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PhD Thesis

Impact of the pricing policy on the environmental performance of urban waste management systems

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TESE DE DOUTORAMENTO

IMPACT OF THE PRICING POLICY ON THE ENVIRONMENTAL PERFORMANCE OF URBAN WASTE MANAGEMENT SYSTEMS

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ESCOLA DE DOUTORAMENTO INTERNACIONAL DA UNIVERSIDADE DE SANTIAGO DE COMPOSTELA
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D./Dna. **Álvaro Fernández Braña**

Título da tese: *Impact of the pricing policy on the environmental performance of urban waste management systems*

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DA TESE

Impact of the pricing policy on the environmental performance of urban waste management systems

D. Gumersindo Feijoo Costa
Dna. Célia Maria Dias Ferreira

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En Santiago de Compostela, 10 de maio de 2022 | En Aveiro, 10 de maio de 2022

Gumersindo Feijoo Costa | Célia Maria Dias Ferreira

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Resumo (galego)

Esta tese de doutoramento aborda as implicacións medioambientais que se poden esperar da aplicación no ámbito da xestión de residuos urbanos (RU) de sistemas tarifarios de tipo variábel para o pagamento por parte da cidadanía deste servizo, que é habitualmente unha responsabilidade correspondente ás entidades municipais (concellos). Unha tarifa variábel consiste en atribuír un prezo unitario que é referido á unidade de medida habitual do ben ou servizo fornecido, calculándose o total do valor a ser pago pola/o utilizador/a en base ao número de unidades adquiridas. O concepto da tarifa variábel é habitual no caso do pagamento doutros servizos básicos para a poboación como poden ser o subministro de auga ou de electricidade, nos que se atribúe un prezo unitario a unidades coma o litro ou o quilovatio-hora (de auga e de electricidade, respectivamente). Na xestión de RU, a súa aplicación está bastante estendida nos países da Europa central e setentrional (Austria, Chequia, Suecia, Suíza,...) mais, agás Italia, non tanto nos países do sur da Europa – nomeadamente en Portugal, onde se desenvolveu o traballo práctico desta tese – sendo nestes países máis común a existencia de *tarifas planas* – ou sexa, de valor fixo e independente da cantidade de residuos producida – ou ben, tarifas indexadas con base noutras variábeis diferentes e independentes da propia xeración de residuos, como pode ser o consumo de auga. Dar a coñecer os potenciais beneficios ambientais desta política alternativa é un propósito desta tese que poderá contribuír para superar as barreiras aínda existentes para a extensión do seu uso efectivo nestes países.

A idea da aplicación da tarifa variábel ao pagamento do servizo de xestión de RU está actualmente a gañar interese dentro do contexto actual no que se tende a promover unha evolución cara á economía circular, tendo como horizonte teórico acadar unha sociedade de "residuo cero". Este novo paradigma esixe a incorporación da sustentabilidade – a ambiental fundamentalmente, mais tamén da económica e social – coma unha dimensión necesariamente a considerar cando se trata de avaliar a eficacia dos sistemas de xestión de RU no desempeño das súas funcións. Neste senso, considérase que a adopción dunha determinada política económica respecto do prezo que debe ser pago pola xestión dos residuos pode ter unha capacidade de influencia decisiva sobre as características da xeración dos mesmos. Noutras palabras: o que se está a propor é utilizar a tarificación do servizo de xestión de RU como medio para acadar un obxectivo determinado de desempeño ambiental. Tal propósito xustifícase pola necesidade de desligar o incremento da xeración de residuos do crecemento económico e así reducir a cantidade de RU efectivamente producida, como única vía para acadar a sustentabilidade ambiental. Esta é precisamente a principal finalidade dos sistemas de tarificación variábel baseados no pagamento por xeración, aliñándose así co principio ético xeral da responsabilidade ambiental, chamado "principio do poluídor-pagador": «quen contamina, paga».

