compost and digestate (organic fertilizers) are applied, due to their C rich composition. However, the mean Nitrogen Use Efficiency obtained for the 100-year time horizon if inorganic fertilizers are applied to land is higher than the Nitrogen Use Efficiency achieved after the soil application of the organic fertilzers, because the share of plant available inorganic N in the latter is low. This implies that more N leaches when the organic fertilizers with a high organic N content are applied to land. These results are supported by previous studies that highlight that the N leaching rate of organic fertilzers is higher than that of inorganic fertilizers.[53,54]

As Figures 3.3.6B and 3.3.6C indicate, the corn production subsystem is the main contributor to the eutrophication impacts. In both scenarios the MEU increase with the CI_N, being the values of these two objective functions higher in the Post-Directive scenario. A similar correlation cannot be established between CI_P and FWE because, unlike N, which tends to leach as nitrate when it is applied to soil, P is strongly sorbed onto soil particles; in fact its major environmental losses can be attributed to erosion.[55]

The Pre-Directive scenario, where the minimum amount of organic waste that must be recycled is not restricted, relies on incineration and the application of industrial fertilizers. Figure 3.3.6A shows that, although the production of industrial fertilizers is very energy intensive,[56] the carbon footprint associated with their land application is lower than that of the organic fertilizers, a fraction of which degrades to CO_2 after their land application. Thus, as Figure 3.3.7A shows, in the Pre-Directive scenario as CI_N increases, the CO_2-eq emissions increase too.

Anaerobic digestion is the unit process that handles organic waste with the lowest carbon footprint. Hence, the minimum carbon footprint achieved at the Post-Directive scenario, the only one where SS-OW can be subjected to anaerobic digestion, is lower than in the Pre-Directive scenario. Moreover, since the N recycling efficiency of anaerobic digestion and the liquid digestate unit processes is higher than that of the other unit processes, the land application of the products derived from anaerobic digestion also maximizes CI_N. Therefore, as shown in Figure 3.3.7B, in the Post-Directive scenario as the CI_N increases, the carbon footprint of the system decreases.

Regarding CI_P, it shows a similar trend to the CI_N in the Pre-Directive scenario, whereas no clear trend can be appreciated in the Post-Directive scenario, where the maximimization of CI_P is based on the application of compost to cover the soil P requirements, and the maximization of CI_N on the application of ammonium sulfate and solid digestate to fulfill the soil N needs, which leads to the accumulation of P in soil. The values of CI_P are lower in the Post-Directive scenario because of the restriction that prevents mix-OW from being recycled.

A sensitivity analysis was performed to ascertain the consequences that a 20% decrease in the values of two key parameters have on the results. The Spanish legislation prioritizes electricity from the biogas produced at landfills and anaerobic digestion facilities over other sources of non-renewable electricity. Notwithstanding, the electricity generated from waste incineration does not have priority access to the grid.[57] The sensitivity analysis considered that 80% of the electricity generated from the incineration of organic waste replaced the electricity from the Spanish grid mix. On the other hand, it is hard to estimate the composition of SS-OW, since pilot experiments for the source separation of organic waste have not been carried out in Cantabria. The sensitivity analysis assumed that the fraction of organic waste in the SS-OW was 78.4%.

The results of the single-objective optimization of each scenario under the conditions of the uncertainty analysis are compiled in Appendix E of the Supporting Information. The main difference in the values of the decision variables after the performance of the sensitivity analysis is that the FWE of incineration exceed those of landfill. Thus, landfill is selected over incineration when the FWE are minimized. As expected, the results of the sensitivity analysis led to slightly higher environmental impacts in both scenarios and lower circularity indicators in the Post-Directive scenario.

Figure 3.3.7 proves that the environmental impacts associated with increasing the circularity of nutrients cannot be overlooked. Whereas in the pre-Directive scenario there is a clear opposite trend between the environmental impacts and the circularity of nutrients, the behavior of the system in the Post-Directive scenario, subject to more restrictions and with more available unit processes, is more complex.

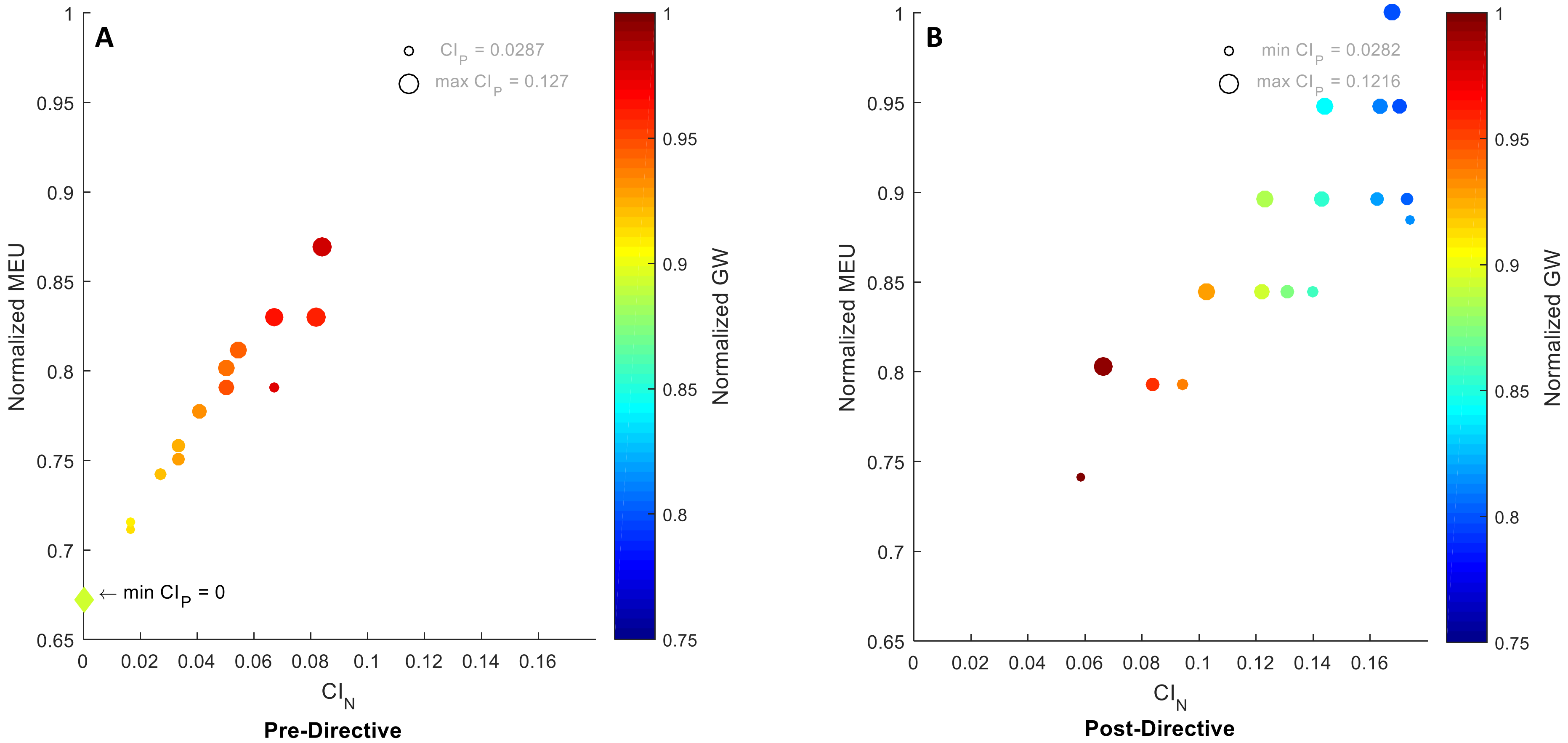

Figure 3.3.7. Pareto points for the Pre-Directive and Post-Directive scenarios

The findings of this study suggest that increasing the SSR of organic waste leads to a reduction in the carbon footprint of the system. Although the results indicate that increasing the circularity of N has detrimental eutrophication impacts, these are highly dependent on the sensitivity of the receiving environment;[58] thus general conclusions cannot be drawn.

Before selecting a system configuration that meets the sustainability concerns and satisfies the interests of all the stakeholders involved in waste management and the purchase of the recovered products, a trade-off between the studied indicators must be identified. Moreover, additional impact categories that quantify the environmental impacts associated with the presence of heavy metals or organic pollutants in the recovered products, such as human toxicity or ecotoxicity, should be included in the analysis. However, the feasibility of any system configuration cannot be demonstrated until an economic analysis is performed."

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CHAPTER 3.4

ECONOMIC ASSESSMENT

"Anyone who believes that exponential growth can go on forever in a finite world is either a madman or an economist."

Kenneth Boulding, English/American economist (1910-1993)

Chapter 3.4 is a literal transcription of this published paper:

Cobo, S.; Levis, J.W.; Dominguez-Ramos, A.; Irabien, A. Economics of enhancing nutrient circularity in an organic waste valorization system. *Environ. Sci. Technol*. **2019**, 53(11), 6123-6132; DOI: 10.1021/acs.est.8b06035.

"Decoupling environmental impacts from economic growth by improving resource efficiency is the goal of the circular economy that the European Union promotes by setting increasingly high recycling targets, which should reach 65% by 2035.[1]

The collection and preparation of more recyclables usually translates into added costs for waste managers.[2,3] To minimize the extra costs assumed by the citizens for recovering more recyclables in both the waste management tax and the price of the recycled products, trade-offs between the interests of the involved stakeholders must be considered.[4-6]

Policy analyses should consider the consumers' willingness to pay more or shift consumption patterns for the sake of sustainability.[7] In some cases, the consumers' reluctance to change their product choices is justified by the reduced performance of the recycled products[8] or the health concerns that could arise from the use of materials that contain undesired substances.[9-10] Hence, social acceptance is important for ensuring that recycled products are competitive in the marketplace, as exemplified by the organic fertilizers recovered from municipal organic waste (OW).

Organic fertilizers have the potential to provide essential nutrients to plants while maintaining the soil organic carbon stock, which is essential to future soil fertility and productivity.[11-13]

One of the challenges of using organic fertilizers created from OW is their potential to release heavy metals and persistent organic pollutants.[14,15] To mitigate the risk of contamination, which can be reduced below the legal limits through source separation,[16] the European Union will require that all the generated OW is either separated and recycled at source or separately collected by 2023.[1]

Nonetheless, other impacts related to the composition of organic fertilizers are unavoidable. Because of the high proportion of organic nitrogen (N) in these products, crops are more efficient at taking up the N in the form of ammonium and nitrates provided by the inorganic fertilizers. Thus, the land application of organic fertilizers leads to increased eutrophication impacts compared to inorganic fertilizers, a rebound effect associated with nutrient recovery.[17]

Moreover, it has been reported that high N recovery rates do not correlate with a decrease in the overall consumption of raw materials,[18] which shows that recycling one waste component does not necessarily entail that the economy is more circular. There could be a host of implications associated with resource looping that remain unexplored. This research aims at filling some of the knowledge gaps in this field by studying the economic consequences of improving the circularity of two key nutrients present in OW, N and phosphorus (P).

The analysis, based on the framework developed by Cobo et al.[19] for the assessment of Circular Integrated Waste Management Systems (CIWMSs), focuses on the Spanish region of Cantabria, with a population of 580.3 thousand and an area of 5,326 km^2.[20]

This study seeks to develop and evaluate waste management strategies with low carbon emissions that comply with the European legislation without compromising their economic interests or those of the farmers that acquire the products recovered from OW. Therefore, the model of a CIWMS that fits the characteristics of the Cantabrian case study was developed and optimized to achieve these goals:

- To determine the configurations of the CIWMS that i) minimize the waste management costs and its carbon footprint, and ii) maximize the circularity of N and P within the system.
- To evaluate how the economic profitability of the system varies as the circularity of N and P changes.

- To estimate the economic margin that enables the recovered organic fertilizers to compete in the market with industrial fertilizers in a mutually beneficial scenario for waste managers and farmers.

SYSTEM DESCRIPTION

Figure 3.4.1 depicts the superstructure comprising the alternative unit processes that may compose the optimal system configurations. The circularity of the system is given by the agricultural application of the products recovered from the OW, which are assumed to be used as fertilizers to grow corn, the crop with the highest cultivated area in Cantabria. This corn is later consumed and partially transformed into OW, closing to a certain extent the loop of nutrients. Thus, the entire corn production of Cantabria was modeled as a subsystem of the CIWMS.

The selected optimal system configurations must meet the 2020 50% OW recycling target set by the Cantabrian waste management plan[21] while abiding by the current legislation that classifies the composted bio-waste that has not been source separated as bio-stabilized material instead of compost.[22] To meet these constraints, at least half of the generated OW must be source separated. Hence, a 50% source separation rate was assumed. Thereby, all the source separated OW (SS-OW) must be recycled, while the only options for the OW separated from the mixed waste stream (mix-OW) are energy valorization and disposal in landfill.

The OW generated in the region is currently separated from the mixed waste stream at the mechanical biological treatment plant. Thus, new strategies for handling the new waste stream, the SS-OW, as well as the mix-OW that was previously composted, are required. The incinerator and the landfill that already exist in the regional waste management plant might be able to process a fraction of the mix-OW, but they were designed to manage primarily the inorganic wastes generated in the region. Hence the construction of a new incinerator and landfill were considered in the study.

The set *s* of waste management unit processes shown in Figure 3.4.1 consists of:

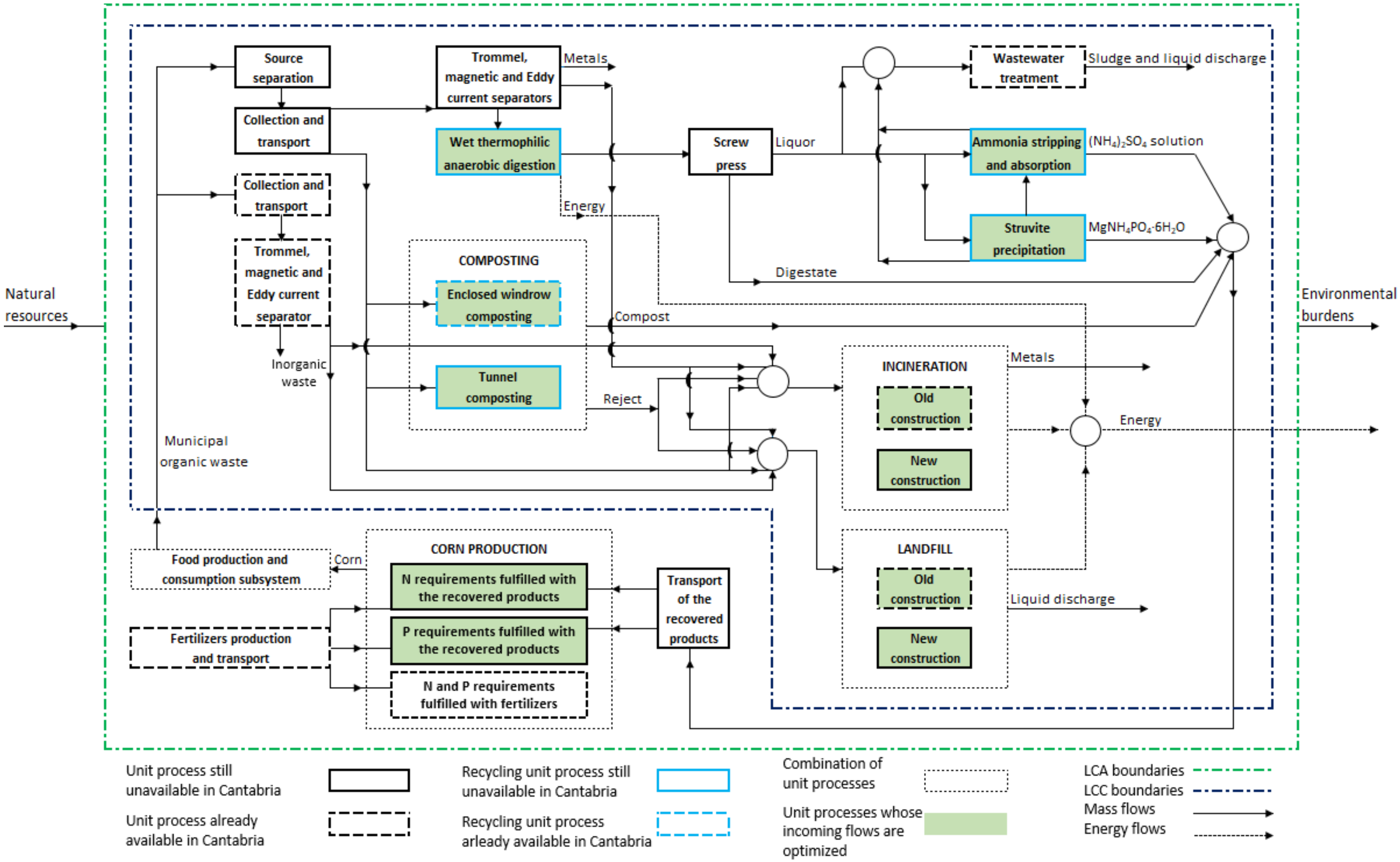

Figure 3.4.1. Diagram showing the system boundaries and the model superstructure

- Unit processes to recycle SS-OW: anaerobic digestion, which includes biogas combustion and power generation, and windrow and tunnel composting. Windrow composting is the only composting option currently available in Cantabria.
- Unit processes to treat mix-OW and the rejects generated in other unit processes: incineration and landfill coupled to the energy valorization of biogas. Two incinerators and two landfills were modeled: the ones that already form part of the Cantabrian municipal solid waste management plant, and the ones that are constructed if the previous ones exceed their capacity.
- Unit processes to manage the liquid digestate: screw press, wastewater treatment, ammonia stripping and absorption, and struvite precipitation.
- Pretreatment unit processes: trommel, and magnetic and Eddy current separators.
- Intermediate unit processes between waste generation and treatment or valorization: source separation, and collection and transport unit processes.

The set *j* of *n* recycling unit processes was defined as a subset of set *s* that is composed of windrow and tunnel composting, anaerobic digestion, struvite precipitation and ammonia stripping and absorption. The products recovered from OW in these unit processes are organic fertilizers (compost and solid digestate), struvite and $(NH_4)_2SO_4$.

Whereas $(NH_4)_2SO_4$ does not contain P, the ratio N/P in the other products is lower than that required by corn. Consequently, the criteria selected to fertilize the soil defines the set *k* of corn production unit processes, which includes transportation:

- Application of the amount of recovered product needed to fulfill the corn P requirements. This leads to a N deficiency, which must be balanced with an industrial fertilizer (NH_4NO_3).
- Application of the amount of the recovered products needed to supply the N required by corn. It results in a P surplus in soil, except if $(NH_4)_2SO_4$ is applied, in which case an industrial fertilizer must be added ($(NH_4)_2HPO_4$).
- Application of the amount of industrial fertilizers (NH_4NO_3 and $(NH_4)_2HPO_4$) needed to cover the N and P requirements of corn.

The heavy metal content of the recovered organic fertilizers is compiled in Table S66 of the Supporting Information. The Spanish Royal Decree 506/2013[23] classifies fertilizing products derived from waste and other organic components into three categories according to their heavy metal content, as shown in Table S67. Because of the high estimated Zn content of the recovered organic fertilizers (251-264 mg·kg^{-1} of dry matter), they must be classified as Type B, or medium quality. Additionally, the new European proposal for the regulation of fertilizing products[24] suggests additional restrictions – which will not be enforced until the proposal is translated into new legislation – for the heavy metal content of organic fertilizers. These limits – except for the Cr VI value, which is unknown – are not exceeded either by the organic fertilizers recovered in the studied system.

METHODOLOGY

A bottom-up mechanistic model of the system was developed through the combination of Material Flow Analysis (MFA), Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) tools. The optimization model was implemented in GAMS 24.7.1 (General Algebraic Modeling System)[25] and optimized with the CPLEX solver.[26] A multi-objective optimization was performed by means of the Ɛ-constraint method.[27] The followed methodological sequence is summarized in Figure 3.4.2.

The MFA and LCA of each waste management unit process was carried out with EASETECH 2.3.6 (Environmental Assessment System for Environmental TECHnologies).[28] The main inputs to EASETECH are the waste composition and the parameters that characterize the unit processes, which are compiled in Appendices A and B of the Supporting Information.

Inorganic waste is assumed to constitute 2% of the SS-OW, which is consistent with real source separation experiences.[29] The share of waste materials that compose the OW and the impurities are compiled from the Cantabrian waste management plan.[21]

The composition of the recovered products provided by EASETECH was introduced in DNDC 9.5 (Denitrification-Decomposition),[30] where the flows of C, N and P resulting from the application of these products as fertilizing agents for corn production were modeled, based on the soil properties and the crop management practices described in Appendix C. The software calculated the amounts and chemical forms in which these nutrients are taken up by corn, stored in soil or dispersed in the environment as gas emissions or dissolved compounds. These results, shown in Appendix C, were transferred to EASETECH to perform the LCA of the corn production unit processes. The LCA results are shown in Appendix B.

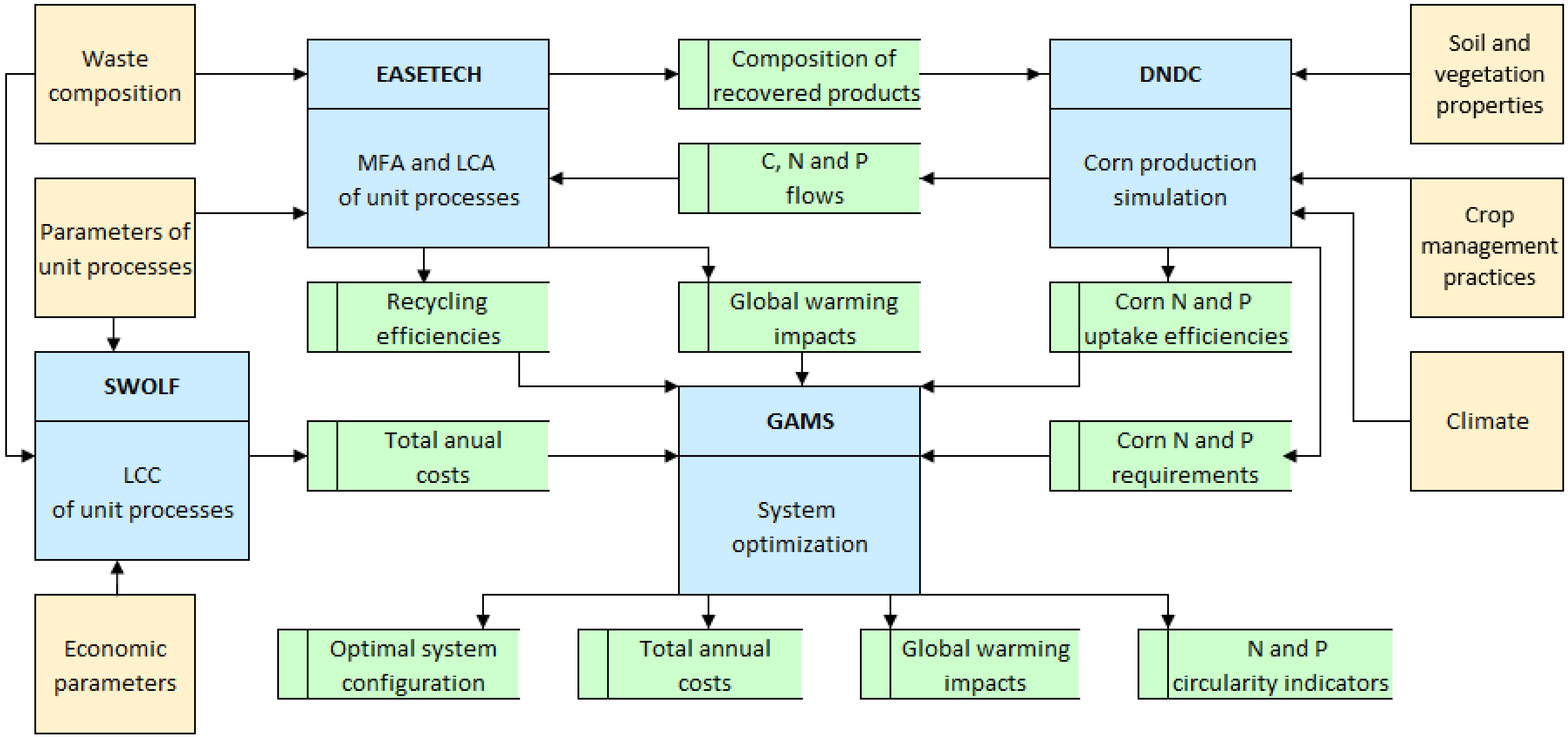

Figure 3.4.2. Data flow diagram

Finally, the LCC sub-models from SWOLF (the Solid Waste Optimization Lifecycle Framework)[31] were adapted to calculate the Total Annual Cost (TAC) of the waste management unit processes, detailed in Appendix F. All the economic data are referred to year 2015.

The struvite precipitation and ammonia stripping and absorption unit processes are not accounted for by either SWOLF or EASETECH. These unit processes were modeled according to the data found in the literature and shown in Appendices B and F.[32,33] Guthrie's modular method was followed to estimate the bare module costs of the equipment required for struvite precipitation,[34] which were later linearized, as shown in Appendix F. The data to calculate the capital investment costs associated with the ammonia stripping and absorption unit process were taken from Errico et al.[35]

LIFE CYCLE MODELING

The applied LCC framework only quantifies financial costs, as opposed to the environmental and social LCC methodologies that calculate the externality costs through the monetization of environmental and social impacts.[36-39]

Previous studies have found that when conducting parallel LCA and financial LCC the appropriate system boundaries for each type of analysis may diverge.[38-40] Although the environmental impacts must be assessed across the entire life cycle of the system under study, the costs associated with different life cycle stages are usually assumed by actors with conflicting or even opposed goals.

In the studied CIWMS, the targets of the stakeholders involved in waste management differ from those of the farmers in charge of corn production. The performed LCC only accounts for the waste management costs, whereas the LCA is carried out from the perspective of the citizens in whose best interest is to minimize the environmental impacts of the entire CIWMS. Consequently, the system boundaries that delimit the unit processes analyzed in the LCA (sum of unit processes in sets *s* and *k*) are wider than those of the LCC (set *s* of unit processes), as shown in Figure 3.4.1.

Following the guidelines provided by Cobo et al.[19] for the analysis of integrated waste management systems against the backdrop of a circular economy, the main goal of the described CIWMS is the recovery of nutrients from OW, rather than waste management. Moreover, the scope of the analysis is broad enough to capture the effects of the land application of the recovered products on the total consumption of fertilizers needed to fulfill the annual nutrient requirements of corn. Hence, the selected functional unit to perform the LCA is the area available in Cantabria for corn production that is fertilized annually via any of the *k* corn production unit processes (4,810 ha).[41]

Nonetheless, this functional unit does not describe the performance of the subsystem within the LCC boundaries. Since the developed stationary model describes one year of operation of the system, the LCC results are referenced to the flow of OW processed annually (83,544 metric ton·year^{-1}).[21] The selection of different functional units for the LCA and LCC analysis does not hinder the interpretation of the results, given that they are expressed on the same annual basis.

An attributional modeling approach was followed to implement the LCA methodology. The direct substitution method was applied to subtract the environmental impacts associated with the secondary system function, power generation, assuming a 1:1 substitution ratio of the electricity generated within the system by the electricity from the Spanish grid mix.

Only the biogenic carbon contained in food waste is quantified as neutral; the carbon emissions derived from the other residual fractions collected with the OW contribute to the global warming impacts of the system, whether they come from a biogenic source or not. The reason is that their complete life cycle is not modeled in this study, and thus it is not correct to subtract from the life cycle inventory the CO_2 that was absorbed by biomass in a photosynthesis process outside the system boundaries.

Regarding the timeframe of the analysis, a 100-year time horizon was considered, consistently with the hierarchist perspective used to characterize the emissions of the system by ReCiPe 1.11.[42] This implies that the long-term emissions and the decommissioning costs should be accounted for.

Due to the lack of reliable data, only the landfill closure costs, incurred during the 30-year after-care period established by Directive 1999/31/EC,[43] were quantified. These costs are related to the activities conducted after the cells are filled, as well as those related to site monitoring and maintenance after closure. They are amortized over the 20-year lifetime of the landfill so that they are reflected in the cost of waste disposal. The long-term emissions associated with the landfill unit process are due to the degradation of organic matter over time under anaerobic conditions.

On the other hand, the first year is not representative of the behavior of the corn production unit processes in terms of nutrient requirements and field emissions;[17] hence, the average N and P requirements and emissions associated with the consecutive crops cultivated throughout the 100-year period were taken as model parameters.

One of the limitations of the work is that the corn P requirements were calculated considering that the P present in OW is in a mineral readily available form for plants, which is not necessarily true for 100% of the P.[44-47]

Another limitation is that the economic and the environmental benefits associated with the recovery of the metals that are sorted out in the pretreatment and incineration unit processes are excluded from the analysis, under the assumption that they are negligible.

Finally, although the infrastructure costs are quantified, the related environmental impacts are omitted, which is supported by a study that concluded that the contribution of infrastructure to the global warming impacts in waste management systems are generally negligible.[48]

ECONOMIC MODELING

The TAC of each unit process is calculated as the sum of the operating costs and the capital costs minus the revenues derived from the sale of electricity in the Spanish market (51.67 €·MWh^{-1})[49] and the recovered products, if applicable. The capital costs were annualized assuming a 15-year amortization period and a 7% interest rate, which was selected based on the interest rates applied by the Spanish credit institutions in 2015.[50]

The market prices of compost in the European Union typically range between 0-14 €·metric ton^{-1},[51] whereas in some instances farmers are paid to accept the digestate.[52] Indeed, it is a common practice for waste managers to cover the expenses related to transport and spreading of the organic fertilizers.[51,53] The hypothesis of this study is that compost and digestate are given away to farmers at a price of 0 €·metric ton^{-1}.

The struvite price was calculated based on its P content. It was assumed that the price of P is the same as the price of $(NH_4)_2HPO_4$ (413.42 €·metric ton^{-1})[54] expressed per kg of P. The price of the recovered $(NH_4)_2SO_4$ corresponds to its market value (128.99 €·metric ton^{-1}).[55]

Furthermore, the Spanish legislation considers providing financial support to the facilities that generate electricity from renewable energy sources.[56] These subsidies were included in the model as an operating income to the landfill and anaerobic digestion unit processes.

The TAC of the landfill also includes the tax that waste managers are charged by the regional government (2 €·metric ton^{-1} of landfilled waste),[57] and the above-mentioned closure costs.

The capital costs of the unit processes already available in the Cantabrian waste management facilities (shown in Figure 3.4.1 with a discontinuous line) are assumed to be already amortized, and hence they are not considered. Only the costs associated with the construction of new cells within the previously excavated landfill are accounted for as capital costs of the old landfill.

PROBLEM FORMULATION

The decision variables in the optimization problem consist of the mass flows of waste and recovered products that enter the green shaded unit processes in Figure 3.4.1; the remaining variables are calculated as a function of these decision variables.

The objective functions according to which the decision variables are optimized are the TAC, the global warming impacts (GW) and the circularity indicators of N and P (CI_N and CI_P). The TAC and GW are minimized, whereas CI_N and CI_P are maximized.

Equations 3.4.1 – 3.4.3 indicate how the objective functions were calculated for the set *s* of *t* waste management unit processes, the set *k* of *m* corn production unit processes, and the set *j* of *n* recycling unit processes. The objective functions depend on the values of the continuous (x) and the binary (y) variables, which indicate the unit processes that are selected in the optimized superstructure.

The TAC of the system is calculated as the sum of the TAC of all the unit processes in set *s* minus the income perceived as the tax paid by the municipalities in exchange for the waste management services, TAX_{WM} (85.28 €·metric ton^{-1}).[58] The GW are calculated as the sum of the GW of the unit processes in sets *s* and *k*.

$$TAC(x,y) = \sum_{s=1}^{t} TAC_s(x,y) - TAX_{WM}(x) \quad \text{(Equation 3.4.1)}$$

$$GW(x,y) = \sum_{s=1}^{t} GW_s(x,y) + \sum_{k=1}^{m} GW_k(x,y) \quad \text{(Equation 3.4.2)}$$

$$CI_i(x,y) = \frac{\sum_{k=1}^{m} \sum_{j=1}^{n} R_{ijk}(x,y) \cdot \eta_{r_{ij}} \cdot \eta_{p_{ik}}}{W_i} \quad \text{(Equation 3.4.3)}$$

The definition of the circularity indicator of component *i* (CI_i) described in Equation 3.4.3 was first proposed by Cobo et al.[17] to quantify how efficient a CIWMS is at providing a service derived from the recovery and valorization of its individual waste components. In this study, CI_i is defined as the amount of nutrient *i* that is recycled, applied to land and taken up by corn with respect to the amount of that nutrient in the collected OW. These are the variables and parameters involved in the calculation of this dimensionless CI_i:

- W_i. Amount of nutrient *i* present in the collected waste (kg).
- R_{ijk}. Amount of nutrient *i* that enters the recycling unit process *j*. The subsequently recovered nutrient *i* enters the corn production unit process *k* (kg).
- $\eta_{r_{ij}}$. Recycling efficiency of the recycling unit process *j* for nutrient *i* (kg of nutrient *i* recovered per kg of nutrient *i* that enters unit process *j*).
- $\eta_{p_{ik}}$. Efficiency of the corn production unit process *k* at taking up the recovered nutrient *i* (kg of nutrient *i* taken up per kg of nutrient *i* entering unit processes *k*).

The values of $\eta_{r_{ij}}$ and $\eta_{p_{ik}}$, calculated from the results provided by EASETECH and DNDC, are compiled in Appendix D.

Additionally, the problem is subject to these restrictions:

- As Directive 1999/31/EC[43] dictates, the amount of biodegradable municipal waste going to landfill must be lower than 35% of the total weight of biodegradable municipal waste produced in 1995 (59,559 metric ton·year^{-1}).[21]
- Windrow and tunnel composting cannot form part of the system simultaneously.
- The area fertilized with the recovered products cannot be larger than the area destined for corn production in Cantabria (4,810 ha).[41]

- A new incinerator or landfill is constructed only if the capacity of the existing one is exceeded.
- Capacity restrictions for certain unit processes.

Following the principles of the economies of scale, the TAC of the unit processes for the treatment and valorization of solid waste decreases with their capacity;[59-63] the TAC is usually modeled as a function of the incoming annual waste flows by means of exponential equations.[64] The introduction of nonlinear equations into the model may lead to nonconvexities, which do not guarantee a unique local optimal solution.[65] To ensure that the solution found by the algorithm is a global optimum, several strategies were considered:

- Breaking down the exponential curves into linear approximations.
- Modeling a set of unit processes with pre-defined capacities, as Hu et al.[5] did.
- Setting minimal capacity restrictions. The effect of the economy of scale on the TAC is assumed to be negligible for the unit processes with capacities larger than the fixed minimum.

The latter was deemed the simplest, easiest to adapt to the SWOLF framework, and least computationally intensive approach. These capacity restrictions are congruent from the decision-making viewpoint, since waste managers are not expected to make large investments in facilities capable of processing just a small fraction of the total waste.

Therefore, the TAC of the unit processes is assumed to vary linearly depending on their size; i.e., the TAC expressed as €·metric ton^{-1} of managed waste remains constant. The minimal capacity restrictions of the unit processes that handle solid waste, which were assigned solely to the unit processes that are still not implemented in Cantabria, were selected based on the exponential curves shown in Figures S11-S19 and extrapolated from the data compiled in the literature.[62,64]

Following the same rationale, minimal capacity restrictions were set for struvite precipitation and ammonia stripping and absorption. These technologies can only be implemented if the flows of liquid digestate treated surpass half of the minimal flow of liquid digestate generated in the anaerobic digestion unit process (29,537.50 m^3·year^{-1}).

Regarding the upper bounds on the capacities of the solid waste management unit processes, constraints for the maximal size of the new facilities were not considered. The existing windrow composting unit in Cantabria currently handles all the OW generated in the region; thus, it is not necessary to fix its maximal capacity restriction. To estimate the maximal annual flows of OW that the current incinerator and landfill can accept, the flows of inorganic municipal waste and other non-domestic wastes that these facilities manage were subtracted from their total capacities.[66,67]

If the flows of OW that must be sent to incineration to achieve an optimal solution exceed the maximal capacity of the existing unit, a new incinerator is constructed. Because of the substantial capital investment that the construction of a new incinerator entails, a minimal incineration capacity higher than that of the existing one is set. On the other hand, waste managers are already considering the construction of a new landfill, since the current landfill is at the end of its lifetime. Thus, a landfill with a capacity of 85,000 metric ton·year^{-1} is assumed to be built if the capacity of the old landfill is exceeded. The capacity restrictions of the solid waste management unit processes are listed in Table S50.

The model, which is composed of a total of 1,180 equations, 1,078 continuous variables and 72 binary variables, was formulated as a single-period mixed integer linear programming problem. It was first posed as a single objective optimization problem for each of the objective functions described above. Then, a multi-objective optimization was performed considering TAC, CI_N and CI_P as objective functions. It was formulated as follows:

$$\min U(x,y) = \{TAC(x,y), -CI_N(x,y), -CI_P(x,y)\} \;\; s.t. \begin{cases} h_a(x,y) = 0 & a = 1,2,\dots,v \\ g_b(x,y) \leq 0 & b = 1,2,\dots,z \\ x \in \Re^n & \\ y \in \{0,1\}^m & \end{cases}$$

RESULTS AND DISCUSSION

Figure 3.4.3 shows the unit processes and flows of OW that achieve the optimal values of the objective functions compiled in Table 3.4.1.

Table 3.4.1. Values of the objective functions for the single objective optimizations

Objective Function	**min TAC**	**max CI_N**	**max CI_P**	**min GW**
TAC (10^6€·yr^{-1})	**-1.96**	1.57-5.00	-1.05-2.00	2.27
CI_N	0.056-0.064	**0.174**	0.066	0.072
CI_P	0.027-0.121	0.033	**0.122**	0.100
GW (10^6kg CO_2-eq·yr^{-1})	54.89-55.00	47.34-47.37	52.31-55.25	**42.66**

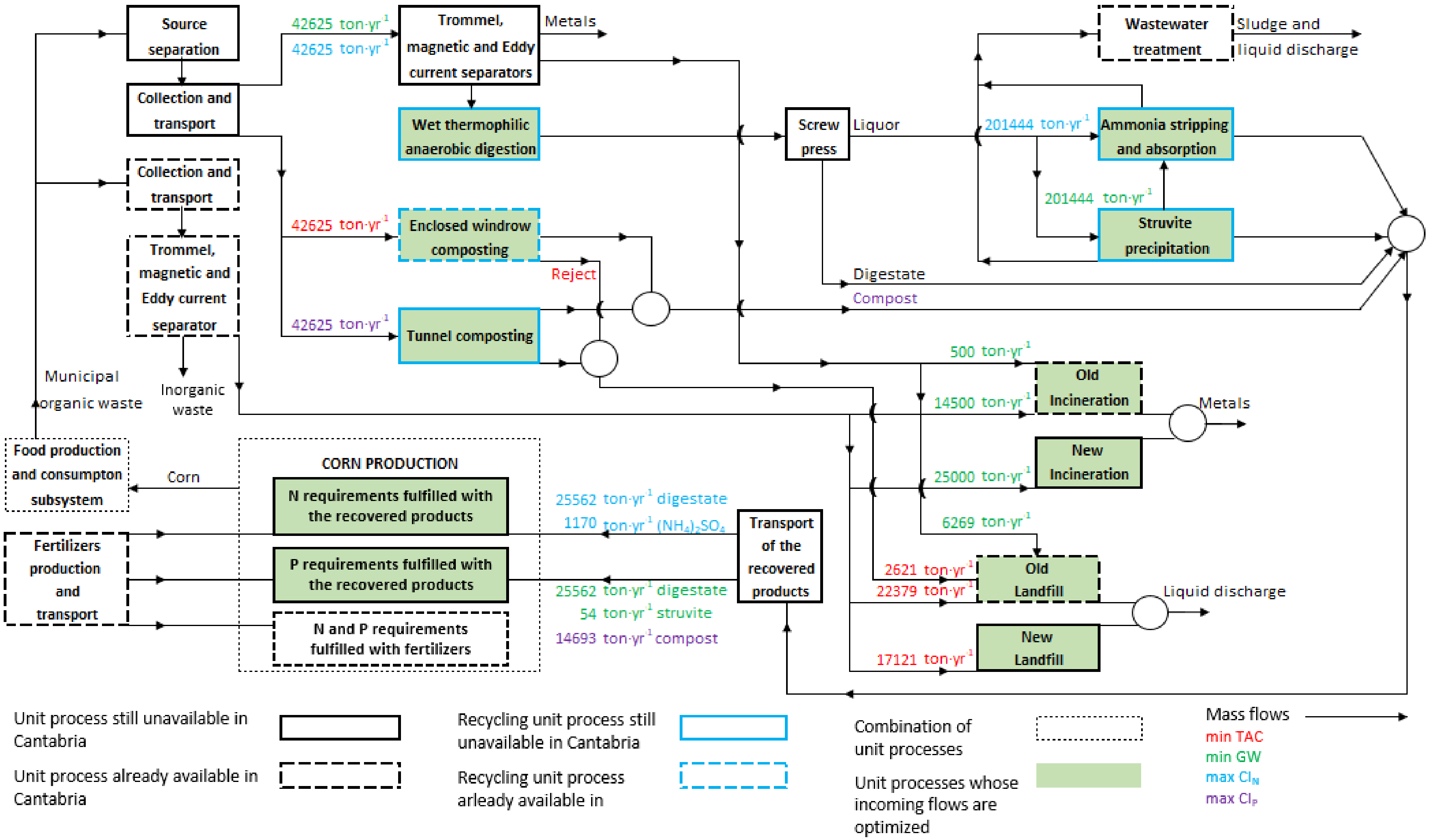

Figure 3.4.3. Values of the decision variables that have an influence on the objective function for each single objective optimization (metric ton·year^{-1})

Only one system configuration minimizes the GW, because all the unit processes contribute to it. However, there are several system configurations that lead to the optimization of the TAC, CI_N and CI_P. The reasons are that the corn production unit processes are located outside the LCC boundaries and they are therefore not needed to calculate the TAC. On the other hand, the selection of the unit processes that handle mix-OW, which cannot be recycled, does not affect the values of the circularity indicators. Consequently, Figure 3.4.3 only displays the values of the decision variables used to calculate the objective functions. Furthermore, the ranges of values in the cells of Table 3.4.1 that do not show a single value, correspond to the ranges of values that the objective functions can take for the different system configurations that optimize a given objective function.

The negative TAC values indicate that the cost to operate the system is less than the revenue received from the TAX_{WM}, the subsidies and the sale of electricity and the recovered products, whereas positive values indicate that the revenues are not large enough to cover all of the costs. The contribution of the unit processes and the TAX_{WM} to the TAC that results from the minimization of the TAC and GW is depicted in Figure S24. It shows that the negative values are due to the TAX_{WM}.

The TAC is minimized primarily using the unit processes that are already available in the waste management system, namely windrow composting and landfill, whereas the new infrastructure required to optimize the other objective functions significantly increase the TAC. As Figure 3.4.3 shows, tunnel composting is used to maximize CI_P, whereas more costly anaerobic digestion with nutrient recovery from the liquid digestate is used to maximize CI_N.

Since no single configuration simultaneously optimizes each objective function, trade-offs must be made. Figure 3.4.4 shows the Pareto points obtained from the three-objective optimization. Each Pareto point represents a solution that cannot be improved in at least one objective without decreasing its performance in at least one other objective.

The multi-objective optimization was peformed for three scenarios: the baseline with the current TAX_{WM}, and scenarios with a 10% and 20% higher TAX_{WM} (93.81 and 102.34 €·metric ton^{-1} respectively). These scenarios account for the fact that policy-makers are likely to raise the TAX_{WM} if more expensive waste management unit processes are implemented.

Figure 3.4.4 shows how increasing the TAX_{WM} improves the circularity of nutrients that is achievable at each level of TAC. With the current TAX_{WM}, CI_N values above 0.13 lead to system configurations that do not generate net profits for the waste managers, while a 10% increase in the TAX_{WM} leads to net profits in every case except the three system configurations with the highest CI_N represented in the Pareto front. A 20% increase in the TAX_{WM} can provide similar net profits to those obtained with the current TAX_{WM} for the most profitable system configuration while increasing CI_N and CI_P by 140 and 270%, respectively.

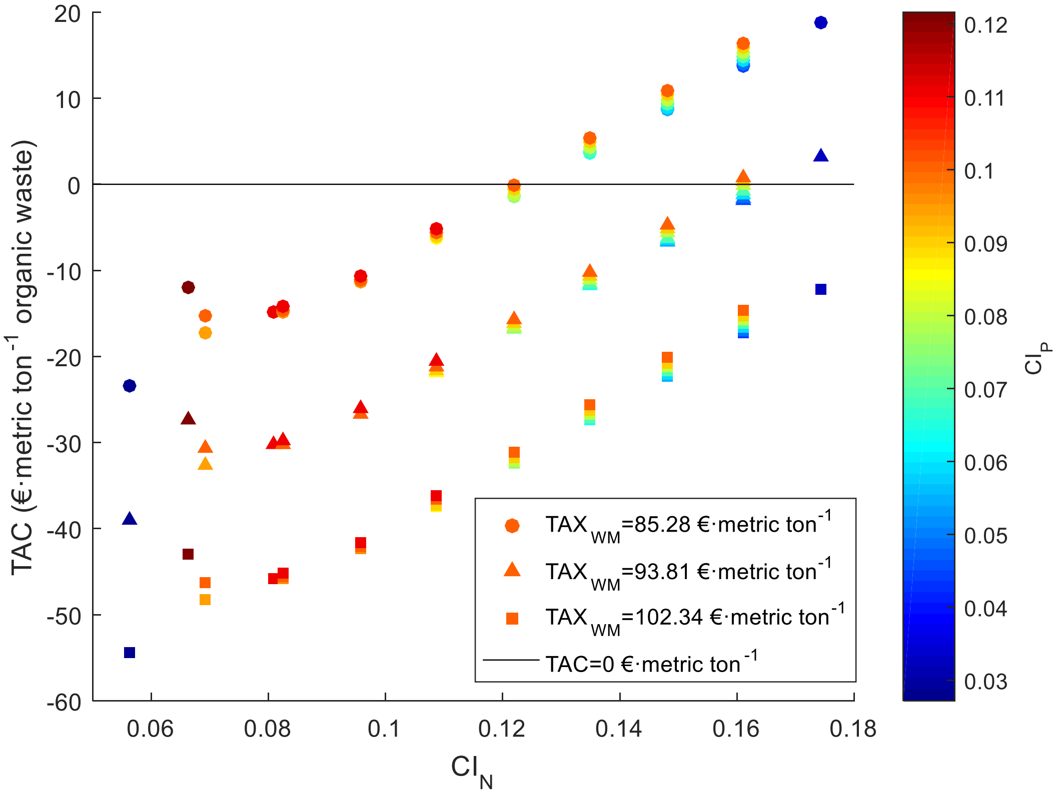

Figure 3.4.4. Pareto points that show the trade-offs among TAC, CI_N and CI_P at the three levels of TAX_{WM}

Increasing CI_N increases the TAC, because it involves a noteworthy investment in new technologies. On the contrary, high CI_P values can be achieved at low TACs. In fact, with the current TAX_{WM}, net profits are generated even when all the SS-OW is composted in a tunnel reactor as long as all the mix-OW is sent to landfill, the least expensive of the modeled unit processes.

System configurations that improve CI_N do not necessarily increase CI_P. The strategy that maximizes CI_N anaerobically digests the SS-OW, recovers $(NH_4)_2SO_4$ from the liquid digestate, and then applies to the soil the amounts of these products needed to cover the corn N requirements. This results in the accumulation of P in soil and the loss of P in the liquid digestate, to the detriment of CI_P.

Under a cooperative approach, win-win solutions in the areas of the decision-making space where all the stakeholders can share the costs and benefits are sought. Therefore, it was investigated how the increased TAC can be shared between the tax payers, the waste managers and the farmers that acquire the recovered products.