Porén, é preciso definir antes o que se entende na práctica por sustentabilidade antes de marcar obxectivos específicos a atinxir. Isto é: estabelecer cales son as variábeis concretas nas que se entende que un sistema de xestión de RU (no ámbito dun municipio, tal como foi entendido neste traballo), debe progresar se efectivamente pretende avanzar no roteiro da sustentabilidade. Este é precisamente o propósito do primeiro traballo abordado nesta tese (no capítulo 3), no que se definiu un conxunto de seis indicadores dinámicos que avalían o progreso interanual da entidade cara á sustentabilidade – ou sexa: comparan o valor da variábel no ano de referencia co valor do ano anterior –, estando representada cadansúa das tres dimensións fundamentais (ambiental, económica e social) por dous dos seis indicadores. O conxunto completo corresponde a estes indicadores:

1. Progreso na redución de RU enviados á lixeira como destino final.
2. Progreso na redución na produción de RU.
3. Progreso cara ao equilibrio financeiro.
4. Progreso na redución de custos.
5. Progreso na accesibilidade á recolla selectiva.
6. Progreso na redución do número de queixas.

Os indicadores non toman valores absolutos, senón que sempre son expresados en forma relativa: o progreso rexistrado en relación co ano anterior. Isto terá sentido namentres o horizonte final non for atinxido, o que pode non ser viábel nalgúns dos indicadores – por exemplo: producir cero residuos ou ter cero custos – mais noutros si: acadar un equilibrio financeiro ou non recibir ningunha queixa. Unha vez obtidos os resultados individuais de cada indicador, tórnase posíbel combinalos todos para os condensar nun único valor, a modo de "índice de sustentabilidade na xestión de RU". Para esta última operación, optouse por non aplicar factor de ponderación algún, dando a todos os indicadores o mesmo peso no resultado global; mais a conveniencia de aplicar ou non algunha ponderación é unha cuestión que fica aberta para futuros desenvolvementos.

Aínda que xeneralizábel a outras entidades, o conxunto de indicadores foi concibido tendo en conta sobre todo o contexto dos municipios en Portugal, onde tódalas informacións requiridas para o cálculo de cada indicador están dispoñíbeis para o público en cada concello. Así, decidiuse experimentar a aplicación do conxunto de indicadores aos casos particulares de dous municipios portugueses A e B, dando como resultado que as políticas de xestión de RU no concello A permitíronlle avanzar cara á sustentabilidade en máis dun 10%, no entanto as do concello B fixérono recuar case un 3%. Porén, se ben o conxunto de indicadores permite en principio a comparación entre concellos por usar datos homoxéneos, resulta evidente que o resultado particular de cada municipio vai estar afectado por circunstancias específicas (características xeográficas, demográficas, etc.) que inflúen no seu resultado, de aí que cumpra tomar con cautela esa comparación e por iso é que se preferiu non identificar os municipios escollidos para non espertar susceptibilidades. Non por iso o cadro de indicadores proposto deixa de ter utilidade como ferramenta de avaliación e comunicativa.

Unha vez definidas as variábeis de relevancia en relación ao obxectivo global da sustentabilidade – especialmente a ambiental –, cómpre estabelecer cales serán as metodoloxías analíticas máis acaídas para ligar esas variábeis cos impactos ambientais por elas xeradas. No traballo desta tese déuselle especial protagonismo como ferramenta de análise do desempeño ambiental da xestión de RU á Análise de Ciclo de Vida (ACV), que na actualidade constitúe un método común e solidamente estabelecido de avaliación dos impactos ambientais de produtos e servizos. Na súa modalidade *atribucional*, a ACV, pode ser empregada para a análise descritiva en termos de impactos ambientais dun determinado conxunto de procesos, como pode ser un sistema de xestión de residuos. Porén, cando se trata de facer servir a ACV coma unha ferramenta de soporte e guía para a toma de decisións, envolvendo a comparación ambiental entre diferentes alternativas, a modalidade *consecuencial* pode resultar máis axeitada, dado que esta pon o foco da análise nas consecuencias ambientais que un determinado sistema experimenta como resultado dunha alteración na demanda no produto ou servizo en cuestión. Esta alteración pode ser provocada por algunha mudanza nos factores externos que gobernan o sistema estudado, coma por exemplo neste caso: unha modificación na política tarifaria da xestión de RU. Concretamente, nesta tese aplicouse a técnica da ACV para a avaliación ambiental dunha experiencia piloto de posta en práctica dun sistema de recolla de RU asociado a unha tarifa variábel co fin de determinar se a mudanza no sistema tarifario conduciu efectivamente a un escenario ambientalmente máis favorábel có anterior e máis próximo da sustentabilidade.