Figure 3.4.5 shows for all the Pareto points in each scenario depicted in Figure 3.4.4 the minimum price that farmers would hypothetically have to pay to fertilize 1 hectare with the recovered products and the complementary industrial fertilizers, in order to create net profits for the waste managers. The system configurations that result in a fertilization price that exceeds the estimated price of industrial fertilizers (75.74 €·ha^{-1}), are excluded from the mutually beneficial decision-making space, since farmers would purchase the industrial fertilizers. The minimum price was calculated under the hypothesis that if the TAC is negative, the price of the organic fertilizers is 0 €·metric ton^{-1} (as originally assumed); otherwise, the price is set at the value necessary to achieve a TAC of 0 €·metric ton^{-1}. The equations needed to calculate the minimum price of the fertilizing products are described in Appendix F.

The system configurations that allow farmers to reduce total fertilization costs (the costs of organic plus supplementary industrial fertilizers) compared to only relying only on industrial fertilizers, are those where waste managers earn net profits without charging farmers for organic fertilizers. This happens for CI_N values below 0.13 in the scenario with the lowest TAX_{WM}, whereas in the scenarios with the two highest TAX_{WM}, waste managers can earn a profit while farmers benefit from the competitive prices of the fertilizers for CI_N values up to 0.16.

The difference between the price of the industrial fertilizers and the price paid by the farmers for the recovered products and the complementary fertilizers leaves a margin for farmers to cover the transport and spreading costs of the organic fertilizers, which are higher than those related to the industrial fertilizers, because the amounts of organic fertilizers needed to fulfill the function of the industrial fertilizers exceed the mass of the latter.

Nonetheless, if the distance between the waste management plant and the field where the products are applied is long enough to surpass this margin, farmers would not be willing to pay for the transport and spreading costs. On the contrary, setting low prices for compost and digestate would be realistic if the crops were grown near the waste management plant.

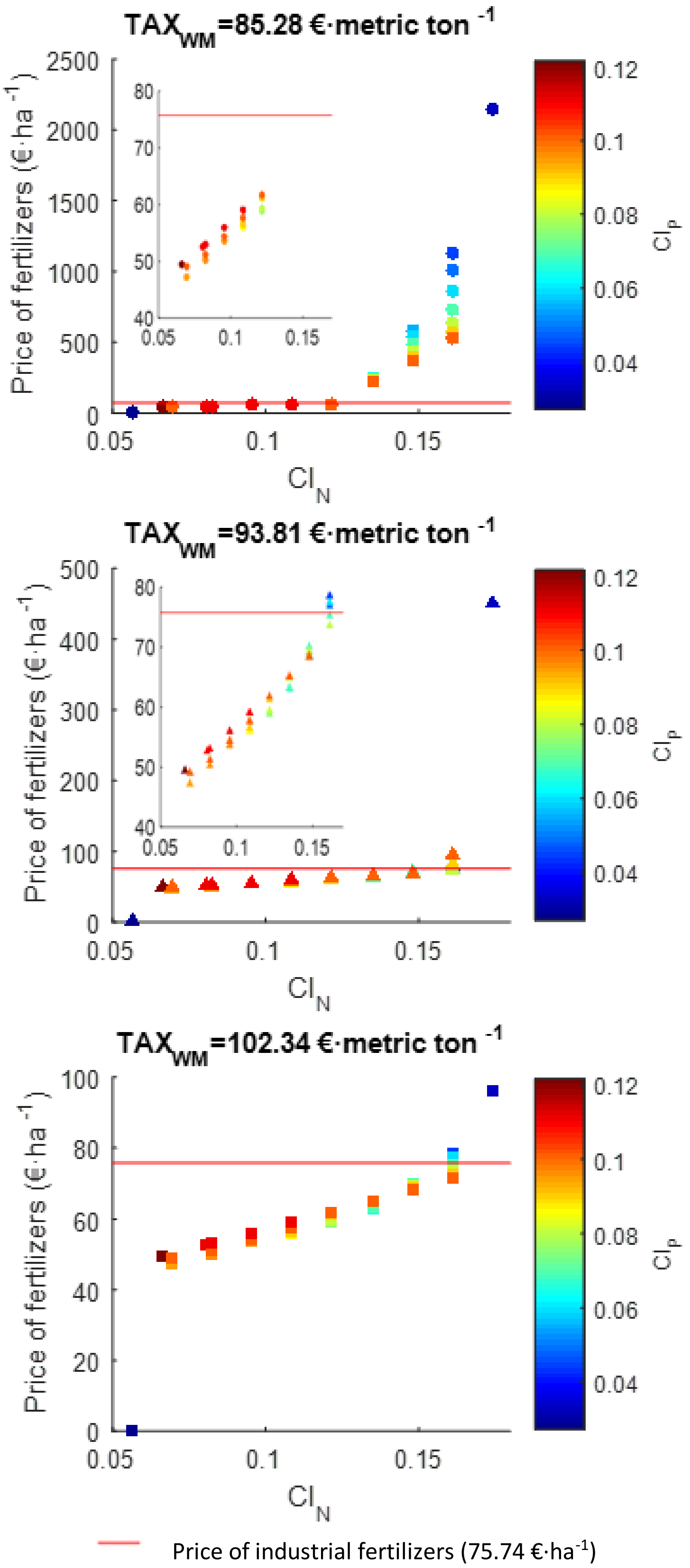

Figure 3.4.5. Minimum price that farmers must pay for fertilizers so that waste managers can generate net profits in the three scenarios

POLICY IMPLICATIONS

The promotion of nutrient circularity is compatible with the economic profit of the waste managers and the sale of the recovered products at competitive prices with industrial fertilizers. However, without economic support in the form of subsidies or an increase in the TAX_{WM}, it would be accomplished at the expense of a reduction in the benefits of the waste managers. The results demonstrate that a relatively modest increase in the TAX_{WM} could lead to significant improvements in the circularity of nutrients. Thus, legislators must incentivize the cooperation between stakeholders to simultaneously achieve the minimal reduction in the economic profit of the waste managers and the minimal increase in the consumer prices, with the minimal rise in the taxes paid by the citizens with respect to the current less circular system.

The results presented in this study challenge the widely assumed premise – on which recent European policy is based – that the improvement in resource efficiency derived from the implementation of a circular economy will decouple environmental impacts from economic growth. These results suggest that the strategies seeking to reduce resource consumption should take precedence over maximizing resource circularity.

Regarding the influence of the uncertainty associated with the input parameters of the model – like waste composition –on the results, it should be addressed in future research. The developed framework is not aimed at determining the precise values of the decision variables and objective functions, but at identifying trends and trade-offs between them, and it can be concluded that it fulfills this objective.

Therefore, although the results are subject to the specific system, data, and assumptions of the case study and cannot be generalized to other regions, the feasibility of the proposed framework – which can be applied to other waste fractions and regions – has been proven. Policy-makers should promote the use of decision support tools based on systems thinking to determine the optimal configuration of waste management systems.“

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CHAPTER 4

CONCLUSIONS

"Scientists who say what should be done as if their recommendations sprang from an objective calculation are abusing public trust."

David Keith, Canadian physicist (born 1963)

A methodological framework aiming at the sustainable integration of waste and resource management was presented and tested in this dissertation. The originality of the research resides in the developed methodological approach and its contribution to unraveling some of the misconceptions often associated with the circular economy.

Scope of the developed decision support tools

The proposed framework is based on the expansion of the boundaries of integrated waste management systems to include the upstream and midstream subsystems where raw materials are extracted, processed and subsequently transformed into waste. The advantage of these expanded systems, identified throughout the thesis as Circular Integrated Waste Management Systems (CIWMSs), is that they enable an in-depth analysis of the consequences of the recirculation of the recovered waste components into the upstream subsystems, facilitating the decision-making processes.

The developed framework relies on the integration of material flow analysis, life cycle assessment (LCA), life cycle costing and multi-objective optimization methodologies to ensure that the optimal design of a CIWMS complies with different sustainability criteria. Moreover, the application of this approach to the integrated management of nutrients and municipal organic waste required the use of a biogeochemical model to determine the distribution and chemical speciation of the nutrients contained in the products used for soil amendment. The combination of these methodologies, specially the integration of agroecosystems modeling within a decision support system, constitutes a novel contribution to the field of process systems engineering. Furthermore, the developed methodological approach can be used to identify the decision-making space where the costs and benefits associated with the implementation of systems that promote resource circularity are fairly distributed among the involved stakeholders.

Regarding the circularity indicator proposed in this dissertation, it allows the quantification and comparison of the circularity of each waste component within a CIWMS. Therefore, these new decision support tools are suitable to explore the sustainability implications of improving resource circularity.

The consequences of improving nutrient circularity in the studied system

Some conclusions – true under the hypotheses of the thesis – about the effects of enhancing nutrient circularity on different aspects of sustainability were drawn from the application of the developed methodological framework to the case study.

- *Resource consumption.*

  It was found that improving nitrogen (N) circularity is related to an increased consumption of non-renewable raw materials in the studied system.

- *Environmental impacts.*

  Enhancing N circularity can lead to a higher level of marine eutrophication with respect to a system that relies to a greater extent on industrial fertilizers. This happens because crops absorb the inorganic N present in industrial fertilizers more efficiently than from the products recovered from organic waste, which have a significant proportion of organic N.

  Contrarily, since the same unit processes are involved in the promotion of N circularity and the decrease in global warming impacts, improving N circularity reduces the carbon footprint of the studied system.

- *Economic profits.*

  It was demonstrated that increasing the economic profits of the waste managers is incompatible with enhancing N circularity in the studied system, because of the remarkable investment in new technologies that it requires.

These results challenge the widely assumed premise – on which recent European policy is based – that the improvement in resource efficiency derived from the implementation of a circular economy will lead to sustained economic growth and an overall reduction in resource consumption and environmental impacts. In the case study, the improvements in some of the studied sustainability dimensions were made at the expense of the deterioration of other aspects, which makes it necessary to find a compromise between them. Thus, the sustainability implications of the recovery and reintroduction into the production cycles of any waste component must be analyzed on a case-by-case basis. However, some general recommendations could be made based on the results of the thesis.

EXTRAPOLATION OF THE RESULTS TO POLICY AND DECISION-MAKING

The optimal system configurations for the management of the municipal organic waste generated in Cantabria are based on the source separation of organic waste, which has proven to reduce the carbon footprint of the studied system as long as it is implemented parallel to the adequate technologies. Nonetheless, given that at the time of writing there are not any plans to invest in the source separation of municipal organic waste, it is highly unlikely that a new regional source separation and collection system will be set up by 2023, as the European legislation establishes.

Along with the technical feasibility of the recycling processes, economic profitability is one of the main limiting factors that will determine the maximum degree of circularity that a given waste component can achieve. To minimize the total costs associated with resource circularity and prevent citizens from assuming these costs in the form of taxes, new policies should incentivize the purchase of the products recovered from waste and the cooperation between all the parties involved in resource recirculation.

Balancing environmental impacts and resource circularity is crucial for the effective implementation of a circular economy. Hence, legislators should promote the use of decision support tools based on systems thinking, like the one presented in this thesis, for the design and analysis of waste and resource management systems. Given the unexpected consequences that could arise from the adoption of a circular economy and offset its benefits, other strategies that seek a reduction in resource consumption (e.g., waste prevention) should take precedence over maximizing resource circularity.

Ultimately, the circular economy is just a concept; politicians should make sure that it is not used as a greenwashing technique. To that end, standardizing a metric to verify the extent of resource circularity, such as the circularity indicator proposed in this dissertation, could be useful.

LIMITATIONS OF THE WORK AND SUGGESTIONS FOR FUTURE RESEARCH

Further improvements in the proposed framework – which could help us better understand how circular systems contribute to sustainability – should focus on two complementary lines of work: the expansion of the scope of the developed model and the integration of new methodological tools within the decision support system.

Model expansion

The integration of other sources of organic matter (such as sewage sludge, manure or forest residues) and other upstream processes (the cultivation of alternative crops), as well as other waste fractions within the developed model would provide a more realistic description of the case study. Moreover, additional trade-offs between the studied impact categories and other environmental impacts associated with the presence of pollutants in the products recovered from organic waste, such as human toxicity or ecotoxicity, should be explored.

Assessing the management of organic waste and nutrients under the perspective of the water-food-energy nexus could help us analyze the synergies between these basic human needs and delineate new strategies to attain the Sustainable Development Goals. On the other hand, expanding the developed framework to account for the consequences of waste prevention activities is another attractive line of research that could lead to new policies oriented to the reduction of food waste, and by extension, to more efficient food systems.

Since energy is an essential input to any process, coupling energy systems modeling to the developed framework would better describe the consequences of alternative system configurations. For instance, in a decarbonized energy scenario, an energy-intensive recycling process could become environmentally friendly, whereas the energy valorization of waste might displace cleaner sources of energy.

The proposed framework should be applied to other waste fractions and types of resources to determine how often the positive sustainability implications associated with the circular economy can be refuted. A regional setting was adequate for the case study because the low market value of the products recovered from organic waste makes their long-distance transport economically infeasible; nevertheless, the application of the developed framework to other waste fractions and other types of resources would require considering the global supply chains of resources and recycled materials.

New methodologies

Such broad frameworks could benefit from a consequential LCA modeling approach, which is particularly suitable for modeling the consequences of changes in the consumption and demand of raw materials. However, the difficulty related to compiling reliable marginal data hinders the application of this modeling approach.

Stochastic programming tools could help us tackle the uncertainty associated with the input parameters of the model, like waste composition, and determine its influence on the results. The effects of the seasonal variations in waste composition should be accounted for by means of dynamic optimization, which could also be used to plan waste management strategies compliant with the timeline established by European policies.

Additionally, game theory optimization would enable a more precise representation of the interests of the multiple actors that must cooperate to obtain a collective successful outcome from the implementation of a circular economy. An agent-based modeling approach could facilitate the analysis of the social dimension of sustainability, which, despite being vital for a complete sustainability assessment, was overlooked in this thesis. Thus, further research should investigate how the adoption of circularity practices affects job creation and other social indicators.

Finally, the automation of the data flow between the different pieces of software could facilitate and extend the use of the developed tools.

CAPÍTULO 4

CONCLUSIONES

"El futuro no se puede predecir, pero se puede inventar."

Dennis Gabor, físico húngaro (1900-1979)

En esta disertación se ha presentado y evaluado la utilidad de un nuevo marco metodológico para la gestión integrada y sostenible de residuos y recursos. La originalidad de la investigación reside en el enfoque metodológico desarrollado y su contribución a resolver algunos de los interrogantes que plantea la economía circular.

Alcance de las herramientas de tomas de decisiones desarrolladas

El marco metodológico propuesto se basa en la expansión de los límites de los sistemas integrados de gestión de residuos para incluir los subsistemas aguas arriba e intermedios en los que las materias primas son extraídas, procesadas, y posteriormente transformadas en residuos. La ventaja de ampliar estos sistemas, identificados en la tesis como Sistemas Circulares Integrados de Gestión de Residuos (SCIGR), es que permiten un análisis detallado de las consecuencias de recircular a los subsistemas aguas arriba los componentes residuales recuperados, facilitando los procesos de toma de decisiones.

Para garantizar que el diseño óptimo de un SCIGR satisfaga diferentes criterios de sostenibilidad, se integraron metodologías de análisis de flujo de materiales, análisis de ciclo de vida (ambiental y económico) y optimización multi-objetivo. Además, la aplicación de este enfoque a la gestión integrada de nutrientes y residuos municipales orgánicos requirió el uso de un modelo biogeoquímico para determinar la distribución y la especiación química de los nutrientes contenidos en los productos utilizados para enmendar el suelo. La combinación de estas metodologías, especialmente la integración del modelo de ecosistemas agrícolas en un sistema de soporte de decisiones, constituye una contribución innovadora al campo de la ingeniería de procesos y sistemas. Asimismo, el enfoque metodológico desarrollado se puede usar para identificar el espacio de toma de decisiones en el que los costes y beneficios asociados a la implementación de sistemas que promueven la circularidad de los recursos se distribuyen de forma justa entre los actores involucrados.

En cuanto al indicador de circularidad propuesto, permite cuantificar y comparar la circularidad de cada componente residual dentro del SCIGR. Por tanto, estas nuevas herramientas de toma de decisiones son adecuadas para explorar las implicaciones de mejorar la circularidad de los recursos sobre la sostenibilidad.

Consecuencias de mejorar la circularidad de los nutrientes en el sistema estudiado

De la aplicación del marco metodológico desarrollado al caso de estudio se extrajeron algunas conclusiones – ciertas bajo las hipótesis planteadas – sobre los efectos de incrementar la circularidad de los nutrientes en diferentes aspectos de la sostenibilidad.

- *Consumo de recursos.*

  Mejorar la circularidad del nitrógeno (N) comporta un incremento en el consumo de materias primas no renovables en el sistema estudiado.

- *Impactos ambientales.*

  Aumentar la circularidad del N puede conducir a un mayor nivel de eutrofización marina que el alcanzado en un sistema que depende en mayor medida de fertilizantes industriales. Esto se debe a que los cultivos absorben el N inorgánico presente en los fertilizantes industriales más eficientemente que de los productos recuperados de los residuos orgánicos, los cuales tienen una elevada proporción de N orgánico.

  Por el contrario, puesto que los mismos procesos unitarios están implicados en el incremento de la circularidad del N y la disminución de los impactos de calentamiento global, intensificar la circularidad del N reduce la huella de carbono del sistema estudiado.

- *Beneficios económicos.*
  Se demostró que en el caso de estudio mejorar la circularidad del N es incompatible con el aumento de los beneficios económicos de los gestores de residuos, debido a la notable inversión en nuevas tecnologías que esto supone.

Estos resultados desafían la premisa comúnmente aceptada – en la que se basa la legislación europea más reciente – de que la mejora en la eficiencia de los recursos derivada de la implementación de una economía circular conducirá a un crecimiento económico sostenido y a una reducción del consumo total de recursos y los impactos ambientales. En el caso de estudio, las mejoras en algunas dimensiones de la sostenibilidad se hicieron a expensas del deterioro de otros aspectos, por lo que es necesario encontrar un compromiso entre ellos. Por tanto, las implicaciones de la recuperación y reintroducción de cualquier componente residual en los ciclos productivos deben ser analizadas caso por caso. Sin embargo, de los resultados de la tesis se pueden derivar algunas recomendaciones generales.

ELABORACIÓN DE POLÍTICAS Y TOMA DE DECISIONES

Las configuraciones óptimas del sistema de gestión de los residuos orgánicos municipales generados en Cantabria se basan en la recogida selectiva, ya que permite reducir la huella de carbono del sistema siempre que se implemente en paralelo a las tecnologías adecuadas. No obstante, es muy improbable que para el año 2023 se haya normalizado la recogida selectiva de residuos orgánicos municipales, como establece la legislación europea, ya que por ahora no existe ningún plan de inversión en un nuevo sistema de recogida selectiva.

La rentabilidad económica es, junto a la viabilidad técnica de los procesos de reciclaje, uno de los principales factores limitantes que determinará el máximo grado de circularidad que un componente residual determinado puede alcanzar. Para minimizar los costes totales asociados a la circularidad de los recursos y evitar que los ciudadanos asuman estos costes en forma de impuestos, la legislación debería incentivar la compra de los productos recuperados de los residuos y la cooperación entre todas las partes implicadas en la recirculación de los recursos.

Equilibrar los impactos ambientales y la circularidad de los recursos es crucial para la implementación efectiva de una economía circular. Por tanto, la administración debería promover el uso de herramientas de tomas de decisiones basadas en una óptica de sistemas, como la presentada en esta tesis, para el diseño y análisis de sistemas de gestión de residuos y recursos. Dado que es posible que algunas de las consecuencias de la adopción de una economía circular contrarresten sus beneficios, las estrategias que buscan reducir el consumo de recursos, como la prevención de residuos, deberían tener precedencia sobre la circularidad de los recursos.

En definitiva, la economía circular no es más que un concepto; la estandarización de un método para verificar el alcance de la circularidad de los recursos, como el indicador de circularidad propuesto en esta tesis, podría ser útil para asegurar que este término no se emplee como una estrategia publicitaria engañosa.

LIMITACIONES DEL TRABAJO Y SUGERENCIAS PARA LA INVESTIGACIÓN FUTURA

Las mejoras adicionales en el marco metodológico propuesto podrían ayudarnos a comprender la relación entre la circularidad de los recursos y la sostenibilidad, y deberían centrarse en dos líneas de trabajo complementarias: la ampliación del modelo y la integración de nuevas herramientas metodológicas en el sistema de toma de decisiones desarrollado.

Ampliación del modelo

Incluyendo en el modelo otras fuentes de material orgánica (lodos de depuradora, estiércol o residuos forestales) y procesos aguas arriba adicionales (como la producción de otros cultivos), así como otras fracciones residuales, se podría reflejar de forma más ajustada a la realidad la complejidad del caso de estudio. Además, se deberían explorar los compromisos entre las categorías de impacto estudiadas y otros impactos ambientales asociados a la presencia de contaminantes en los productos recuperados de la materia orgánica, como la toxicidad humana o la ecotoxicidad.

La evaluación de la gestión de los residuos orgánicos y los nutrientes bajo la perspectiva del nexo agua-alimento-energía podría ayudar a analizar las sinergias entre estas necesidades humanas básicas, y delinear nuevas estrategias para cumplir con los Objetivos de Desarrollo Sostenible. Por otra parte, ampliar el marco metodológico desarrollado para cuantificar las consecuencias de las actividades de prevención de residuos es otra línea de investigación atractiva que podría conducir a nuevas políticas orientadas a reducir los residuos alimentarios, y por extensión, a sistemas alimentarios más eficientes.

Ya que un aporte de energía es esencial para el funcionamiento de cualquier proceso, acoplando el modelado de sistemas energéticos al marco metodológico desarrollado se podría conseguir una mejor descripción de las consecuencias de las configuraciones alternativas del sistema. Por ejemplo, en un escenario que considere un sistema energético con una penetración alta de energías renovables, un proceso de reciclaje con un alto consumo energético podría considerarse ambientalmente favorable, mientras que la valorización energética de los residuos podría desplazar fuentes de energía más limpias.

El marco metodológico desarrollado debería aplicarse a otras fracciones residuales y a otros tipos de recursos para determinar con qué frecuencia se pueden refutar las características de sostenibilidad atribuidas a la economía circular. Debido al bajo valor de los productos recuperados de los residuos orgánicos en el mercado – lo cual hace su transporte a larga distancia económicamente inviable –, el enfoque regional de este trabajo se adecua al caso de estudio. Sin embargo, la aplicación del marco desarrollado a otras fracciones residuales y otros tipos de recursos requeriría considerar las cadenas de suministro globales de recursos y materiales reciclados.

Nuevas metodologías

El marco metodológico desarrollado se podría beneficiar de un análisis de ciclo de vida consecuencial, apropiado para determinar las consecuencias de los cambios en el consumo y la demanda de materias primas. Sin embargo, la dificultad para encontrar los datos que este tipo de modelado requiere obstaculiza su aplicación.

Las herramientas de programación estocástica podrían contribuir a abordar la incertidumbre asociada a los parámetros del modelo, como la composición del residuo, y determinar su influencia en los resultados. Los efectos de las variaciones estacionales en la composición del residuo deberían ser considerados en una optimización dinámica, la cual también puede resultar útil para planificar estrategias de gestión de residuos conforme al cronograma establecido por la legislación europea.

Asimismo, la aplicación de los fundamentos de la teoría de juegos permitiría una representación de los intereses de los múltiples actores que deben cooperar para obtener un resultado común satisfactorio de la implementación de estrategias de economía circular. El modelado multi-agente podría facilitar el análisis de la dimensión social, la cual, a pesar de ser imprescindible para llevar a cabo una completa evaluación de la sostenibilidad, no ha sido estudiada en esta tesis. Por tanto, en el futuro se deberá investigar cómo la adopción de prácticas de economía circular afecta a la creación de empleo y a otros indicadores sociales.

Por último, la automatización de los flujos de datos entre los diferentes programas empleados podría facilitar y extender el uso de las herramientas desarrolladas.

NOMENCLATURE

η	N uptake efficiency
C	Carbon
CI_C	Carbon circularity indicator
CI_I	Circularity indicator of component *i*
CI_N	Nitrogen circularity indicator
CI_P	Phosphorus circularity indicator
CIWMS	Circular Integrated Waste Management System
DOC	Dissolved Organic Carbon
EFA	Energy Flow Analysis
Eh	Redox potential
FWE	Freshwater Eutrophication
IWMS	Integrated Waste Management System
GW	Global Warming
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LFA	Landfill area
MCDM	Multi-Criteria Decision-Making
MEU	Marine Eutrophication
MFA	Material Flow Analysis
MILP	Mixed Integer Linear Programming
MINLP	Mixed Integer Non-Linear Programming
MIP	Mixed Integer Programming
mix-OW	Organic waste separated from the mixed waste stream
MSW	Municipal Solid Waste
N	Nitrogen
NLP	Non-linear programming
NR-RM	Non-renewable raw materials
P	Phosphorus
SFA	Substance Flow Analysis
SOC	Soil Organic Carbon
SS-OW	Source Separated Organic Waste
SSR	Source Separation Rate
TAC	Total Annual Cost
TAX_{WM}	Waste management tax

SUPPORTING INFORMATION

APPENDIX A. WASTE COMPOSITION

Tables S1 and S2 show the energy content and the composition of the organic waste coming from the different waste collection systems. The energy content, Total Solids (TS), and water contents of each waste fraction, as well as the fossil C (FossilC), the biogenic C (BioC) and the anaerobically degradable biogenic carbon ($BioC_{AD}$) present in the organic waste are shown. It is assumed that the composition of the source separated organic waste (SS-OW) is 98% organic matter and 2% impurities. Even higher contents of organic matter have been achieved in real source separation experiences.[1] The composition of the organic waste collected in the mixed waste collection system and after the trommel separation (mix-OW) is taken from the Cantabrian waste management plan,[2] which also provides the amount of organic waste generated yearly in Cantabria (83,544 metric ton).

The estimated densities of SS-OW and mix-OW are 362.69 and 116.43 kg·m^{-3} respectively.

Table S1. Composition of SS-OW (kg· metric ton^{-1} organic waste)

Fraction name	Energy (MJ· metric ton^{-1})	TS	Water	BioC	BioC$_{AD}$	FossilC	N
Vegetable food waste	1363.51	94.75	317.04	45.01	40.08	0.23	1.80
Animal food waste	1070.87	55.49	73.97	30.74	19.97	0.63	3.88
Yard waste, flowers	2091.70	197.89	184.21	103.71	17.99	2.12	2.97
Wood	668.40	47.74	9.00	24.49	2.82	0.37	0.38
Non-recyclable glass	0.00	10.18	1.17	0.00	0.00	0.00	0.00
Food cans (tinplate/steel)	0.00	0.65	0.10	0.00	0.00	0.00	0.00
Beverage cans (Al)	0.00	0.17	0.02	0.00	0.00	0.00	0.00
Other metals	0.00	0.01	0.00	0.00	0.00	0.00	0.00
Paper and carton containers	30.54	2.59	0.74	1.16	0.43	0.01	0.01
Plastic bottles	3.70	0.13	0.02	0.00	0.00	0.10	0.00
Soft plastic	24.02	0.76	0.13	0.00	0.00	0.62	0.00
Hard plastic	2.67	0.09	0.00	0.00	0.00	0.07	0.01
Non-recyclable plastic	11.52	0.46	0.04	0.00	0.00	0.32	0.00
Juice cartons	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Textiles	1.96	0.13	0.01	0.05	0.00	0.02	0.00
Other non-combustibles	0.00	2.45	0.16	0.02	0.00	0.02	0.00

Table S2. Composition of mix-OW (kg· metric ton^{-1} organic waste)

Fraction name	Energy (MJ· metric ton^{-1})	TS	Water	BioC	$BioC_{AD}$	FossilC	N
Vegetable food waste	1341.46	93.22	311.91	44.28	39.43	0.22	1.77
Animal food waste	1053.33	54.58	72.76	30.24	19.65	0.62	3.82
Yard waste, flowers	2057.96	194.70	181.24	102.04	17.70	2.08	2.92
Wood	18.97	1.35	0.26	0.69	0.08	0.01	0.01
Non-recyclable glass	0.00	45.77	5.26	0.00	0.00	0.00	0.00
Food cans (tinplate/steel)	0.00	2.95	0.45	0.00	0.00	0.00	0.00
Beverage cans (Al)	0.00	0.75	0.07	0.00	0.00	0.00	0.00
Other metals	0.00	0.07	0.01	0.00	0.00	0.00	0.00
Paper and carton containers	137.41	11.67	3.34	5.24	1.93	0.03	0.02
Plastic bottles	16.77	0.58	0.07	0.00	0.00	0.45	0.00
Soft plastic	107.97	3.43	0.56	0.01	0.00	2.80	0.01
Hard plastic	12.04	0.41	0.01	0.00	0.00	0.33	0.02
Non-recyclable plastic	51.79	2.06	0.16	0.01	0.00	1.46	0.01
Juice cartons	0.31	0.02	0.00	0.01	0.00	0.00	0.00
Textiles	8.83	0.57	0.04	0.22	0.00	0.07	0.02
Other non-combustibles	0.00	11.03	0.71	0.07	0.00	0.07	0.00

APPENDIX B. LIFE CYCLE MODELING

Technologies for the recovery of nutrients from the liquid digestate

Table S3 and S4 show the life cycle inventories (LCI) – compiled from the literature –[3-5] of the technologies modeled to treat the liquid digestate. The content of NH_4 and phosphorus (P) in the liquid digestate is 781.97e^{-6} and 60.74e^{-6} kg·kg^{-1} respectively.

Table S3. LCI of the ammonia stripping and absorption (ASA) process

INPUTS	
3.02	kg H_2SO_4 (96% wt)·kg NH_4^{-1} in the digestate
0.87	kg NaOH·kg NH_4^{-1} in the digestate
5.48	kg H_2O·kg NH_4^{-1} in the digestate
20.96	kWh heat·metric ton^{-1} digestate
0.81	kWh electricity·metric ton^{-1} digestate
OUTPUTS	
1.11E-02	kg SO_3·kg NH_4^{-1} in the digestate
4.80E-03	kg NH_3·kg NH_4^{-1} in the digestate
5.81E-03	kg $(NH_4)_2SO_4$·kg^{-1} liquid digestate

Table S4. LCI of the struvite precipitation (SP) process

INPUTS	
1.561	kg MgO·(kg $(PO_4)^{3-})^{-1}$–P in the liquid digestate
0.064	kWh·m^{-3} digestate
OUTPUT	
4.45	kg $MgNH_4PO_4·6H_2O$·(kg $(PO_4)^{-3})^{-1}$–P in the liquid digestate

Trommel separation

The trommel electricity consumption was assumed to be 0.55 kWh·metric ton^{-1} for the SS-OW and 1.35 kWh·metric ton^{-1} for the mixed waste. The data was extrapolated from commercial trommels,[6] considering that the densities of the materials that compose waste are those provided by EPA Victoria.[7]

Table S5 compiles the trommel separation efficiencies for each material; i.e., how much of each material ends up in the organic waste stream after the sorting process. The efficiencies were calculated from the data provided by the Cantabrian waste management plant.[2]

Only mix-OW and the SS-OW sent to anaerobic digestion (AD) are subjected to the trommel pretreatment.

Table S5. Trommel separation efficiencies

Fraction name	Sorting efficiencies (%)
Vegetable food waste	90.23
Animal food waste	90.23
Yard waste, flowers	90.23
Wood	3.20
Non-recyclable glass	51.10
Food cans (tinplate/steel)	20.00
Beverage cans (aluminium)	20.00
Other metals	20.00
Paper and carton containers	3.50
Plastic bottles	51.10
Soft plastic	51.10
Hard plastic	51.10
Non-recyclable plastic	51.10
Juice cartons	0.10
Textiles	0.50
Other non-combustibles	8.40

The efficiency of the magnetic and Eddy current separators is 73.42%.[2]

Landfill

Figure S1 shows the sub-models associated with the landfill unit process in EASETECH.[8]

The generation of landfill gas is modeled according to equation S1, which indicates that the initial amount of anaerobically degradable biogenic carbon in fraction *i* of waste ($BioC_{AD_i}0$) experiences a first order decay.

$$BioC_{AD_i} = BioC_{AD_i}0 \cdot e^{-k \cdot t} \qquad \text{(Equation S1)}$$

The decay rates (k) of each waste fraction are shown in Table S6. The assumed time horizon is 100 years.

Table S6. 1st order decay rate for the anaerobically degradable biogenic C in landfill

Fraction name	k (year^{-1})
Vegetable food waste	0.137
Animal food waste	0.137
Yard waste, flowers	0.162
Wood	0.014
Non-recyclable glass	0.000
Food cans (tinplate/steel)	0.000
Beverage cans (aluminium)	0.000
Other metals	0.000
Paper and carton containers	0.019
Plastic bottles	0.000
Soft plastic	0.000
Hard plastic	0.000
Non-recyclable plastic	0.000
Juice cartons	0.019
Textiles	0.021
Other non-combustibles	0.000

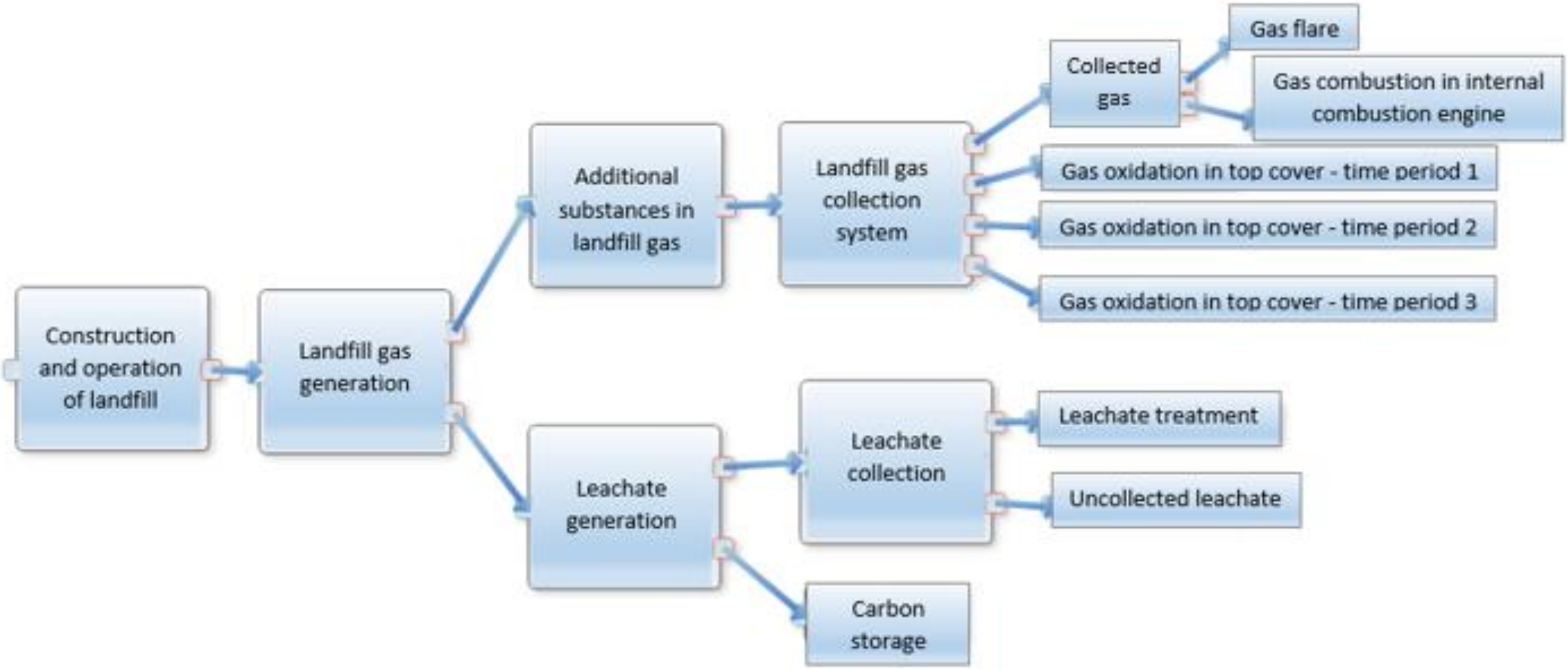

Figure S1. Sub-models of the landfill unit process

63% of the generated biogas is CH_4, the remaining 37% is CO_2. The concentration of additional trace substances in the biogas per m^3 of the CO_2 and CH_4 mixture is compiled in Table S7.

Table S7. Additional substances in biogas

Substance	Amount ($g \cdot m^{-3}$)
Phenol	0.001
CO	0.028
Dichlorobenzene	0.006
Ethylchloride	0.010
Chloromethane	0.0003
Naphtalene	0.0006
Hg	1E-6
H_2S	0.04
NMVOC	0.03
VC	0.004
TCE	0.004
PCE	0.01
Benzene	0.008
Chlorobenzene	0.002
Ethylbenzene	0.02
Propylbenzene	0.002
Dichloromethane	0.02
Chloroform	0.0003
Carbon tetrachloride	5E-05
Xylenes	0.04
Toluene	0.11
CFC11	0.001
CFC12	0.005
CFC113	0.0005
HCFC21	0.01
HCFC22	0.003

Table S8 shows the efficiency of the gas collection system and the fraction of the not collected CH_4 that is oxidized.

Table S8. Efficiency of the gas collection system and the fraction of the not collected CH_4 that is oxidized

Time period (years)	Collected gas (%)	Not collected CH_4 oxidized to CO_2 (%)
0-5	45	10
5-15	80	20
15-55	95	36
55-100	0	36

Table S9 compiles the fraction of each substance in the collected gas that is oxidized in the combustion process (the combustion in flare of the gas that leaks, which is 7.21% of the collected gas, and the gas that is combusted for energy generation). These assumptions are valid for well monitored, state-of-art landfills with liners. All the substances are oxidized to CO_2, except for H_2S, which is transformed into SO_2.

Table S9. Fraction of each substance that is oxidized in the combustion process (%)

Substance	In flare	For energy generation
CH_4	99.0	99.0
NMVOC	97.7	97.2
H_2S	97.7	97.2
Vynil chloride	98.0	93.0
Trichloroethylene	98.0	93.0
Perchloroethene	98.0	93.0
Benzene	99.7	86.1
Chlorobenzene	99.7	86.1
Dichlorobenzene	99.7	86.1
Ethylbenzene	99.7	86.1
Propylbenzene	99.7	86.1
Ethylchloride	98.0	93.0
Chloromethane	98.0	93.0
Dichloromethane	98.0	93.0
Chloroform	98.0	93.0
Carbon tetrachloride	98.0	93.0
Xylenes	99.7	86.1
Toluene	99.7	86.1
CFC11	98.0	93.0
CFC12	98.0	93.0
CFC113	98.0	93.0
HCFC21	98.0	93.0
Phenol	99.7	86.1
Naphtalene	2.3	86.1

Table S10 shows the additional substances (besides CO_2) that are generated in the combustion process.

Table S10. Additional substances generated in the combustion and treatment of landfill gas ($kg \cdot m^{-3}$ CH_4)

	In flare	For energy generation
CO	7.40E-04	8.46E-03
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	6.70E-13	3.60E-12
NO_x	9.10E-04	8.20E-03
PAH, polycyclic aromatic hydrocarbons	1.00E-06	1.00E-06
Particulates (2.5-10µm)	7.00E-05	1.80E-04
Polychlorinated biphenyls	1.00E-06	1.00E-06
SO_2	1.70E-04	1.70E-04
HCl	1.70E-04	4.00E-05
HF	4.00E-05	4.00E-05

The leachate generation was calculated assuming a net infiltration rate of 300 $mm \cdot year^{-1}$. Table S11 compiles the concentration of different substances in the leachate. It was assumed that 99.9% of the leachate generated during the first 80 years of operation was collected, whereas in the remaining 20 years of the time horizon, 87% of the generated leachate is collected.

Table S11. Concentration of different substances in the leachate (mg·L^{-1})

Substance	Period 1 (1 year)	Period 2 (2 years)	Period 3 (7 years)	Period 4 (90 years)
DEHP	1.00E-02	1.00E-02	1.00E-02	1.00E-02
Ethylchloride	1.00E-01	1.00E-01	1.00E-01	1.00E-01
Propylbenzene	2.00E-03	2.00E-03	2.00E-03	2.00E-03
Fe	7.80E+02	6.60E+02	3.20E+02	1.50E+01
SO_4	5.00E+02	4.40E+02	2.50E+02	8.00E+01
Se	1.00E-02	9.00E-03	6.00E-03	3.00E-03
Ca	1.20E+03	1.00E+03	5.00E+02	6.00E+01
Cl	2.12E+03	1.90E+03	1.10E+03	3.60E+02
Na	7.00E+02	6.00E+02	3.50E+02	1.00E+02
Ag	1.20E-01	1.00E-01	5.00E-02	1.00E-02
As	3.00E-02	3.00E-02	3.00E-02	3.00E-02
Ba	5.00E-01	4.50E-01	3.00E-01	1.60E-01
Cd	1.30E-02	1.20E-02	9.00E-03	6.00E-03
Cr	7.00E-02	6.50E-02	5.20E-02	4.00E-02
Cu	7.00E-02	7.00E-02	7.00E-02	7.00E-02
Hg	4.00E-04	3.00E-04	2.00E-04	1.00E-04
Mg	4.70E+02	4.10E+02	2.30E+02	6.00E+01
Ni	7.00E-02	7.00E-02	7.00E-02	7.00E-02
Pb	5.00E-02	4.50E-02	3.20E-02	2.00E-02
Zn	4.00E+00	3.50E+00	2.00E+00	7.00E-01
NH_3	3.50E+03	2.90E+03	1.60E+03	1.10E+02
PO_4	1.00E+00	1.00E+00	1.00E+00	1.00E+00
COD	2.00E+04	1.50E+04	5.00E+03	4.00E+02
BOD	1.60E+04	1.00E+04	1.00E+03	4.00E+01
TSS	6.00E+01	6.00E+01	6.00E+01	6.00E+01
VC	5.00E-03	4.80E-03	4.40E-03	4.00E-03
TCE	5.00E-03	5.00E-03	6.00E-03	6.00E-03
PCE	1.00E-02	9.00E-03	6.00E-03	3.00E-03
Benzene	6.00E-03	5.60E-03	4.80E-03	4.00E-03
Chlorobenzene	3.00E-03	3.00E-03	3.00E-03	3.00E-03
Dichlorobenzene	6.00E-03	6.00E-03	6.00E-03	6.00E-03
Ethylbenzene	3.00E-02	2.80E-03	2.40E-02	2.00E-02
Dichloromethane	3.00E-02	2.50E-02	1.30E-02	3.00E-03
Chloroform	3.00E-04	3.00E-04	3.00E-04	3.00E-04
Carbon tetrachloride	2.00E-04	2.00E-04	2.00E-04	2.00E-04
Xylenes	8.00E-02	7.50E-02	6.20E-02	5.00E-01
Toluene	9.00E-02	8.00E-02	5.00E-02	2.00E-02
Phenol	8.00E-04	3.00E-03	5.00E-03	3.00E-03
Naphtalene	3.00E-02	2.80E-02	2.40E-02	2.00E-02

Table S12 shows the LCI of the landfill unit process. The electricity generated in the gas combustion process is 2.64 kWh·m^{-3} CH_4.

Table S12. LCI of landfill operation

INPUTS		
Gravel*	0.18	kg·kg^{-1} TWW
Clay*	8.2E-02	kg·kg^{-1} TWW
Copper*	9.87E-09	kg·kg^{-1} TWW
Steel sheets*	1.40E-04	kg·kg^{-1} TWW
Aluminum*	5.80E-08	kg·kg^{-1} TWW
Polyvinylchloride resin*	1.00E-05	kg·kg^{-1} TWW
Polyethylene high density granulate*	2.30E-04	kg·kg^{-1} TWW
Polypropylene fibers*	4.00E-08	kg·kg^{-1} TWW
Diesel oil	3.20E-03	kg·kg^{-1} TWW
Electricity consumption (construction and operation of landfill)	8.00E-03	kWh·kg^{-1} TWW
Electricity consumption (leachate treatment)	4.43E-02	kWh·kg^{-1} leachate

*Construction material for the landfill cells

Incineration

Figure S2 shows the sub-models of the incineration unit process modeled with EASETECH.[8]. Table S13 compiles the percentages associated with the material transfer of the different substances present in waste after the incineration process. The energy efficiency of the incineration process is 23.8%.

Table S13. Substance transfer in the incineration unit process (%)

	Air	Fly Ash	Fe scrap	Al srap	Wastewater	Degradation	Bottom ash
Water						100.0000	
VS						100.0000	
C	99.90						0.1000
Ca	0.00	20.5900					79.4100
Cl	0.1073	32.1300			62.4600		5.3000
F						0.2300	99.7700
H						100.00	
K		20.5900					79.4100
N						4.2400	95.7600
Na		20.5900					79.4100
O		18.3600				10.0700	71.5700
S	0.0990	60.9100			15.0000		23.9910
Al		8.6000		58.2300			33.1700
As	0.0121	58.9200			0.4554		40.6100
Cd	0.0064	88.1300			0.0311		11.8300
Cr	0.0394	16.7700			0.0455		83.1500
Cu	0.0026	7.3500			0.0157		92.6300
Fe		3.1900	84.5000				12.3100
Hg	0.7476	96.2500			0.0936		2.9090
Mg		20.5900					79.4100
Mn		20.5900					79.4100
Mo		2.5400			0.8517		96.6100
Ni	0.0329	12.5600			0.0873		87.3200
Pb	0.0008	51.2900			0.2384		48.4700
Sb	0.1190	59.8400			1.2340		38.8100
Zn		48.1800			0.0643		51.7600

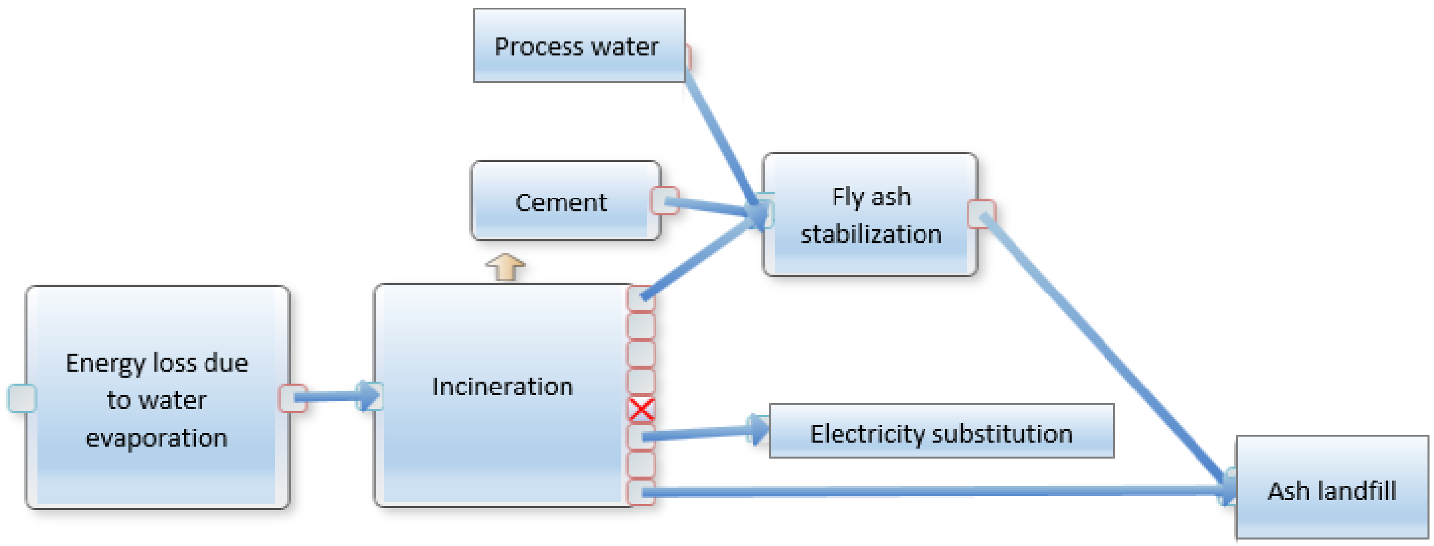

Figure S2. Sub-models of the incineration unit process

Table S14 compiles the substances generated in the incineration process and present in the gas stream (besides CO_2).