A referida experiencia piloto foi realizada dentro do ámbito do proxecto LIFE PAYT, que precisamente pretendeu promover a introdución das tarifas variábeis indexadas á xeración de RU en tres países do sur da Unión Europea (Chipre, Grecia e Portugal). Esta experiencia tivo lugar nun barrio da cidade portuguesa de Aveiro, especificamente seleccionado para este propósito. O traballo práctico de campo consistiu nunha recompilación de información relativa á xeración e composición dos RU no barrio – realizada por medio de campañas de caracterización física e pesaxes – e, paralelamente, unha caracterización sociolóxica das actitudes e opinións da poboación residente a respecto da problemática dos residuos, realizada por medio dunha enquisa á poboación residente. Estas tarefas foron realizadas por duplicado en dúas fases temporais diferentes: antes da implementación do novo sistema, para coñecer a situación de base inicial, e posteriormente despois do final da experiencia, para avaliar os resultados obtidos e os efectos observados. Os datos obtidos nos traballos de campo foron os que serviron para elaborar dúas análises de ciclo de vida: unha primeira que adoptou unha perspectiva atribucional para describir a situación ambiental inicial da xestión de RU no barrio, e unha segunda, desta volta cunha perspectiva consecuencial, que se xustifica precisamente pola intención declarada de avaliar as consecuencias ambientais derivadas da mudanza no esquema tarifario.

A análise atribucional corresponde á Sección II desta tese, que foi dividida nunha análise do sistema de recolla de RU inicialmente existente (no capítulo 4), seguida dunha análise global do sistema de xestión, con énfase na pegada de carbono, ligada a unha análise de custos (no capítulo 5).

Na análise atribucional do sistema de recolla, elaborouse un inventario de ciclo de vida contabilizando as entradas e saídas – relativas a material, enerxía e emisións poluíntes – que poden ser imputadas a tódolos elementos existentes nun sistema de recolla de RU, que para o caso do barrio analizado resúmense en:

- <u>Sacos plásticos (de polietileno)</u>: para o almacenamento e transporte inicial de residuos.
- <u>Caixotes de lixo</u>: onde colocar os sacos namentres non están cheos.
- <u>Contedores de rúa</u>: para o almacenamento dos residuos previamente a seren recollidos.
- <u>Vehículos de recolla (camións)</u>: necesarios para realizar o recolla dos residuos desde os contedores e o transporte até a instalación de tratamento.

Após a análise dos impactos ambientais asociados a estes elementos (feito coa metodoloxía ReCiPe), atopouse que os principais impactos eran os correspondentes ao consumo de gasóleo polos camións de recolla naquelas categorías de impacto ambiental máis relacionadas coa calidade do ar – acidificación terrestre, alteracións climáticas, formación de materia particulada, formación de oxidantes foto-químicos, alén da ecotoxicidade terrestre. Por contra, noutras categorías relevantes coma o consumo de recursos fósiles e a ecotoxicidade da auga doce, os maiores impactos viñan dados polo uso dos sacos de plástico. Perante este resultado, achouse que tiña interese profundar na análise do impacto dos sacos para verificar se outros materiais alternativos ao polietileno conseguirían reducir eses impactos negativos. Porén, tras experimentar comparativamente o impacto de sacos fabricados a partir de dous biopolímeros diferentes – amidón termoplástico e poli-hidroxialcanoatos – e máis polietileno producido enteiramente con material reciclado, obtívose que só este último modelo de saco era quen de mitigar significativamente os impactos, comparado cos outros. Complementariamente, tamén se analizou a influencia do deseño dos sacos no seu impacto: analizando as medidas de varios modelos de sacos de lixo de diversos tamaños, observouse que a cantidade de polietileno utilizada para fabricar un saco non varía de maneira proporcionalmente linear en relación ao tamaño deste, senón que tanto os sacos máis pequenos como os sacos maiores precisan dunha cantidade relativamente maior de material, sendo o tamaño óptimo aquel situado por volta dos 40 litros de volume.

No relativo ao consumo de combustíbel, concluíuse que, dada a grande influencia que supón no resultado global, resulta fundamental que a orixe dos datos usados para calcular a cantidade consumida na recolla sexan preferentemente obtidas directamente dos rexistros dos vehículos implicados na recolla analizada – como foi feito neste traballo –, pois calquera outra fonte de información alternativa vai resultar en estimacións que poden ficar moi afastadas da realidade que se pretende representar.