Table S14. Substances generated in the incineration unit process

CO	3.30E-05	kg·kg^{-1} TWW
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	1.80E-14	kg·kg^{-1} TWW
HCl	5.3 E-06	kg·kg^{-1} TWW
HF	3.90E-07	kg·kg^{-1} TWW
NOx	8.49E-04	kg·kg^{-1} TWW
SO_2	2.91E-06	kg·kg^{-1} TWW
Particulates > 10 µm	3.00E-05	kg·kg^{-1} TWW

Table S15 shows the LCI of the incineration process. The inputs associated with the operation of the ash landfill are excluded from the inventory.

Table S15. LCI of the incineration unit process

INPUTS		
Activated carbon	9.29E-04	kg·kg^{-1} TWW
Process water	5.05E-01	kg·kg^{-1} TWW
Hydrated Lime	8.57E-03	kg·kg^{-1} TWW
Natural Gas	1.45E-02	MJ·kg^{-1} TWW
Cement (fly ash stabilization)	5E-01	kg·kg^{-1} fly ash
Process water (fly ash stabilization)	5E-01	kg·kg^{-1} fly ash

Anaerobic digestion

Figure S3 shows the different stages of the anaerobic digestion process considered in EASETECH.[8]

Table S16 compiles the gas yields of the different materials present in the organic waste in the anaerobic digestion process. The yield is defined as the fraction of $BioC_{AD}$ of each material that is transferred to the gas phase. 63 % of the biogas is CH_4, the remaining fraction is CO_2. 2% of the generated biogas leaks and is combusted in a gas flare, whereas the rest of the biogas is combusted for energy generation. The electricity generated in the biogas combustion process is 2.64 kWh·m^{-3} CH_4.

Table S16. Gas yield in the anaerobic digestion process

Fraction name	Yield $BioC_{AD}$ (%)
Vegetable food waste	70
Animal food waste	70
Yard waste, flowers	70
Wood	45
Paper and carton containers	45
Juice cartons	45

The fractions of each substance that are oxidized in the biogas combustion processes are those shown in Table S8, whereas the additional substances that are generated in the combustion process are compiled in Table S10.

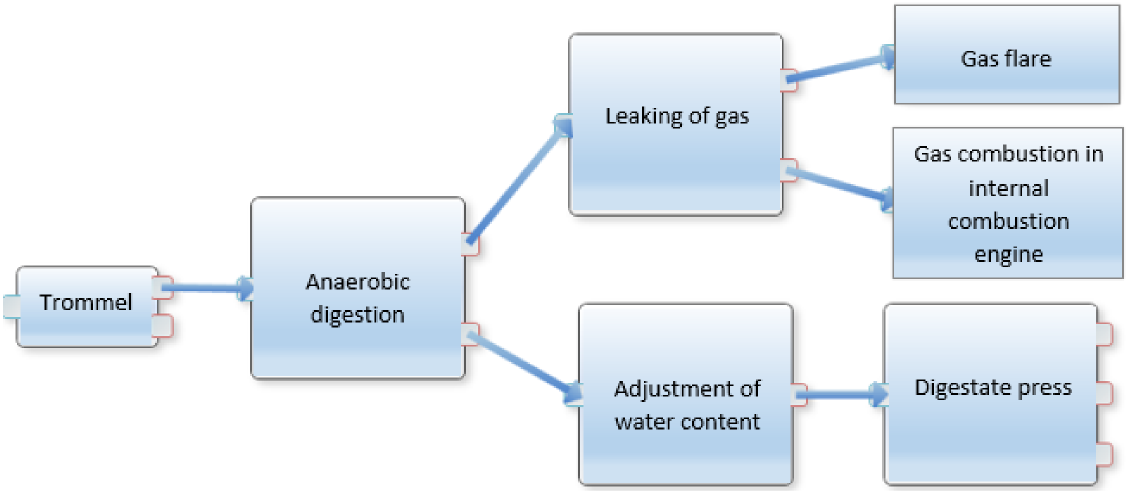

Figure S3. Sub-models of the anaerobic digestion unit process

The water content in the digestate is 96% (in weight). Table S17 compiles the distribution of different components of the liquid digestate between the liquid and the solid digestate after the screw press separation.[9] The rest of the components of the digestate are assumed to be transferred to the solid digestate. Regarding the energy consumption of the screw press, it is assumed to be 10 MJ per metric ton of digestate.[10]

Table S17. Distribution of the digestate components between the liquid and the solid digestate (%)

Substance	Liquid digestate	Solid digestate
Water	91.43	8.57
N	54.11	45.89
C	38.24	61.76
Mg	20.5	79.5
Ca	3.18	96.82
P	27.32	72.68
K	92.37	7.63
VS	30.56	69.44
Ash	6.95	93.05

Table S18 shows the LCI of the anaerobic digestion process. The results are expressed per kg of TWW that enters the pretreatment process. However, the table does not include the inputs associated with the trommel and the screw press. The heat required to achieve a temperature of 55 ^{o}C in the reactor was calculated assuming that the specific heat of the solids is 3 $kJ \cdot kg^{-1}$.

Table S18. LCI of the anaerobic digestion unit process

INPUTS		
Electricity consumption	4.12E-2	$kWh \cdot kg^{-1}$ TWW
Diesel	7.57E-4	$l \cdot kg^{-1}$ TWW
Process water	5.113	$kg \cdot kg^{-1}$ TWW
Heat from natural gas	0.14	$kJ \cdot kg^{-1}$ TWW

Composting

Figures S4 and S5 show the sub-models of windrow (W) and tunnel (T) composting processes modeled with EASETECH.[8].

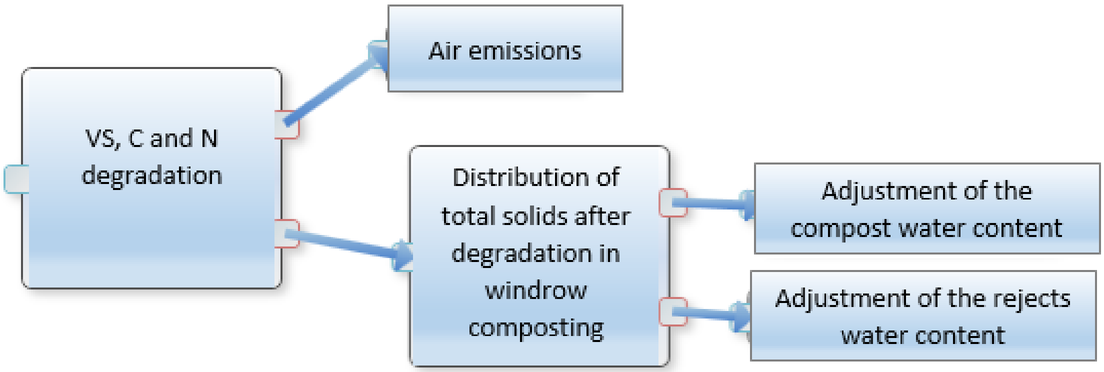

Figure S4. Sub-models of the windrow composting unit process

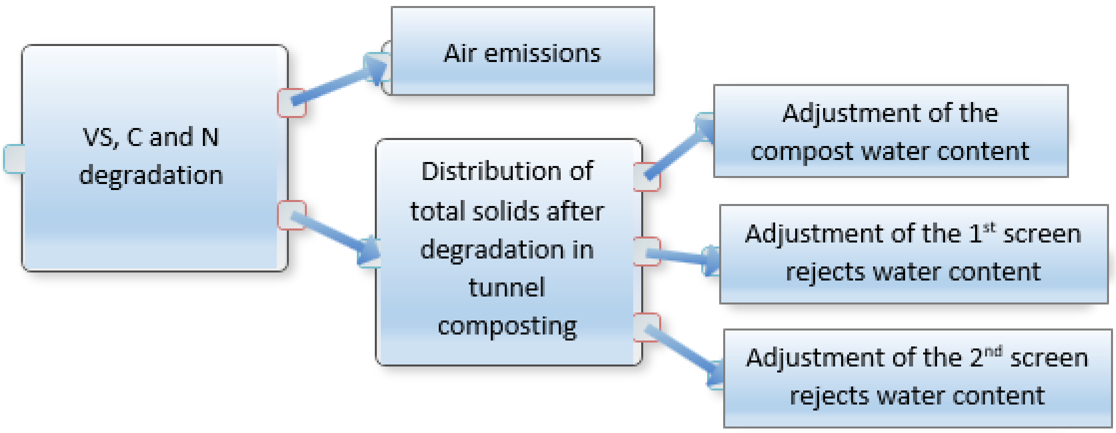

Figure S5. Sub-models of the tunnel composting unit process

Table S19 compiles the percentages of carbon (C) in each fraction of the organic waste that is degraded in the composting processes. On the other hand, 71% of the nitrogen (N) present in organic waste is degraded in the windrow and in the tunnel composting processes.

Table S19. Percentage of C in each fraction of the organic waste that is degraded in the composting processes

Fraction name	Windrow	Tunnel
Vegetable food waste	74.56	73.54
Animal food waste	74.56	73.54
Yard waste, flowers	74.56	63.79
Wood	11.28	20
Paper and carton containers	10	10
Juice cartons	5	5
Textiles	5	5
Others	0	0

Table S20 shows the percentages of each degraded element that are converted into new compounds in the composting process.

Table S20. Transformation of degraded elements into new compounds

Element	Conversion (%)	Compound
C	99.990	CO_2
C	0.010	CH_4
N	0.100	N_2
N	0.985	NH_3
N	1.400	N_2O
N	97.515	NO_X

Table S21 shows the additional gas emissions that are generated in the composting process.

Table S21. Additional Substances generated in the composting process

Substance	Amount ($kg \cdot kg^{-1}$ TWW)
Terpenes	1.22E-06
H_2S	1.93E-07

Table S22 shows the distribution of the materials that compose organic waste between the generated compost (comp) or bio-stabilized material (BS) and rejects for different composting technologies.

Table S22. Distribution of the materials between compost and reject

	Tunnel composting			**Windrow composting**	
Fraction name	Comp /BS	1st screen reject	2nd screen reject	Comp /BS	Rejects
Vegetable food waste	95	0	5	95	5
Animal food waste	95	2	3	95	5
Yard waste, flowers	95	0	5	95	5
Wood	80	20	0	50	50
Non-recyclable glass	5	30	65	50	50
Food cans (tinplate/steel)	0	30	70	0	100
Beverage cans (aluminium)	5	30	65	5	95
Other metals	5	30	65	5	95
Paper and carton containers	10	45	45	50	50
Plastic bottles	5	30	65	20	80
Soft plastic	5	30	65	20	80
Hard plastic	5	30	65	20	80
Non-recyclable plastic	5	30	65	20	80
Juice cartons	10	45	45	5	95
Textiles	5	30	65	50	50
Other non-combustibles	5	30	65	5	95

After the curing phase, the water content in the compost/bio-stabilized material is reduced to 30%.

The LCI of the composting processes is compiled in Table S23.

Table S23. LCI of the composting unit processes

INPUTS		
Electricity consumption		
Grinder	10.600	kWh·metric ton^{-1}
Windrow turner	0.24	kWh·metric ton^{-1}
Post-screening	0.9000	kWh·metric ton^{-1}
Front end loader	8.064	kWh·metric ton^{-1}
Diesel		
Grinder fuel consumption	0.250	l·kWh^{-1}
Windrow turner fuel consumption	0.127	l·kWh^{-1}
Front end loader	0.260	l·kWh^{-1}

Land application of products

Table S24 compiles the LCI of the application of products to land. The required amounts of nutrients from each type of recovered product are compiled in Tables S30 and S34.

Table S24. LCI of the application of products to land

Diesel	0.010	l·kg^{-1} applied N
Diesel	0.010	l·kg^{-1} applied P
NH_4NO_3 (as N)	1.000	kg N·kg^{-1} N from mineral fertilizer
$(NH_4)_2HPO_4$ (DAP)	0.235	kg DAP·kg^{-1} P from mineral fertilizer

Transport

The waste collected in Cantabria is either directly taken to the mechanical-biological treatment plant located in the municipality of Meruelo, or to one of the seven transfer stations of the region (to be subsequently transported to the mechanical-biological treatment plant). Table S26 compiles the distance of each regional municipality to the closest transfer station, as well as the distance from the transfer stations to Meruelo. The table also shows the population of each municipality.[11] The average distance that waste must be transported is calculated assuming that all the citizens generate the same amount of waste, regardless of where they live. It was calculated that the average distance that waste must go through from the municipalities to the transfer station is 24.7 km, and from the transfer station to Meruelo 21.6 km. Table S25 shows the LCI of the transport process.

Table S25. LCI of waste transport

Transport	Vehicle type	LCI
Curbside collection	Collection Vehicle, 10t Euro3, urban traffic	0.00157 L diesel·kg^{-1} TWW
From municipality to transfer station	Truck, 7.5t-12t, Euro5, urban traffic	24.7·2 km·kg^{-1} TWW
From transfer station to Meruelo	Truck, 20t-26t, Euro5, highway	21.6·2 km·kg^{-1} TWW

It was assumed that the products recovered from organic waste need to travel from the Meruelo facility to the different municipalities of Cantabria. On the other hand, it was assumed that all the mineral fertilizers that are applied in the region are produced in Bilbao. Table S27 compiles the cultivated areas of each Cantabrian municipality[11] and the distances from each municipality to Meruelo and Bilbao. Under the hypothesis that the amount of fertilizers and products recovered from organic waste that are sent to each municipality is directly proportional to the cultivated area in each municipality, the average distance between Meruelo and the land where the recovered products are applied is 66.54 km, and the distance between Bilbao and the fertilized land, 105.05 km.

Table S26. Population and distance of each Cantabrian municipality to the closest transfer station

Municipality	Population (inhabitants)	Transfer stations (km) 1	2	3	4	5	6	7	8
Alfoz de Lloredo	2,466	13							
Ampuero	4,181						12		
Anievas	314		39						
Arenas de Iguña	1,711		40						
Argoños	1,720								12
Arnuero	2,091								40
Arredondo	480						13		
Astillero, El	18,134								31
Bárcena de Cicero	4,124								14
Bárcena de Pie de Concha	710		45						
Bareyo	1,999								8
Cabezón de la Sal	8,345	0							69
Cabezón de Liébana	601			0					133
Cabuérniga	1,012	13							
Camaleño	977			12					
Camargo	30,611								32
Campoó de Yuso	696				25				
Cartes	5,733		17						
Castañeda	2,687		0						39
Castro-Urdiales	31,901							0	48
Cieza	556		35						
Cillorigo de Liébana	1,310			13					
Colindres	8,331								22
Comillas	2,228	10							
Corrales de Buelna, Los	11,003		25						
Corvera de Toranzo	2,074		20						
Enmedio, Campoó de	3,778				16				
Entrambasaguas	4,943								13
Escalante	747								11
Guriezo	2,359								41
Hazas de Cesto	1,522								9
Hermandad de Campoó de Suso	1,643				0				105
Herrerías	641	33							
Lamasón	298			37					
Laredo	11,446								25
Liendo	1,227								30
Liérganes	2,372								25
Limpias	1,813								25
Luena	615		34						

Municipality	Population (inhabitants)	Transfer stations (km) 1	2	3	4	5	6	7	8
Marina de Cudeyo	5,174	6							21
Mazcuerras	2,119								
Medio Cudeyo	7,482								23
Meruelo	1,965								0
Miengo	4,741		24						
Miera	395								38
Molledo	1,589		37						
Noja	2,562								11
Penagos	2,060								26
Peñarrubia	349			25					
Pesaguero	311			8					
Pesquera	71				24				
Piélagos	24,574		16						
Polaciones	237			33					
Polanco	5,794		19						
Potes	1,360	6							
Puente Viesgo	2,877		7						
Ramales de la Victoria	2,827						0		40
Rasines	959						11		
Reinosa	9,496				11				
Reocín	8,318		21						
Ribamontán al Mar	4,422								13
Ribamontán al Monte	2,231								8
Rionansa	1,058	31							
Riotuerto	1,610								19
Rozas de Valdearroyo, Las	275				27				
Ruente	1,044	7							
Ruesga	870						10		
Ruiloba	765	12							
San Felices de Buelna	2,381		18						
San Miguel de Aguayo	160				30				
San Pedro del Romeral	453					31			
San Roque del Río Miera	386					23			
Santa Cruz de Bezana	12,679								38
Santa María de Cayón	9,078								32
Santander	172,656								39
Santillana del Mar	4,184		23						
Santiurde de Reinosa	265				21				
Santiurde de Toranzo	1,602					25			
Santoña	11,085								16

		Transfer stations (km)							
Municipality	**Population (inhabitants)**	1	2	3	4	5	6	7	8
San Vicente de la Barquera	4,196	18							
Saro	512					15			
Selaya	1,931					0			52
Soba	1,249								58
Solórzano	1,012								11
Suances	8,579		28						
Tojos, Los	399	26							
Torrelavega	52,819		13						
Tresviso	71			51					
Tudanca	147	41							
Udías	903	5							
Valdáliga	2,272	14							
Valdeolea	987				26				
Valdeprado del Río	331				34				
Valderredible	1,001				45				
Val de San Vicente	2,763	26							
Vega de Liébana	788			19					
Vega de Pas	795					19			
Villacarriedo	1,636					8			
Villaescusa	3,883								34
Villafufre	1,017					17			
Valle de Villaverde	327							29	
Voto	2,725								30

Table S27. Cultivated area of each municipality and distances to Meruelo and Bilbao

Municipalities	Cultivated area (ha)	Distance to Meruelo (km)	Distance to Bilbao (km)
Alfoz de Lloredo	104.26	65	133
Ampuero	22.68	29	68
Anievas	7.28	77	145
Arenas de Iguña	51.99	77	145
Argoños	0.64	12	72
Arnuero	190.55	40	82
Arredondo	0.61	52	91
Astillero (El)	15.68	31	94
Bárcena de Cicero	163.01	14	68
Bárcena de Pie de Concha	5.33	77	146
Bareyo	158.32	8	82
Cabezón de la Sal	64.61	69	137
Cabezón de Liébana	13.18	133	202
Cabuérniga	23.32	82	150
Camaleño	22.97	137	205
Camargo	56.42	32	101
Campoo de Yuso	11.54	68	148
Cartes	1.59	53	122
Castañeda	87.66	39	107
Castro-Urdiales	145.15	48	35
Cieza	59.35	71	139
Cillorigo de Liébana	12.86	134	202
Colindres	0.12	22	61
Comillas	24.09	76	144
Corrales de Buelna (Los)	49.35	60	129
Corvera de Toranzo	40.04	55	124
Campoo de Enmedio	33.54	95	164
Entrambasaguas	75.87	13	83
Escalante	6.73	11	70
Guriezo	24.71	41	53
Hazas de Cesto	36.12	9	75
Hermandad de Campoo de Suso	4.55	105	185
Herrerías	71.06	101	170
Lamasón	8.56	115	184
Laredo	8.73	25	60
Liendo	6.84	30	53
Liérganes	20.22	25	94
Limpias	3.16	25	64
Luena	12.55	70	138
Marina de Cudeyo	134.58	21	90
Mazcuerras	60.13	66	135
Medio Cudeyo	48.07	23	92
Meruelo	80.01	0	81
Miengo	46.93	47	116
Miera	4.49	38	106
Molledo	30.98	74	142
Noja	1.15	11	82
Penagos	40.86	26	94
Peñarrubia	26.00	120	188
Pesaguero	7.17	141	210
Pesquera	35.96	86	167
Piélagos	302.42	41	109
Polaciones	0.46	133	201
Polanco	39.47	49	118
Potes	0.08	130	199
Puente Viesgo	89.10	43	111
Ramales de la Victoria	9.06	40	79
Rasines	7.70	35	74

Municipalities	Cultivated area (ha)	Distance to Meruelo (km)	Distance to Bilbao (km)
Reinosa	8.61	106	162
Reocín	49.93	57	125
Ribamontán al Mar	610.78	13	93
Ribamontán al Monte	78.94	8	80
Rionansa	13.76	113	181
Riotuerto	3.73	19	89
Rozas de Valdearroyo (Las)	2.03	108	127
Ruente	4.11	76	145
Ruesga	13.58	49	88
Ruiloba	93.64	77	146
San Felices de Buelna	33.21	54	122
San Miguel de Aguayo	0.06	92	161
San Pedro del Romeral	10.09	76	144
San Roque de Riomiera	8.07	42	110
Santa Cruz de Bezana	85.34	38	107
Santa María de Cayón	181.02	32	100
Santander	16.93	39	102
Santillana del Mar	200.13	58	127
Santiurde de Reinosa	28.87	85	154
Santiurde de Toranzo	11.96	52	120
Santoña	1.10	16	71
San Vicente de la Barquera	36.19	83	152
Saro	6.89	40	108
Selaya	9.16	52	121
Soba	62.72	58	97
Solórzano	102.26	11	76
Suances	203.11	63	131
Tojos (Los)	0.20	95	163
Torrelavega	128.98	50	28
Tresviso	4.00	153	221
Tudanca	0.22	123	191
Udías	5.61	70	139
Valdáliga	104.51	84	152
Valdeolea	683.14	110	179
Valdeprado del Río	81.70	113	181
Valderredible	1,352.17	124	193
Val de San Vicente	175.27	92	160
Vega de Liébana	21.29	143	211
Vega de Pas	2.71	71	104
Villacarriedo	87.47	45	113
Villaescusa	11.87	34	102
Villafufre	13.00	43	111
Valle de Villaverde	12.12	58	40
Voto	120.47	30	75

Additional data

The global warming (GW), marine eutrophication (MEU) and freshwater eutrophication (FWE) impacts compiled in Table S28 have been taken from Ecoinvent 3.3.[12] The data reflect the average environmental impacts of European production processes, except for the environmental impacts related to the generation of electricity, which represent the Spanish grid mix.

Table S28. GW, MEU and FWE of different commodities

	Functional unit	GW (kg CO_2-eq)	MEU (kg N-eq)	FWE (kg P-eq)
Spanish electricity	1 kWh	3.05E-01	4.00E-05	1.46E-03
Heat from natural gas	1 MJ	1.41E-01	4.63E-06	2.79E-03
Cement	1 kg	8.84E-01	7.06E-05	1.40E-02
Steel sheet	1 kg	3.31E-01	6.66E-05	4.95E-02
Aluminum	1 kg	4.90E00	1.40E-03	5.81E-01
PEHD granulate	1 kg	1.92E00	1.30E-04	7.38E-07
PP fibers	1 kg	2.334E00	2.05E-04	2.00E-05
PVC	1 kg	2.71E00	2.18E-04	5.81E-05
Urea	2.14 kg	3.18E00	1.02E-03	1.28E-01
H_2SO_4	1 kg	2.57E-01	4.22E-04	2.76E-01
MgO	1 kg	1.04E00	1.58E-04	4.85E-02
NaOH	1 kg	1.41E00	1.22E-04	2.92E-04
HCl	1 kg	1.14E00	3.89E-04	2.25E-01
Water	1 kg	6.5E-03	3.25E-06	1.36E-06
$(NH_4)_2HPO_4$	1 kg	1.43E00	4.33E-04	7.55E-04
NH_4NO_3	1 kg N	8.72E00	2.35E-03	8.46E-04

APPENDIX C. DNDC MODELING

The soil properties that the DNDC[13] software demands as input data are the following:

- Texture: loam.[14]
- pH: 4.55.[14]
- Organic C content (0 cm - 10 cm): 0.0412 kg $C \cdot kg^{-1}$ soil.[15]
- Bulk density (0 cm - 10 cm): 1.03 $g \cdot cm^{-3}$.[15]

Table S29 shows the distribution of N in the recovered products.

Table S29. Distribution of N in the compost/bio-stabilized material and the solid digestate (SD) (%)

	COMP/BS[16]	SD[17]
Organic-N	93.00	62.96
NH_4^+-N	1.00	37.04
NO_3^--N	6.00	0

Appendix C1 makes reference to the application of the recovered products to cover the corn N requirements, whereas Appendix C2 refers to the corn P requirements. Tables S30 and S34 cover the amount of nutrients present in the different products that need to be applied to land to produce 7.11 metric tons of corn per ha and year. Tables S31-S33 and S35-S37 show the distribution of the C, N and P present in the applied products in the environment.

Appendix C1. Fertilizing products applied to cover the corn N requirements

Table S30. Amount of nutrients from different sources applied to the soil to cover the N requirements

Product	Unit process	kg N·ha^{-1} ·year^{-1}	kg P·ha^{-1}·year^{-1}	kg C·ha^{-1} ·year^{-1}
Fert (NH_4NO_3)		128.5	0.0	0.0
COMP	T	196.9	88.9	571.2
BS	T	208.6	96.6	496.8
COMP	W	210.4	96.9	482
BS	W	218.9	101.34	438.9
SD	AD	232.3	49.4	464.4
$(NH_4)_2SO_4$	ASA	128.5	0.0	0.0
$MgNH_4PO_4·6H_2O$	SP	128.5	442.5	0.0

Table S31. Distribution of the applied N (%)
Application of recovered product needed to cover the N requirements

Product	Unit process	N uptake	Leach NO_3	N_2O	NO	Stored N
Fert (NH_4NO_3)	-	0.7929	0.1989	0.0045	0.0037	0.0000
COMP	T	0.5987	0.2926	0.0084	0.0007	0.0996
BS	T	0.5936	0.3033	0.0084	0.0007	0.0940
COMP	W	0.5936	0.3054	0.0084	0.0007	0.0919
BS	W	0.5872	0.3121	0.0083	0.0007	0.0917
SD	AD	0.5284	0.4136	0.0096	0.0010	0.0474
$(NH_4)_2SO_4$	ASA	0.7929	0.1989	0.0045	0.0037	0.0000
$MgNH_4PO_4·6H_2O$	SP	0.7929	0.1989	0.0045	0.0037	0.0000

Table S32. Distribution of applied P (%)
Application of recovered product needed to cover the N requirements

Product	Unit process	P uptake	Leach P	Stored P in soil
COMP	T	0.1073	0.7129	0.1798
BS	T	0.0987	0.7314	0.1699
COMP	W	0.0995	0.7307	0.1698
BS	W	0.0943	0.7403	0.1654
SD	AD	0.1928	0.5623	0.2448
$MgNH_4PO_4·6H_2O$	SP	0.0217	0.9248	0.0536

Table S33. Distribution of applied C (%)
Application of recovered product needed to cover the N requirements

Product	Unit process	CO_2	Leach C	Stored C in soil
COMP	T	0.9221	0.0439	0.0334
BS	T	0.9164	0.0478	0.0358
COMP	W	0.9156	0.0486	0.0358
BS	W	0.9102	0.0515	0.0383
SD	AD	0.9280	0.0492	0.0228

The distribution of nutrients was calculated from the results of the DNDC software.

Appendix C2. Fertilizing products applied to cover the corn P requirements

Table S34. Amount of nutrients from different sources applied to the soil to cover the P requirements

Product	Fert ($(NH_4)_2HPO_4$)	COMP	BS	COMP	BS	SD	$MgNH_4PO_4\cdot$ $\cdot 6H_2O$
Unit process	**-**	**T**	**T**	**W**	**W**	**AD**	**SP**
kg N·ha^{-1}·year^{-1} (Fertilizer N)	12.3	110.9	112	111.9	112.5	102.7	122.4
kg P·ha^{-1}year^{-1} (Fertilizer P)	13.6	0	0	0	0	0	0
kg N·ha^{-1}·year^{-1} (Recovered N)	0	30	29.3	29.8	29.3	63.8	6.1
kg P·ha^{-1}·year (Recovered P)	0	13.6	13.6	13.6	13.6	13.6	13.6
kg C·ha^{-1}·year^{-1} (Recovered C)	0	872	697	682	588	1274	0

Table S35. Distribution of the applied N (%)
Application of recovered product needed to cover the P requirements

Product	Unit process	N uptake	Leach NO_3	N_2O air emissions	NO air emissions	Stored N in soil
Fert $(NH_4)_2HPO_4$	-	0.7929	0.1989	0.0045	0.0037	0.0000
COMP	T	0.7538	0.2376	0.0050	0.0036	0.0000
BS	T	0.7562	0.2353	0.0049	0.0036	0.0000
COMP	W	0.7557	0.2358	0.0049	0.0036	0.0000
BS	W	0.7572	0.2344	0.0048	0.0036	0.0000
SD	AD	0.6710	0.3206	0.0050	0.0034	0.0000
$MgNH_4PO_4 \cdot 6H_2O$	SP	0.7929	0.1989	0.0045	0.0037	0.0000

Table S36. Distribution of applied P (%)
Application of recovered product needed to cover the P requirements

P uptake	Leach P	Stored P in soil
0.3993	0.2041	0.3996

Table S37. Distribution of applied C (%)
Application of recovered product needed to cover the P requirements

Product	Unit process	CO_2	Leach	Stored
COMP	T	0.90481	0.09519	-0.0832
BS	T	0.90059	0.09941	-0.0907
COMP	W	0.90012	0.09988	-0.0911
BS	W	0.89764	0.10236	-0.0955
SD	AD	0.91230	0.08770	-0.0631

The distribution of nutrients was calculated from the results of the DNDC software.

APPENDIX D. EFFICIENCIES OF THE UPSTREAM PROCESSES

Tables S38 and S39 compile the efficiencies of the recycling unit processes for each component. The efficiencies are expressed as kg of component *i* recovered per kg of component *i* that enters the recycling process.

Table S38. Efficiency of the recycling processes that handle organic waste

Unit process Product	AD SD	W COMP	T COMP	W BS	T BS
C	0.3422	0.2666	0.3421	0.2473	0.2931
N	0.3766	0.2561	0.2595	0.2737	0.2730
P	0.6466	0.9416	0.945	0.9374	0.9331

Table S39. Efficiency of the recycling processes that handle the liquid digestate

	ASA	SP
C	0.0000	0.0000
N	0.9520	0.0172
P	0.0000	0.5625

Tables S40 and S41 compile the efficiencies of the corn production process for the application of each type of product (to cover N or P requirements respectively). They are expressed as either kg of nutrient taken up by corn per kg of nutrient applied to the soil (for N and P), or kg of C consumed by microbes per kg of C applied to the soil.

Table S40. Efficiencies of the corn production processes for the application of each type of recovered product. Application of recovered product needed to cover the N requirements

Product Unit process	COMP T	BS T	COMP W	BS W	SD AD	$(NH_4)_2SO_4$ ASA	$MgNH_4PO_4{\cdot}6H_2O$ SP
C	0.4159	0.4497	0.4574	0.4819	0.4233	0.0000	0.0000
N	0.6618	0.6284	0.6234	0.6015	0.5687	0.9743	0.9743
P	0.0960	0.0890	0.0897	0.0854	0.1591	0.3993	0.0212

Table S41. Efficiencies of the corn production processes for the application of each type of recovered product. Application of recovered product needed to cover the P requirements

Product **Unit process**	**COMP** **T**	**BS** **T**	**COMP** **W**	**BS** **W**	**SD** **AD**	**$(NH_4)_2SO_4$** **ASA**	**$MgNH_4PO_4 \cdot 6H_2O$** **SP**
C	0.5476	0.5694	0.5725	0.5856	0.5223	0.0000	0.0000
N	0.7245	0.7149	0.7098	0.7050	0.5258	0.0000	0.9743
P	0.3993	0.3993	0.3993	0.3993	0.3993	0.0000	0.3993

APPENDIX E. RESULTS OF THE SENSITIVITY ANALYSIS

Figures S6 and S7 show the results of the single-objective optimizations of each scenario in Chapter 5 under the assumptions of the sensitivity analysis.

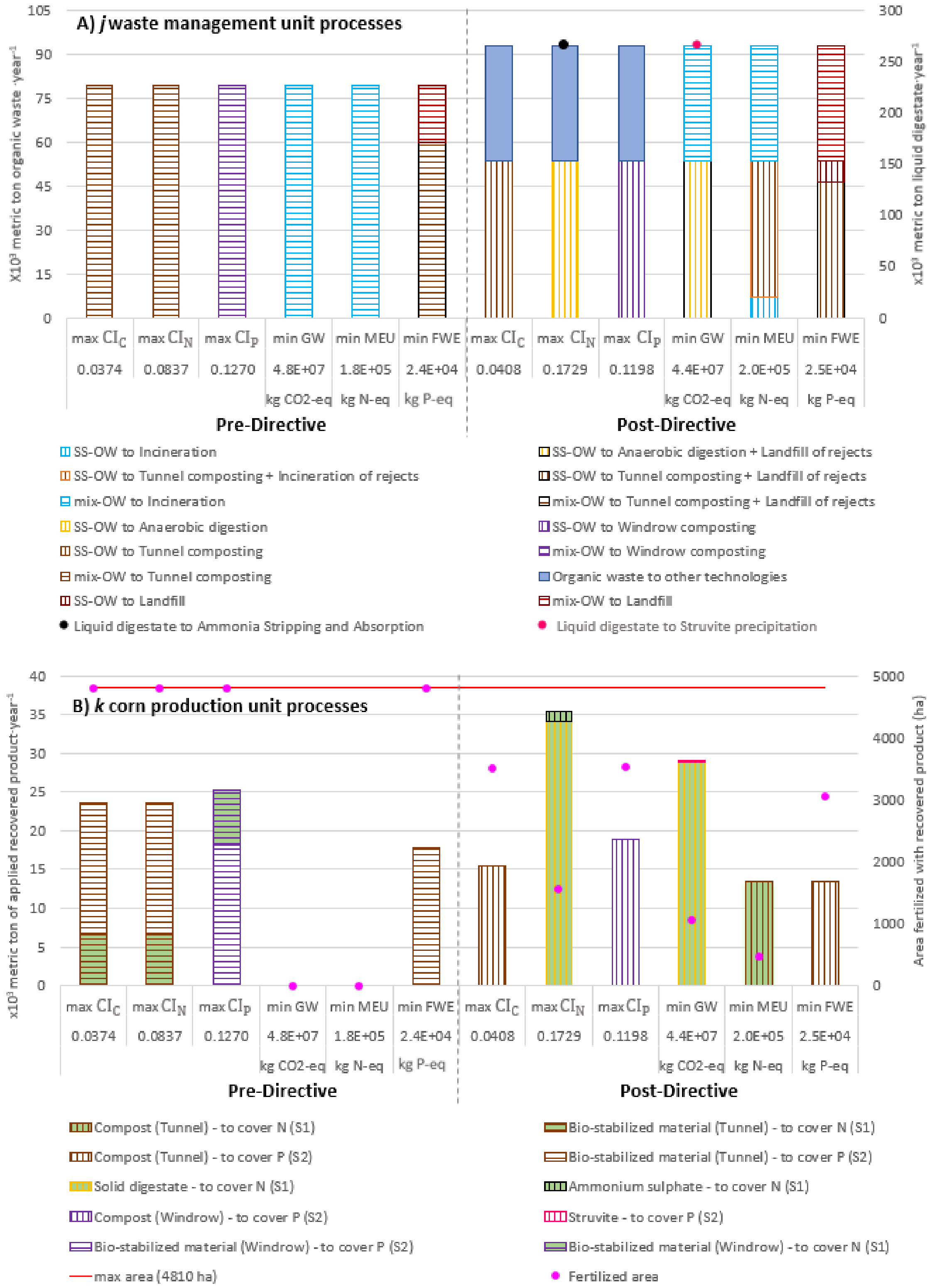

Figure S6. Values of the objective functions and decision variables for the optimization of the Pre-Directive and Post-Directive scenarios

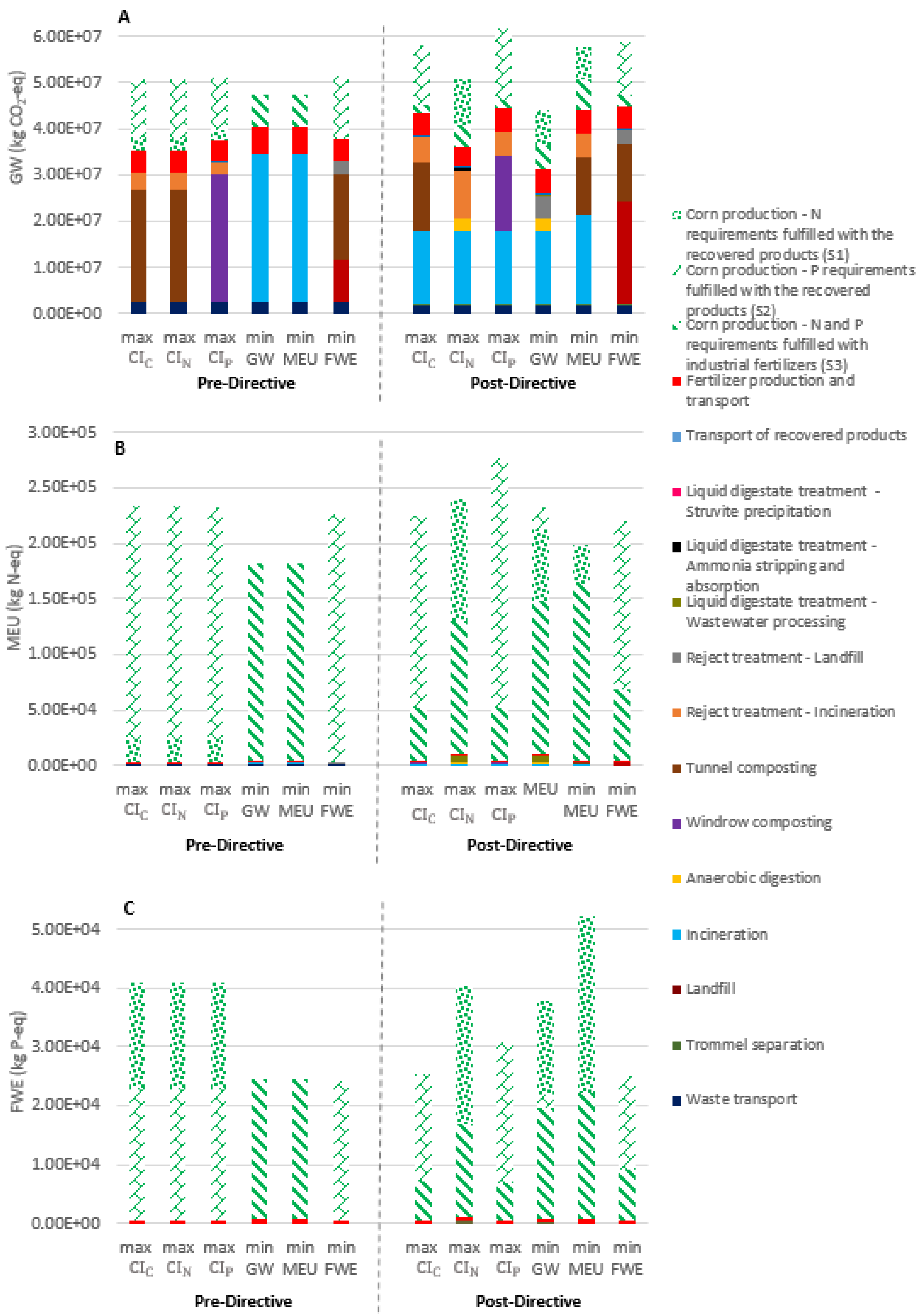

Figure S7. Contribution of the unit processes to the environmental impacts in the Pre-Directive and Post-Directive scenarios

APPENDIX F. ECONOMIC MODELING

Unless indicated otherwise, all the economic data refer to year 2015. The default data is taken from the SWOLF framework.[18]

Common data to all unit processes

Conversion from 2015€ to 2015$:[19] 1.11 $

Interest rate:[20] 7%

Amortization period: 15 years

- Time parameters

Annual operating days: 260 days·year^{-1}

Daily operating hours: 8 hours·day^{-1}

- Wages

Operator wage:[21] 10.69 2015€·hour^{-1}

Manager wage:[21] 14.70 2015€·hour^{-1}

Laboratory technician wage:[21] 13.57 2015€·hour^{-1}

Driver wage:[21] 10.11 2015€·hour^{-1}

- 2015 prices in Cantabria

Price of gasoline:[22] 1.204 2015€·L^{-1}

Price of diesel:[22] 1.0961 2015€·L^{-1}

Price of water for industrial use:[23] 0.96 €·m^{-3}

Price of agricultural land:[24] 13971 2015€·ha^{-1}

- Price of chemicals

Price of MgO:[25] 0.255 €·kg^{-1}

Price of H_2SO_4:[26] 40.54 2015€·metric ton^{-1}

Price of NaOH:[27] 297.30 2015€·metric ton^{-1}

Price of natural gas:[28] 0.0224 2015€·metric ton^{-1}

Collection and transport

Two types of waste containers were considered, a 240 L container for the SS-OW, and an 800 L container for the mix-OW. Assuming that the collection frequency of SS-OW and mix-OW is once and twice per week respectively, the estimated number of required containers is 10,658 for the SS-OW and 17,541 for the mix-OW. It was assumed that the number of stops that the trucks make is the same as the number of containers. The number of trucks required for the collection of the SS-OW and the mix-OW (33 and 120) was calculated from the characteristics of the trips of the collection trucks (Table S42).

Table S42. Characteristics of the trips of the collection trucks

	SS-OW	mix-OW	
Number of trips per year	8,892	97,562	
Number of containers emptied per trip	62	18	
Average distance from container to transfer station	24.7	24.7	km
Average speed from garage to collection point	60	60	$km \cdot h^{-1}$
Loading time per container	1	2	min
Travel time between stops	5	2.5	min
Unloading time per trip	15	15	min

It was assumed that that the compaction density of organic waste and inorganic materials is increased 1.5 and 3 times respectively at the transfer stations.[29] The characteristics of the trips of the trucks that carry the waste from the transfer stations to the waste management facilities are compiled in Table S43. With those data it was estimated that 2 trucks are needed for the SS-OW and 7 trucks the mix-OW.

Table S43. Characteristics of the trips of the trucks that transport the waste from the transfer stations to the waste management facilities

	SS-OW	mix-OW	
Number of trips per year	2,847	17,625	
Average distance from transfer station to waste plant	21.6	21.6	km
Average speed from garage to collection point	80	80	$km \cdot h^{-1}$
Loading time per truck	15	15	min
Unloading time per truck	15	15	min

The costs associated with the waste containers and trucks are compiled in Tables S44-S46.

Table S44. Annualized capital costs (CC) and operation and management costs (OM) of waste containers[30]

	240 L container	800 L container	
Price	75.42	301.68	2015€·unit
Amortization period	5	5	years
Washing costs	4.64	5.80	2015€·unit^{-1}
Number of wash cycles	7	7	year^{-1}
Other maintenance costs	6.96	8.12	2015€·unit^{-1}

Table S45. CC and OM of trucks[30]

	Collection trucks	Trucks departing from transfer station	
Capacity	15	30	m^3
Price	139,236	260,843	2015€·vehicule^{-1}
Amortization period	10	10	years
Maintenance costs	18,101	36,201	2015€·vehicule^{-1}·year^{-1}
Number of operators	2	1	Operators·vehicle^{-1}
Overhead personnel	10	10	%

The TAC of the studied collection and transport system, with a 50% source separation rate (SSR), is compared in Table S49 with the TAC of the current collection and transport system implemented in the region (SSR=0%).

Table S46. TAC of collection and transport (2015€·metric ton^{-1})

	SSR=50%	SSR=0%
CC	24.70	22.35
OM	58.66	50.71
TAC	83.37	73.06

Pretreatment

The characteristics of the trommel are shown in Table S47. The TAC of the trommel, magnet and Eddy current separator are compiled in Table S48.

Table S47. Characteristics of trommel

Trommel cost	125,863	2015€·unit^{-1}
Installation cost	30	% equipment cost
Number of units	0.0025	unit·(metric ton·day^{-1})$^{-1}$

Table S48. TAC of trommel, magnet and Eddy current separator (2015€·metric ton^{-1} output)

	Trommel	Eddy current separator	Magnet
CC	0.57	0.71	1.17
OM	0.63	0.22	1.24
TAC	1.20	0.93	2.41

Summary of the TAC associated with the management of solid organic waste

The Total Annual Cost (TAC) related to the management of 1 metric ton of organic waste entering the different waste management unit processes are compiled in Table S49. The TAC is divided into annualized capital costs (CC), operation and management costs (OM), closure costs (clos), the landfill tax (tax), the revenues derived from the sale of electricity (elec) and the operation subsidies (OS). It includes the TAC associated with the collection, transport and pretreatment of the amount of organic waste required to obtain 1 metric ton of organic waste at the inlet of the unit processes that manage the solid organic waste. The TAC related to the treatment of the rejects in incineration or landfill (+I or +L) is also included in Table S52. It is expressed as €·metric ton^{-1} of organic waste that enters the unit process.

Table S49. TAC of the unit processes that handle the organic waste (2015€·metric ton^{-1})

	AD+I	AD+L	W+I	W+L	T+I	T+L	Lold	Lnew	Iold	Inew
CC	31.44	31.32	0.16	0.04	14.48	14.30	0.66	10.94	2.02	67.64
OM	38.08	30.35	27.75	24.69	34.31	32.37	8.07	8.07	57.25	57.25
clos	0.12	0.50	0.09	0.20	0.11	0.12	3.17	3.17	1.07	1.07
elec	-8.99	-5.72	-1.21	-0.17	-0.79	-0.25	-0.78	-0.78	-11.53	-11.53
OS	-1.73	-1.82	0.00	-0.05	0.00	-0.04	-0.64	-0.64	0.00	0.00
tax	0.00	0.32	0.00	0.12	0.00	0.07	2.00	2.00	0.00	0.00
TAC	58.92	54.96	26.78	24.83	48.10	46.56	12.48	22.77	48.81	114.43

Since the costs of the unit processes that are already available in Cantabria are assumed to be amortized, only the costs associated with the construction of new cells within the previously excavated landfill are accounted for as capital costs of the old landfill. The CC of the new incinerator are due to the construction of new cells for the ash landfill. Likewise, the CC of windrow composting, 12 €·metric ton^{-1}, was not taken into account.

Capacity restrictions

Table S50 lists the capacity restrictions of the solid waste management unit processes. They are expressed as annual flows of organic waste. In the case of incineration and landfill, the capacities refer to the sum of flows of organic waste and rejects from other unit processes.

Table S50. Capacity restrictions of the solid waste management unit processes

	Lower bound (metric ton·year^{-1} organic waste)	**Upper bound (metric ton·year^{-1} organic waste)**
Old landfill	–	25,000
New landfill	–	–
Old incinerator	–	15,000
New incinerator	25,000	–
Anaerobic digestion	12,500	–
Windrow composting	–	–
Tunnel composting	10,000	–

Composting

The parameters required to perform the LCC of the composting unit processes (obtained from SWOLF) are the following:

- Time parameters

Active composting time (windrow composting): 70 days

Active composting time (tunnel composting): 20 days

Curing time: 30 days

Frequency of turning in active composting (windrow): 0.25 day^{-1}

Frequency of turning during curing phase (windrow and tunnel): 0.143 day^{-1}

- Odor control system

OC blower power required per air flow rate:0.0579 kW·(m^{3}·min^{-1})$^{-1}$

Motor efficiency: 64.94%

Odor control air flow required: 54000 m^{3}·(metric ton·day^{-1})$^{-1}$

- Construction cost (2010$)

Grading cost per acre: 31400 $\$\cdot ha^{-1}$

Paving cost: 123000 $\$\cdot ha^{-1}$

Building cost (equip., stag.): 75 $\$\cdot m^{-2}$

Office cost: 430 $\$\cdot m^{-2}$

Fencing cost: 30 $\$\cdot m^{-1}$

- Additional construction costs

Engineering, design, supervision: 20% DPC (Direct Project Costs)

Management overheads: 20% DPC

Commissioning: 5% IPC

Contingency: 15% IPC

Contractor's fees: 10% IPC

Interest during construction: 10% IPC

- Construction requirements

Grading requirement: 0.03 $ha\cdot(metric\ ton\cdot day^{-1})^{-1}$

Paving required: 0.02 $ha\cdot(metric\ ton\cdot day^{-1})^{-1}$

Warehouse requirement: 34 $m^2\cdot(metric\ ton\cdot day^{-1})^{-1}$

Office requirement: 2.36 $m^2\cdot(metric\ ton\cdot day^{-1})^{-1}$

Land requirement: 0.4 $ha\cdot(metric\ ton\cdot day^{-1})^{-1}$

Fencing requirement: 13 $m\cdot(metric\ ton\cdot day^{-1})^{-1}$

- Workers

Operators required: 0.1 $person\cdot(metric\ ton\cdot day^{-1})^{-1}$

Managers required: 0.041 $person\cdot(metric\ ton\cdot day^{-1})^{-1}$

Overhead percentage: 10%

- Equipment Requirements

Windrow Turner: 0.0022 $units\cdot(metric\ ton\cdot day^{-1})^{-1}$

Tub grinder: 0.0038 $units\cdot(metric\ ton\cdot day^{-1})^{-1}$

Front end loader: 0.003 units·(metric ton·day^{-1})$^{-1}$

Bobcat: 0.003 units·(metric ton·day^{-1})$^{-1}$

Screen: 0.0025 units·(metric ton·day^{-1})$^{-1}$

Blower (windrow composting): 0.1 units·(metric ton·day^{-1})$^{-1}$

In-vessel reactor: 0.01 units·(metric ton·day^{-1})$^{-1}$

Equipment installation cost: 0.3%

- Office area

Office Area required per metric ton per day of material: 2.36m^2·(metric ton·day^{-1})$^{-1}$

Energy required to power an office: 290 kWh·m^{-2}·year^{-1}

Table S51. Equipment costs

Equipment	Cost per unit (2010$·unit^{-1})	Equipment Life (years)	Repair Cost (%Initial Cost)
Windrow turner	231,380	10	60
Tub grinder	321,362	10	60
Pre-trommel	128,545	10	60
Front End Loader	192,817	10	60
Bobcat	38,563	10	60
Post trommel	128,545	10	60
Blower	306	10	60
Vacuum system	52,063	10	60
In-vessel reactor	260,314	15	60

Tables S52 and S53 show the distribution of the CC and TAC in the windrow and tunnel composting unit processes.