A seguir, no capítulo 5 a análise atribucional amplíouse para integrar toda a fase de tratamento e valorización dos RU, que no caso do barrio estudado ten lugar a nivel rexional por medio dunha planta de tratamento mecánico-biolóxico (TMB), destinada a recibir a fracción dos RU non separada previamente a través da recolla selectiva. Nas instalacións TMB, a fracción orgánica dos RU é separada doutros materiais para ser degradada nun proceso de dixestión anaerobia onde é producido biogás rico en metano, que serve como combustíbel para a xeración de enerxía eléctrica, namentres o resto da materia orgánica sobrante é enviada a un proceso de compostaxe. Do resto dos RU, sepáranse aqueles materiais máis valiosos para a reciclaxe (metais e algúns plásticos), que se xuntan a aqueles xa recuperados por medio da recolla selectiva (metais, papel/cartón, plásticos e vidro). A todos estes materiais reciclados, máis á enerxía renovábel obtida do biogás e ao composto, élles descontado nos seus valores de impacto ambiental un "crédito" asignado en función da cantidade de produción de materiais e enerxía primarios que contribuíron a evitar. O refugo non apto para a reciclaxe é depositado nunha lixeira controlada ou aterro, sendo os seus impactos ambientais modelados con base na composición obtida nos balances de materia elaborados a partir da composición inicial dos RU, que é coñecida polas campañas de caracterización. Da análise deste conxunto de procesos, concluíuse que, con diferenza, son as emisións poluíntes causadas pola lixeira o maior impacto ambiental rexistrado, confirmando o protagonismo dado a esta forma de tratamento nos indicadores de sustentabilidade desenvolvidos no capítulo 3. Isto tamén é así no caso da pegada de carbono, que pola súa relevancia actual foi a categoría seleccionada para xerar unha serie de escenarios alternativos, nos que se tratou de descubrir cales serían os parámetros a mellorar dentro da instalación TMB para acadar unha pegada de carbono, se non neutra, polo menos o máis próxima posíbel da neutralidade. Finalmente, esta neutralidade carbónica foi atinxida no máis ideal dos escenarios teóricos proxectados, no que a separación en orixe dos materiais reciclábeis sería a máxima posíbel, a separación da fracción orgánica na TMB sería tamén a mellor posíbel e facilitaría un rendemento óptimo da produción de biogás e composto, e por último habería polo menos parte do refugo que aínda sería aproveitado en forma de combustíbel derivado de residuos (CDR), apto por exemplo para a industria do cemento.

No relativo á análise de custos, tense que, partindo da situación deficitaria atopada no intre da primeira análise, os sucesivos escenarios de melloría proxectados permiten mellorar o balance global entre custos e beneficios da xestión de RU, mais neste caso sen chegar a acadar o equilibrio – aínda que non por moita diferenza. En calquera caso, é preciso lembrar que tampouco o equilibrio entre os beneficios e custos propios da xestión de RU é unha prioridade desta actividade, sendo cuberta a diferenza precisamente grazas ao valor pago pola cidadanía na tarifa de residuos.

Como xa foi dito anteriormente, no capítulo 6 a análise aplicada ao caso de estudo adopta unha perspectiva diferente e complementaria da anterior atribucional, pasando agora a ser consecuencial. Esta abordaxe permitiu a avaliación dos efectos que sobre o impacto ambiental dos RU poden provocar as alteracións no tarifario dende un punto de vista máis amplo, non limitado só aos efectos directamente provocados pola xestión de RU, mais abranguendo tamén outros sistemas externos a esta, porén cos cales esta indirectamente interactúa. En relación con esta implicación, modificouse, relativamente aos capítulos 4 e 5, a forma en que son contabilizados na ACV os beneficios derivados da substitución de fontes de enerxía primaria e materias primas extraídas da natureza pola enerxía renovábel e os materiais reciclados obtidos a partir da valorización de RU, para incluír agora tamén os procesos de fondo que serán marxinalmente afectados polas alteracións experimentadas nos produtos de saída da xestión de RU, nomeadamente:

- Electricidade a partir de biogás: substitúe a combinación de fontes de electricidade prevista en Portugal no período 2020 – 2030.
- Composto: substitúe o esterco coma axente estruturante do solo, que en alternativa pode ser tratado por dixestión anaerobia para obter biogás, que por súa vez substitúe gas natural como combustíbel para xerar electricidade e calor.
- Papel reciclado: substitúe papel non reciclado, o que deixa dispoñíbel biomasa forestal para ser utilizada en alternativa como combustíbel para xerar electricidade e calor, substituíndo por súa vez gas natural.
- Metais, plásticos e vidro reciclados: substitúen os seus respectivos equivalentes non reciclados.