Table S52. Annualized capital costs (CC) in windrow and tunnel composting (2010$·(metric ton·day^{-1})$^{-1}$)

	WINDROW	TUNNEL
Grading cost	942	942
Paving cost	2,460	2,460
Building cost (equip., stag.)	2,550	2,550
Office cost	1,014	1,014
Fencing cost	390	390
Land acquisition cost	5,707	5,707
Equipment	2,859	5,903
DIRECT PROJECT COSTS	15,922	18,966
Engineering, design, supervision	3,184	3,793
Management overheads	3,184	3,793
INSTALLED PROJECT COSTS	22,291	26,553
Commissioning	1,115	1,328
Contingency	3,344	3,983
Contractor's fees	2,229	2,655

The OM are calculated as the sum of the labor, overhead, electricity, fuel and equipment repair costs.

Table S53. TAC of windrow and tunnel composting (2015€· metric ton^{-1})

	WINDROW	TUNNEL
CC	11.98	14.27
OM	24.32	32.14
TAC	36.30	46.42

The CC, OM and TAC are compared with the exponential curves obtained by Tsilemou and Panagiotakopoulos[31] for windrow (Figures S8-S10) and tunnel composting (Figures S11-S13).

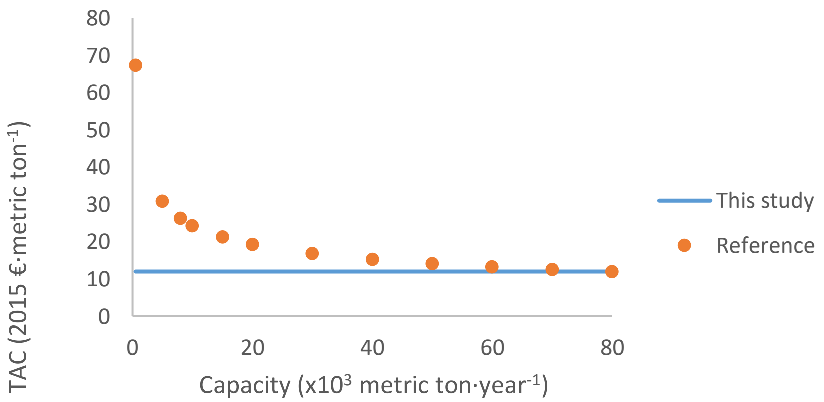

Figure S8. CC of windrow composting

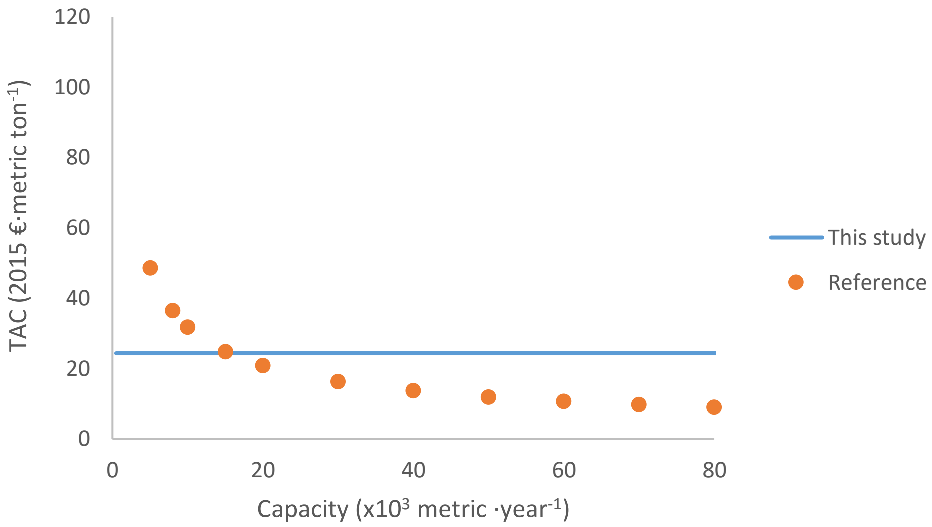

Figure S9. OM of windrow composting

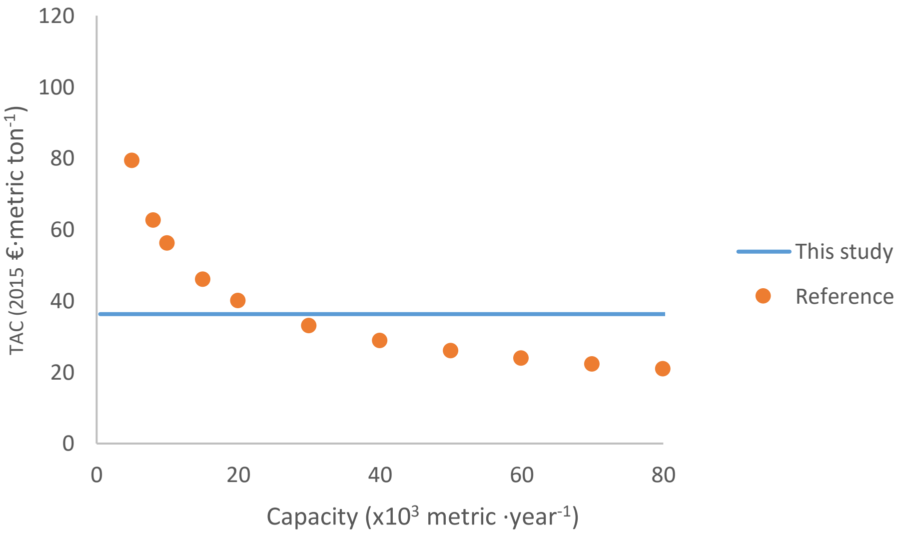

Figure S10. TAC of windrow composting

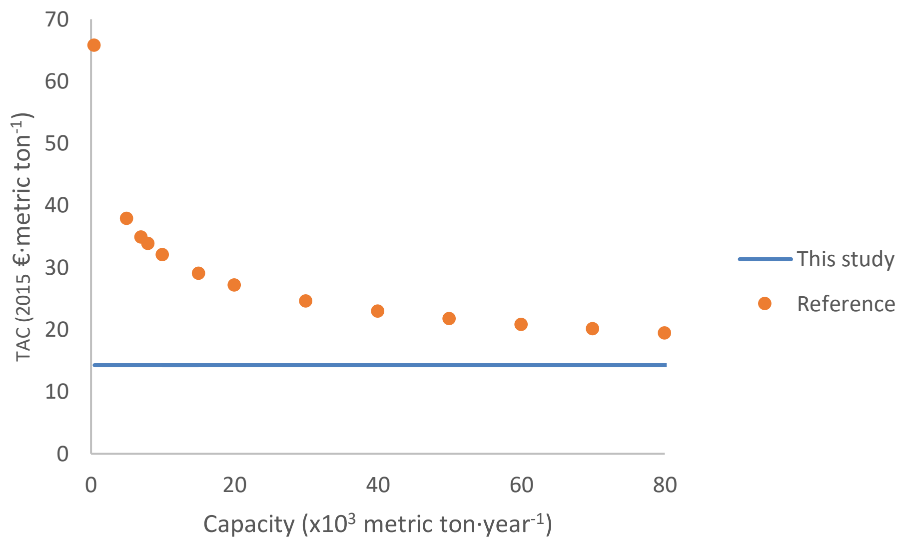

Figure S11. CC of tunnel composting

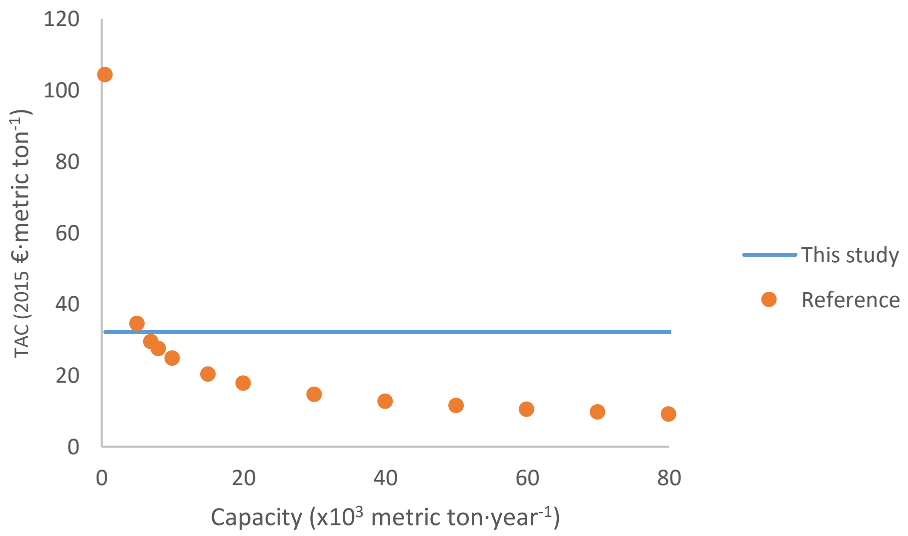

Figure S12. OM of tunnel composting

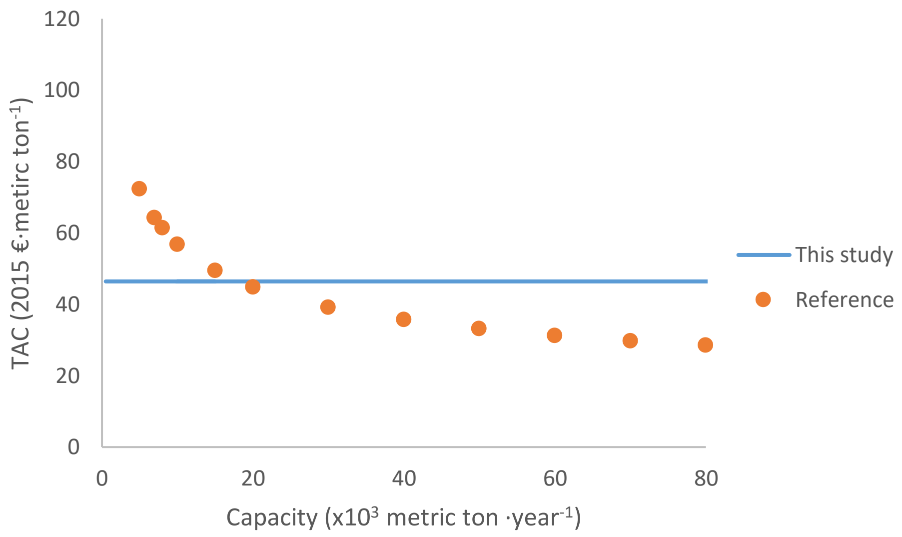

Figure S13. TAC of tunnel composting

Anaerobic digestion

The data required for the economic evaluation of the wet anaerobic digestion unit process was extrapolated from a feasibility study.[32] The contributions to the CC, the OM and the TAC are shown in Tables S54- S56. The required electricity is assumed to be subtracted from the on-site power generation.

Table S54. CC of wet anaerobic digestion (2005$·(metric ton·year^{-1})$^{-1}$)

General site costs	8.66
New buildings	51.15
Major tankage	62.50
Pre-treatmente equipment	1.07
Wet processing equipment	69.00
Flaring and odor control	5.25
Electrical and steam generation	42.26
Engineering costs	23.99

Table S55. OM of wet anaerobic digestion (2015€· metric ton^{-1})

Staff requirements	
Plant manager	4.13
Operators	7.32
Utilities and fuel	
Fuel	0.91
Water	4.07
Electricity	0.00
Natural gas	0.45
Maintenance	
Equipment	4.49
Builidings	0.30
Tanks and odor	0.68
Wastewater treatment	4.70

Table S56. TAC of wet anaerobic digestion (2015€·metric ton^{-1})

CC	31.22
OM	29.40
elec	-5.59
TAC	55.02

The CC, OM and TAC are compared with the exponential curves obtained by Arnò et al[33] for wet anaerobic digestion in Figures S14-S16. These figures do not include the revenues derived from the sale of electricity.

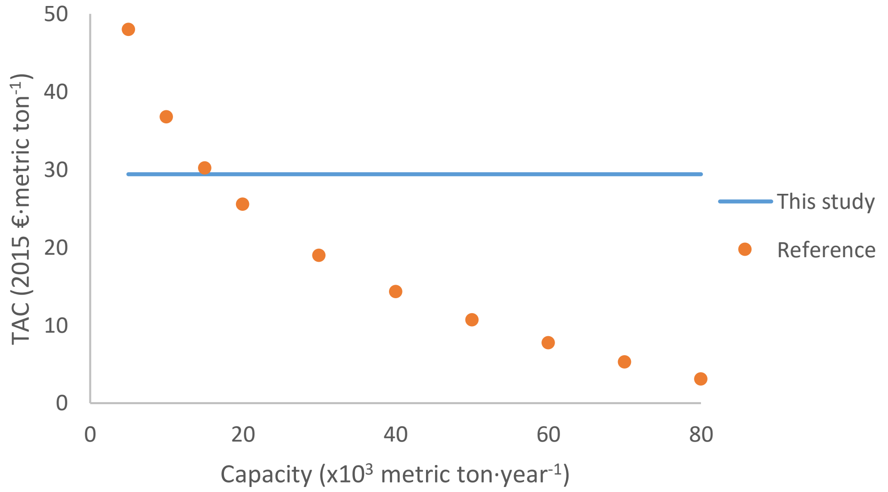

Figure S14. CC of wet anaerobic digestion

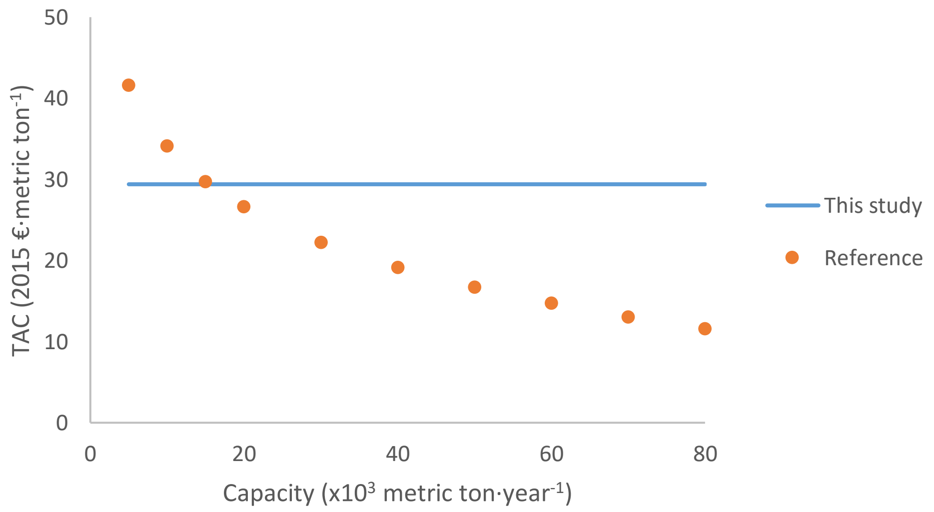

Figure S15. OM of wet anaerobic digestion

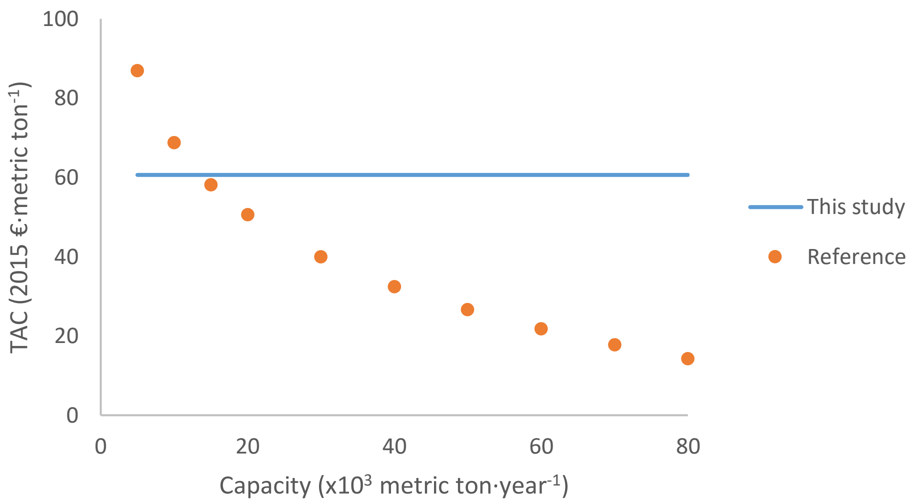

Figure S16. TAC of wet anaerobic digestion

Incineration

Table S57 shows the distribution of the TAC associated with the incineration unit process.

Table S57. TAC of incineration (2015€· metric ton^{-1})

CC	65.62
OM	53.42
elec	-11.53
TAC	107.52

The CC, OM and TAC of incineration are compared with the exponential curves obtained by Tsilemou and Panagiotakopoulos.[31] In Figure S18 the electricity revenues are excluded.

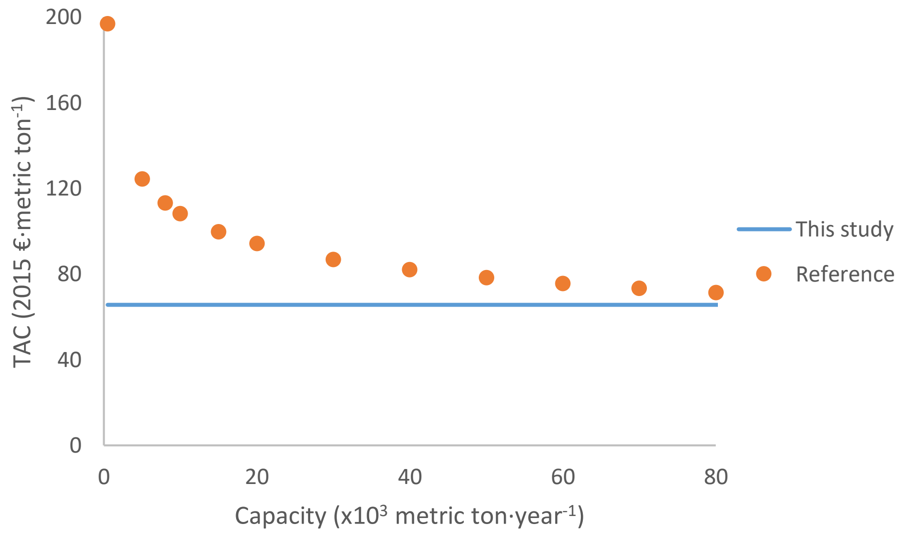

Figure S17. CC of incineration

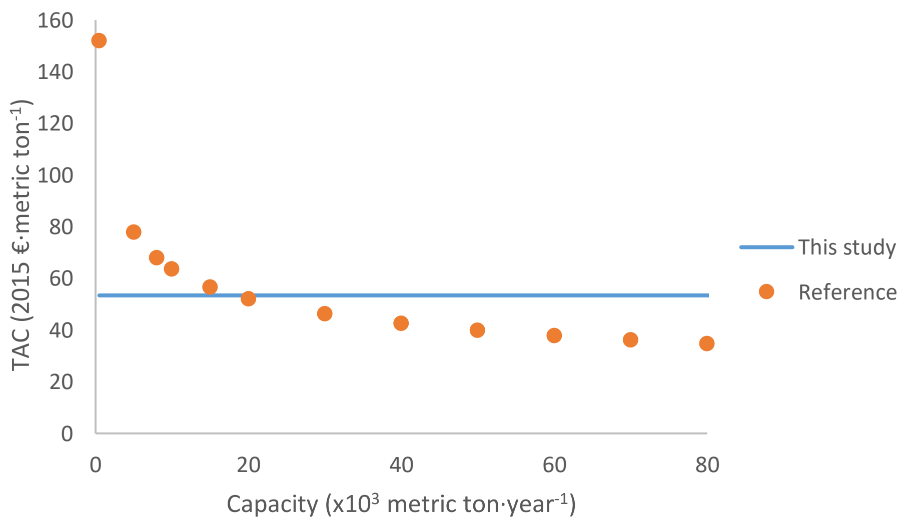

Figure S18. OM of incineration

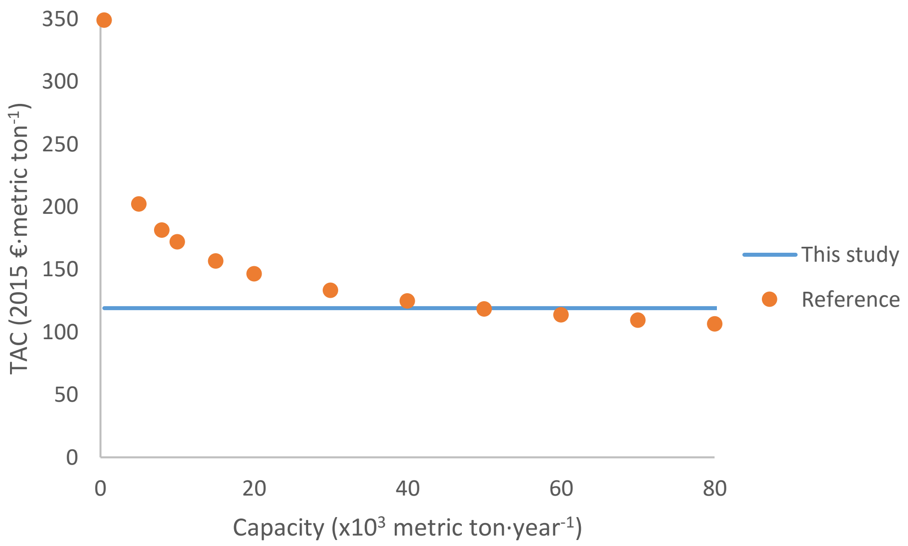

Figure S19. TAC of incineration

Landfill

The parameters required by the SWOLF landfill sub-model are the following:

- General site geometry

Landfill length/width ratio: 1

Buffer zone: 91.44 m

Fraction of buffer zone cleared/landscaped: 0.05

Height of waste above grade: 12.19 m

Slope of above grade region (rise over run): 0.33

Slope of below grade region (rise over run): 0.33

Off-site road upgrade for heavy vehicles: 1.6 km

On site roads: 182.88 m

- Basic design and economic parameters

Number of cells: 20.00

Distance between groundwater monitor wells: 152 m

Engineering rate (capital): 0.1

- Labor costs

Minimum number of laborers: 3

Overhead costs: 10% wage

- Other operating costs

Equipment and maintenance cost (excluding fuel): 2,015.6 2010$·year^{-1}·(metric ton·day^{-1})$^{-1}$

Groundwater monitoring: 2,571 2010$·well^{-1}·year^{-1}

Post-closure care: 285,369 2010$·year^{-1}

- Construction costs

Land clearing: 3,149 2010$·acre^{-1}

Low level landscaping: 1,864 2010$·acre^{-1}

High level landscaping: 5 2010$·(metric ton·day^{-1})$^{-1}$

Standard excavation: 3.36 2010$·m^{-3}

Difficult excavation (i.e., muck, clay, etc.): 5.04 2010\$·m^{-3}

On-site earth hauling: 1.91 2010\$·m^{-3}·km^{-1}

Off-site earth hauling: 0.41 2010\$·m^{-3}·km^{-1}

Distance for disposal of excess soil: 1.61 km

Berms: 4.20 2010\$·m^{-3}

Mixing and compacting soil for liners: 12.61 2010\$·m^{-3}

- Purchased material for berms, liners, and final cover

Soil for berm: 4.489 2010\$·m^{-3}

Clay for final cover or liner: 11.77 2010\$·m^{-3}

Clay additive for liner permeability: 100.88 2010\$·m^{-3}

Sand: 13.54 2010\$·m^{-3}

Gravel: 13.95 2010\$·m^{-3}

Geotextile for final cover - cost of procurement: 0.50 2010\$·m^{-2}

Geotextile for final cover - cost of installation: 0.83 2010\$·m^{-2}

HDPE for final cover - cost of procurement and installation: 15.22 2010\$·m^{-2}

Daily cover – HDPE: 15.22 2010\$·m^{-2}

Daily cover - off site soil: 9.25 2010\$·yd^{-3}

Flexible membrane liner (includes installation): 15 2010\$·m^{-2}

Leachate pump(s), piping and electrical: 10 2010\$·(metric ton·day^{-1})$^{-1}$

Leachate storage tank: 114 2010\$·(metric ton·day^{-1})$^{-1}$

- Road construction

New road construction (heavy vehicle): 149 2010\$·m^{-1}

Existing road upgrade (heavy vehicle): 74 2010\$·m^{-1}

- Buildings

Maintenance building: 302 2010\$·m^{-2}

Building area required per waste throughput: 2 m^2·(metric ton·day^{-1})$^{-1}$

Gatehouse/personnel facility: 320 2010\$·(metric ton·day^{-1})$^{-1}$

- Utilities

Electrical connection: 10 2010\$·(metric ton·day^{-1})$^{-1}$

Sanitary sewer connection and piping: 43.02 2010\$·m^{-1}

Public water connection: 10 2010\$·(metric ton·day^{-1})$^{-1}$

Well drilling and installation: 28.28 2010\$·m^{-1}

Well water connection: 48 2010\$·(metric ton·day^{-1})$^{-1}$

Gas connection: 10 2010\$·(metric ton·day^{-1})$^{-1}$

Industrial fencing: 30 2010\$·m^{-1}

Industrial truck scale (50 metric ton capacity): 89981 2010\$

PVC piping: 43 2010\$·m^{-1}

Site pre-operational studies and activities: 321,362 2010\$

- Internal combustion engine for energy recovery

Capital cost: 1586 2010\$·kW^{-1}

Operation and management cost: 24 2010\$·MWh^{-1}

Life of engine: 20 years

Engine capacity: 6.24 kW·(metric ton·day^{-1})$^{-1}$

The costs of a landfill with a capacity of 85 x10^3 metric ton·year^{-1} are summarized as follows:

Cost of gas collection and energy generation: 3.63 2015€·metric ton^{-1}

Capital cost of leachate treatment plant: 4.02 2015€·metric ton^{-1}

Equipment costs (scraper and front end loader): 0.87 2015€·metric ton^{-1}

Table S58. Landfill initial construction costs (2010$)

Land acquisistion costs	388,563
Site fencing costs	62,621
Site building / structure costs	301,738
Platform scales costs	89,981
Site utilities installation costs	6,226
Site access roads costs	256,155
Monitoring wells cost	3,879
Initial landscaping costs	3,024
Leachate pump and storage costs	40,468
Site preoperational studies and activities	321,362

Table S59. Cell construction costs (2010$·cell^{-1})

Site clearing and excavation	238,327
Berm construction costs	11,199
Liner costs (per active region)	263,078
Leachate collection piping costs	36,895

Table S60. Landfill OM costs (2010$·year^{1})

Annual labor costs	71,135
Annual equipment costs	28,976
Annual utilities costs	711
HDPE daily cover	13,115
OM of Engine	302,624

Table S61. Landfill closure costs (2010$)

Total Final Cover Cost	6,225,601
Replacing final cover costs	622,560
Perpetual care total present costs	3,828,277

Table S62. TAC of landfill (2015€·metric ton^{-1})

CC	10.94
OM	5.98
clos	3.17
elec	-0.78
OS	-0.64
tax	2.00
TAC	20.67

Unlike for the other unit processes, SWOLF does not consider that the costs of landfill vary linearly with its capacity. The CC, OM and TAC for different capacities of the landfill are compared with the exponential curves obtained by Tsilemou and Panagiotakopoulos[30] in Figures S20-S22. The costs in Figure S21 exclude the clos, elec, OS and tax.

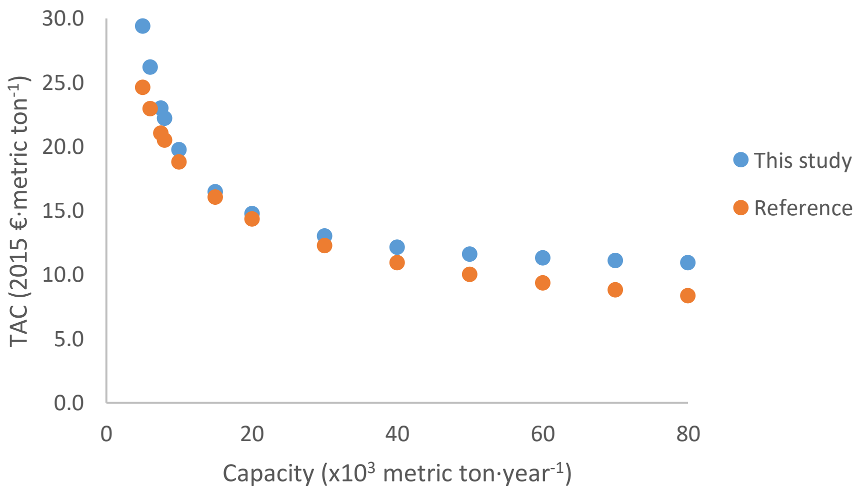

Figure S20. CC of landfill

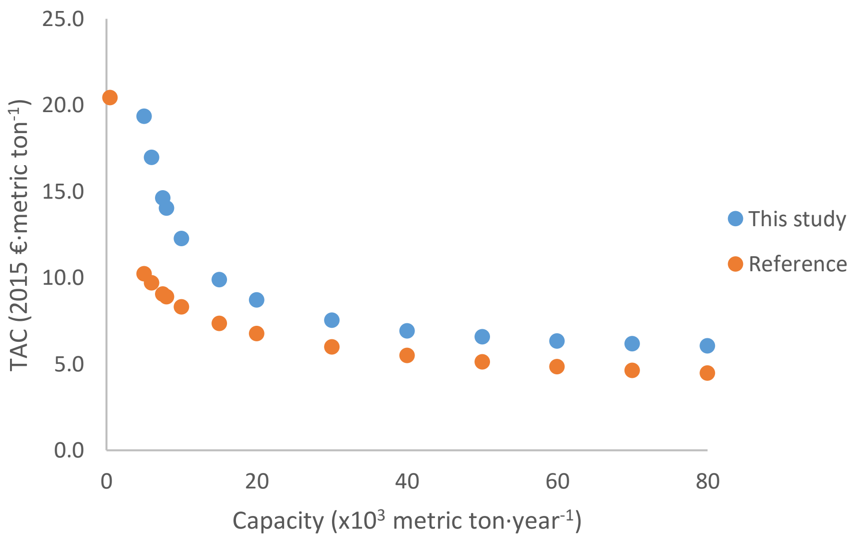

Figure S21. OM of landfill

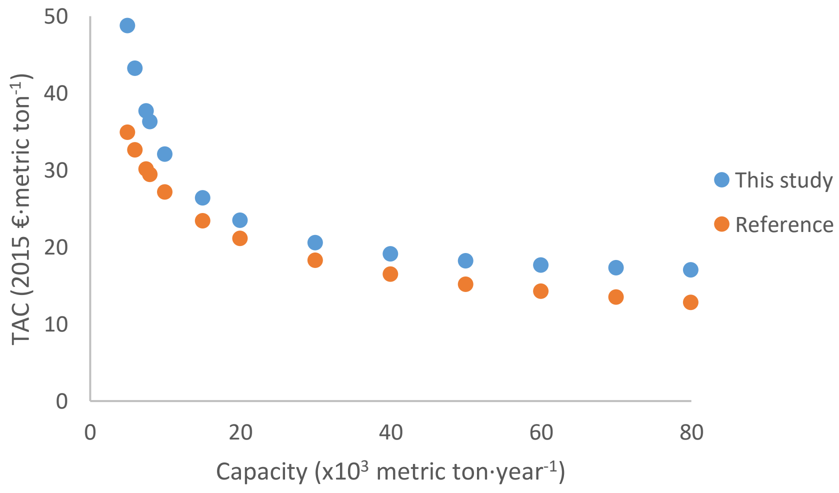

Figure S22. Sum of CC and OM of landfill

Struvite precipitation

According to Guthrie's modular method,[34] the equipment purchase costs, or base cost (BC) is calculated with an exponential equation based on its size.

Equation S2 was applied to estimate the costs of the crystallizer and stripper.

$$BC = C_0 \cdot \left(\frac{L}{L_0}\right)^{\alpha} \cdot \left(\frac{D}{D_0}\right)^{\beta} \qquad \text{(Equation S2)}$$

The sizes of the crystallizer and the stripper were estimated assuming that the vessels have identical length (L) and diameter (D) with the experimental data from Moerman et al.,[4] who required a vessel volume of 0.04 and 0.036 m^3 per $m^3 \cdot day^{-1}$ of liquid digestate for the crystallizer and stripper respectively.

The cost of the centrifugal pump was estimated with equation S3:

$$BC = C_0 \cdot \left(\frac{S}{S_0}\right)^{\alpha} \qquad \text{(Equation S3)}$$

The pump power (S) was calculated assuming a pressure drop of 50 kPa, a pump efficiency of 50% and a motor efficiency of 80%.

The parameters needed to calculate the BC of these pieces of equipment are compiled in Table S63.

Table S63. Parameters needed to calculate the BC for pressure vessels and centrifugal pumps[27]

Equipment type	C_0(1969$) Cost	S_0 (W) Power	L_0 (m) Length	D_0 (m) Diameter	α	β	MF
Horizontal vessel	690		1.22	0.91	0.78	0.98	3.18
Centrifugal pump	650	869.98			0.36		3.38

The update factor (UF) accounts for inflation. It is calculated as the ratio between the present cost index and the base cost index. The CEPCI (Chemical Engineering Plant Cost Index) from 1969 and 2015 is 115 and 556.8 respectively.[35]

$$UF = \frac{2015\ cost\ index}{1969\ cost\ index} \qquad \text{(Equation S4)}$$

The installation costs are estimated with the module factor (MF). The Materials and Pressure correction Factor (MPF) accounts for the costs associated with special materials and high pressures. It was assumed that the equipment is made of carbon steel, whose MPF is 1.
The Bare Module Costs (BMC) are calculated as follows:

$$BMC = UF \cdot BC \cdot (MPF + MF - 1) \qquad \text{(Equation S5)}$$

The BMC of the equipment required for the struvite precipitation (crystallizer, stripper and pump) were calculated for the range of liquid digestate flows that the system can handle. The high R^2 values that result from the linear regression shown in Figure S23 indicate that the linearization of the BMC is a good approximation for the studied range of flowrates.

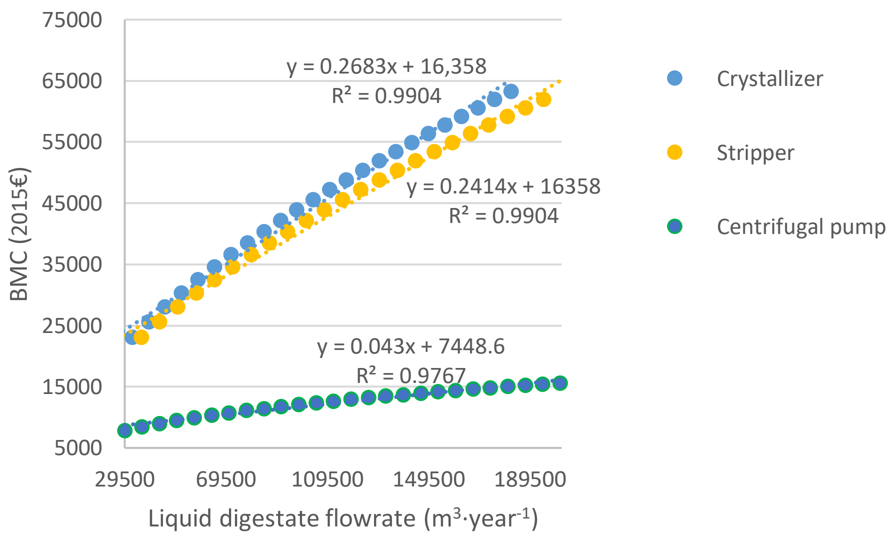

Figure S23. Linearized BMC of the equipment required for struvite precipitation

The contingency costs and indirect capital costs are estimated according to Biegler et al.[34] The fixed capital (FC) is calculated as the sum of the manufacturing (MC) and non-manufacturing capital (NMC), with equations S6-S8.

$$MC = 1.25 \cdot BMC \qquad \text{(Equation S6)}$$

$$NMC = 0.40 \cdot BMC \qquad \text{(Equation S7)}$$

$$FC = 1.65 \cdot BMC \qquad \text{(Equation S8)}$$

The working capital (WC) is estimated as 19.4% of the fixed investment:

$$WC = 0.194 \cdot FC \qquad \text{(Equation S9)}$$

The Total Capital Investment (TCI), which was later annualized, is calculated as the sum of FC and WC:

$$TCI = 1.97 \cdot BMC \qquad \text{(Equation S10)}$$

Regarding the OM of struvite precipitation, they were calculated as the price of the required electricity and MgO. The estimated operating costs are 0.0382 €·m^{-3} of liquid digestate.

Labor costs are not included in this figure because the operators are assumed to belong to the anaerobic digestion facilities. The equipment repair costs are accounted for in the FC.

Ammonia stripping and absorption

The total BMC of the equipment required for the ammonia stripping and absorption (heat exchangers, blowers, flash drum, stripper and absorber) was estimated as 31.77 €·(m^3·$year^{-1}$)$^{-1}$ with the data provided by Errico et al.[36] The TCI was calculated with equation S10.

The OM were calculated as the sum of the costs of ammonium sulfate, sodium hydroxide, electricity and natural gas required to operate the unit process. As in struvite precipitation, labor costs and the equipment repair costs are not included in the OM of ammonia stripping and absorption.

Table S64 shows the contribution of the CC and OM to the TAC.

Table S64. TAC of the ammonia stripping and absorption unit process (2015€·(m^3·$year^{-1}$)$^{-1}$)

CC	6.87
OM	4.40
TAC	11.27

Results of the single objective optimization

Figure S24 shows the contribution of the unit processes and the TAX_{WM} to the TAC that results from the minimization of the TAC and GW.

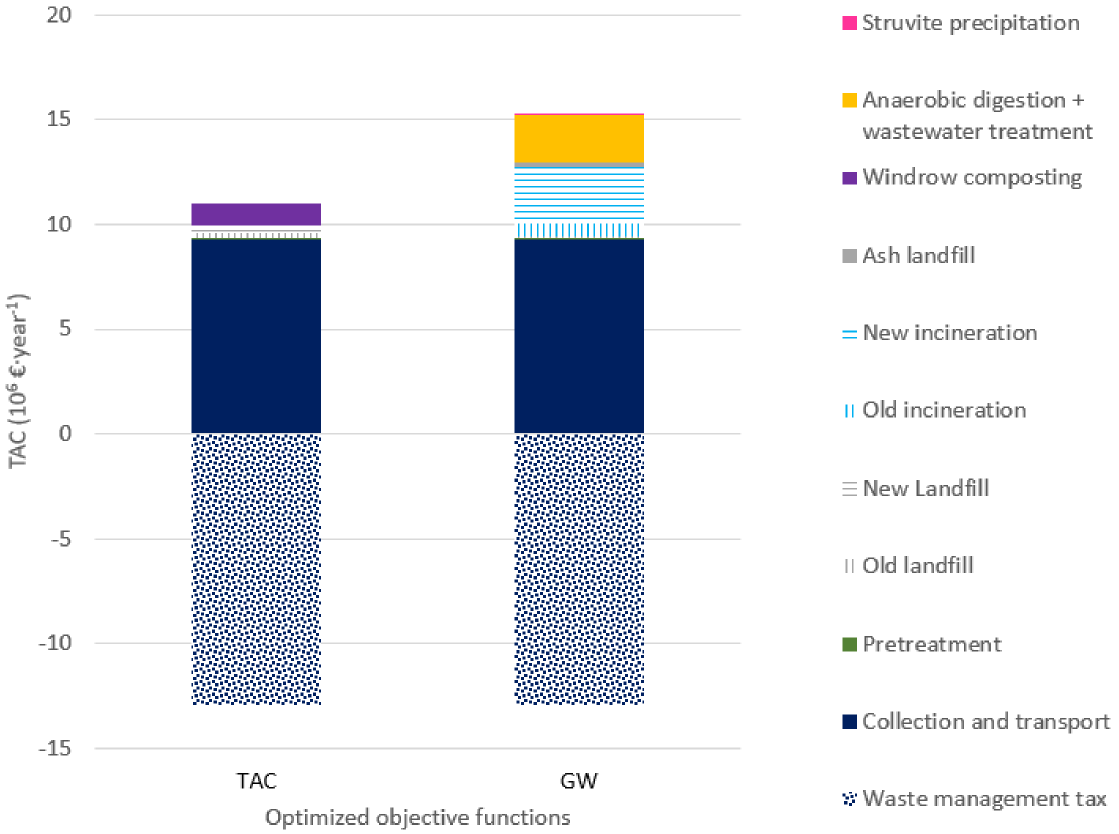

Figure S24. Contributions to the overall costs (2015€)

Price of fertilizers

The price of the fertilizing products is calculated with equations S11 and S12, and it is expressed as € per ha fertilized with the recovered products and the required complementary industrial fertilizers to grow corn.

$$P_F = \frac{TAC + P_{IND} + P_{RI}}{ha_{rec}} \leftrightarrow TAC > 0 \qquad \text{(Equation S11)}$$

$$P_F = \frac{(P_{IND} + P_{RI})}{ha_{rec}} \leftrightarrow TAC < 0 \qquad \text{(Equation S12)}$$

Change of the value of money with time

The conversion factors required to account for the change of the value of money (US$) with time were extrapolated from the data provided by the Bureau of Economic Analysis[37] and compiled in Table S65.

Table S65. Conversion factors

Year	Conversion factor to 2015$
1980	2.48
1981	2.27
1982	2.13
1983	2.05
1984	1.98
1985	1.92
1986	1.88
1987	1.84
1988	1.77
1989	1.71
1990	1.65
1991	1.59
1992	1.56
1993	1.52
1994	1.49
1995	1.46
1996	1.43
1997	1.41
1998	1.40
1999	1.37
2000	1.34
2001	1.31
2002	1.29
2003	1.27
2004	1.23
2005	1.20
2006	1.16
2007	1.13
2008	1.11
2009	1.10
2010	1.09
2011	1.06
2012	1.05
2013	1.03
2014	1.01
2015	1.00
2016	0.99
2017	0.97

APPENDIX G. HEAVY METALS CONTENT OF THE ORGANIC FERTILIZERS

The heavy metal content of the organic fertilizers recovered from the SS-OW (Table S66) is compared to the heavy metal limit values established by the Spanish legislation[38] and the new European proposal for the regulation of fertilizing products[39] (Table S67).

Table S66. Heavy metals content of the organic fertilizing products (mg·kg^{-1} of dry matter)

Product Unit process	COMP T	COMP W	SD AD
Cd	0.3964	0.4077	0.4987
Cu	27.41	28.10	32.13
Hg	0.0005	0.0025	0.3126
Ni	4.731	7.948	10.32
Pb	5.927	7.946	44.805
Zn	251.3	243.5	264.2
Hg	0.0005	0.0025	0.3126
Cr	0.5454	5.032	15.86
Cr VI	Unknown	Unknown	Unknown

Table S67. Limit of heavy metals in fertilizing products according to the legislation (mg·kg^{-1} of dry matter)

	New European proposal[40]	Spanish Royal Decree 506/2013[38]		
	Solid organic fertilizers	Type A	Type B	Type C
Cd	1.5	0.7	2	3
Cu	-	70	300	400
Hg	1	-	-	-
Ni	50	25	90	100
Pb	120	45	150	200
Zn	-	200	500	1000
Hg	-	0.4	1.5	2.5
Cr	-	70	250	300
Cr VI	2	-	-	-

NOMENCLATURE

AD	Anaerobic digestion
AD+I	Anaerobic digestion plus incineration of the rejects
AD+L	Anaerobic digestion plus landfill of the rejects
ASA	Ammonia stripping and absorption
BC	Equipment base cost
BioC	Biogenic C
$BioC_{AD}$	Anaerobically degradable biogenic carbon
$BioC_{ADi}0$	Initial amount of anaerobically degradable biogenic carbon in fraction *i* of waste
BMC	Equipment bare module cost
BS	Bio-stabilized material
C	Carbon
CC	Annualized capital costs
clos	Closure costs
COMP	Compost
DAP	Diammonium phosphate
DPC	Direct Project Costs
elec	Revenues from the sale of electricity
FC	Fixed capital
Fert	Inorganic Fertilizers
FossilC	Fossil C
FWE	Freshwater eutrophication impacts
GW	Global warming impacts
I	Incineration
Inew	New incineration
Iold	Old incineration
IPC	Installed Project Costs
L	Landfill
LCI	Life cycle inventory
Lnew	New landfill
Lold	Old landfill
MC	Manufacturing capital
MEU	Marine eutrophication impacts
MF	Module factor
mix-OW	Organic waste recovered from the mixed waste stream after the trommel separation
MPF	Material Pressure Factor
N	Nitrogen
NMC	Non-manufacturing capital
OM	Operation and management costs
OS	Operation subsidies
P	Phosphorus
P_{EQ}	Price of the industrial fertilizers that would be required to cover the same area as that fertilized by the recovered products and their complementary fertilizers

P_F	Minimum price that farmers must pay for all fertilizers to make the CIWMS economically feasible for waste managers
P_{IND}	Price of industrial fertilizers required to complement the fertilizers recovered from organic waste
P_{RI}	Price of the recovered inorganic fertilizers $(NH_4)_2SO_4$ and $(NH_4)_2HPO_4$
SD	Solid digestate
SP	Struvite precipitation
SS-OW	Source-separated organic waste
SSR	Source separation rate
T	Tunnel composting
T+I	Tunnel composting plus incineration of the rejects
T+L	Tunnel composting plus landfill of the rejects
Tax	Landfill tax
TCI	Total Capital Investment
TS	Total Solids
TWW	Total Wet Weight
UF	Update factor
VS	Volatile solids
W	Windrow composting
W+I	Windrow composting plus incineration of the rejects
W+L	Windrow composting plus landfill of the rejects
WC	Working capital
WW	Wastewater treatment

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Some of the most concerning sustainability challenges faced by humanity today are systems problems; they are deeply interconnected, and they cannot be solved separately. Thinking in systems should allow us to better understand the interconnections between their components, simulate possible future scenarios and address systems redesign based on more sustainable criteria. The management of waste and resources is one of those pressing challenges that could benefit from systems thinking.

The circular economy has been proposed as a resilient economic model to simultaneously tackle this problem and embrace sustainability. However, a systematic approach to measure, assess or optimize the performance of a circular economy has not been developed yet. This thesis contributes to filling this research gap by developing a framework that enables devising sustainable solutions to optimize integrated waste and resource management systems.

The application of the proposed framework to a case study suggests that the advantages of a circular economic model cannot be taken for granted. The successful implementation of a circular economy will require a proper balance between resource circularity and its sustainability implications.

Ultimately, there is not a unique road to sustainability, and certainly not a perfect one; any contribution to rebuild and reframe the current economic model that helps us transition toward a less resource-dependent and more sustainable production and consumption system should be encouraged.