A intención primeira foi empregar esta análise consecuencial para estabelecer unha comparación entre os dous momentos inicial e final da experiencia piloto coa tarifa variábel. Os datos de campo obtidos nas campañas amosaron unha redución global de xeración de RU superior ao 17%, dentro da cal os RU mesturados sen separar (fracción “resto”) diminuíron máis dun 30%, namentres os RU encamiñados á recolla selectiva aumentaron nun 48%. Na tradución destes números nos impactos ambientais correspondentes por medio da ACV, o resultado foi que, sendo o impacto da deposición na lixeira tan dominante sobre os demais – especialmente nas categorías de alteracións climáticas, ecotoxicidade para a auga doce e toxicidade para a humanidade –, calquera diminución na produción de RU tal como a observada, vai resultar nunha redución neta do impacto global do sistema, se ben que o impacto noutras categorías menos dependentes da lixeira aumenta, por causa do maior esforzo requirido para unha recolla selectiva en expansión e tamén polo impacto dos propios procesos de reciclaxe, que o beneficio descontado pola substitución de materiais e enerxía non chega a compensar de todo.

Porén, resultou ser do máximo interese aproveitar a potencialidade da ACV consecuencial en representar a variación dos impactos ambientais no longo prazo para tentar construír un modelo que permitise dalgunha maneira predicir a evolución da xeración de RU en función do comportamento da poboación afectada pola mudanza na tarifa. Para isto fíxose uso da metodoloxía de modelización baseada en axentes (MBA). Esta técnica considera a poboación envolvida na experiencia como un conxunto de axentes, caracterizados por unha serie de propiedades distintivas, que en función de variábeis que lles son internamente atribuídas, son capaces de adaptar o seu comportamento por medio da toma de decisións – supostamente racionais – cando se lles somete a unha alteración das circunstancias nas que actúan.

Para este caso de estudo, construíuse nunha plataforma informática de MBA unha poboación de axentes que reproduce as características demográficas do barrio analizado: os axentes equivalen a familias (entendidas como comunidade que habita unha vivenda), tendo en conta se teñen membros menores de 15 anos ou maiores de 65 e se a vivenda é un apartamento ou unha casa isolada con xardín. Para trataren dos seus residuos, os axentes escollen entre catro niveis de implicación, sendo o primeiro non preocuparse nada en relación aos residuos, e consistindo os outros tres en adoptaren acumulativamente prácticas cada vez máis sustentábeis, nomeadamente:

- Separar materias reciclábeis para a recolla selectiva.
- Modificar os hábitos de consumo para reducir a produción de residuos: ben adquirindo menos produtos ou pasando a adquirir produtos máis sustentábeis (exemplo: con menos embalaxe).
- Practicar a compostaxe doméstica da fracción orgánica: axeitada para tratar os residuos alimentares e de actividades agrícolas ou xardinaría.

Os axentes escollen cal será o seu nivel de implicación en función das súas características demográficas e con base no valor (atribuído aleatoriamente) que adopten dúas variábeis internas que cada axente posúe, representando respectivamente a «forza de vontade», requirida para adoptar prácticas máis sustentábeis e máis a «sensibilidade perante o prezo», que representa a dispoñibilidade para pagar máis ou menos pola xestión de RU. É claro que unicamente a segunda variábel será directamente afectada por unha mudanza no sistema tarifario, mais iso non diminúe a importancia da outra. De feito, diversos estudos publicados demostraron xa como a preocupación co prezo en si non é o único factor que inflúe na maior ou menor implicación persoal cunha xestión de residuos e un estilo de vida máis sustentábeis – até porque o valor a pagar na factura do lixo nunca será tan considerábel coma o pago por outros servizos básicos –, senón que hai outros factores – como a consciencia ambiental ou o sentido cívico e de responsabilidade coa comunidade – que son igual ou até máis importantes na decisión a tomar. A respecto disto, debe ser lembrado que unha das chaves para explicar o éxito das tarifas variábeis de RU é a percepción de que son unha forma máis xusta de distribuír os custos, pois recompensa a quen tenta producir menos residuos e penaliza a quen non o fai – de novo, o principio do poluídor-pagador.

Porén, atendendo ás respostas dadas pola poboación nas enquisas, considerouse que non tódolos axentes reaccionan do mesmo xeito ás alteracións nas variábeis. Para alén das diferenzas demográficas, propúxose unha distribución dos axentes en catro perfís de personalidade:

- Insensíbeis: non reaccionan a ningunha das dúas variábeis (ou sexa, non teñen disposición a mudar nada).
- Conscientes: actúan con base na súa consciencia, sexa esta dirixida por unha preocupación polo medio ambiente ou por un sentido cívico de responsabilidade. Son sensíbeis á variábel da forza de vontade.
- Aforradores: actúan con base na preocupación polo diñeiro que gastan. Son sensíbeis á variábel da preocupación polo prezo.
- Sensíbeis a todo: combinación das dúas anteriores. Actúan atendendo ás dúas variábeis de forza de vontade e preocupación polo prezo.

Existe, aínda, unha terceira variábel a influenciar a toma de decisións: a presión social – aquela que inflúe na toma de decisións non por unha convicción moral senón co argumento de que «é o que fai o resto» e o desexo por evitar conflitos. Neste modelo representouse a presión social coma unha posibilidade aleatoria de que os axentes situados nas actitudes máis comprometidas coa sustentabilidade poidan persuadir aos menos comprometidos para mudaren o seu comportamento.

Finalmente, o modelo así configurado foi calibrado cos datos de campo obtidos da análise das enquisas feitas á poboación do barrio. Este paso posibilitou o uso posterior do modelo para xerar hipotéticos escenarios futuros, alterando a presión exercida pola tarifa sobre os axentes. Os resultados amosan que, con base na información real recollida na experiencia piloto, se no inicio da mesma a grande maioría dos axentes – ou sexa, as familias do barrio – situábanse no grupo unicamente comprometido coa separación de materiais para a recolla selectiva (66% dos axentes), após a fin da experiencia formaban unha leve maioría aqueles que ademais tentaron reducir o seu consumo (54% dos axentes). Esta maioría viuse ampliada no escenario –este, xa hipotético – de que a presión exercida pola tarifa fose aumentada nun factor de 1,5: daquela, os axentes que optarían por reducir o seu consumo somarían o 85%. Traducido a impactos ambientais por medio da ACV consecuencial, este último resultado supón unha diminución do impacto global grazas á redución dos residuos depositados na lixeira, mais non así – como xa explicado anteriormente – naqueles impactos referidos á recolla e aos procesos de reciclaxe.

Os outros dous grupos, máis extremos, foron sempre minoritarios, mais o interese en descubrir canto sería preciso aumentar a presión tarifaria para que máis axentes/familias escollesen practicar a compostaxe doméstica, proxectouse un novo escenario coa tarifa multiplicada por 2, e se ben nesta situación o nivel comprometido até o punto de reducir o consumo seguiu a ser o maioritario – agora co 65% dos axentes –, o seguinte nivel comprometido ademais coa compostaxe doméstica acadou a segunda posición co 31% dos axentes. Porén, o impacto ambiental correspondente amosou un resultado un tanto paradoxal: se ben os impactos ambientais diminuíron aínda máis en relación ao escenario anterior, houbo categorías – como a da contribución para as alteracións climáticas – nas que esta diminución non foi moi relevante. Isto pode ser explicado porque a retirada na composición dos RU da fracción orgánica, agora desviada cara á compostaxe doméstica, prexudicou a obtención de enerxía renovábel que con ela tiña lugar na instalación TMB, na lixeira e nos procesos secundarios derivados. Isto quere dicir que, chegado un certo punto de aumento da presión tarifaria, o esforzo requirido para mellorar aínda máis a avaliación ambiental comeza a ser proporcionalmente maior ca antes. Dito doutro xeito: a medida que o avance cara á sustentabilidade vai progresando máis, a marxe de melloría dispoñible para conseguir novos avances vai ser menor, así como o esforzo para obter eses novos avances será comparativamente maior. Unha conclusión que serve para mellor ponderar o contributo das tarifas variábeis nesa progresión cara á sustentabilidade da xestión de residuos, servindo como fechamento final do traballo desenvolvido nesta tese.

Resumo (português)

Esta tese de doutoramento aborda as implicações ambientais que podem esperar-se da aplicação no âmbito da gestão de resíduos urbanos (RU) de sistemas tarifários de tipo variável para o pagamento por parte da cidadania deste serviço, que é habitualmente uma responsabilidade correspondente às entidades municipais (concelhos). Uma tarifa variável consiste em atribuir um preço unitário que é referido à unidade de medida habitual do bem ou serviço fornecido, calculando-se o total do valor a ser pago pela/o utilizador/a com base no número de unidades adquiridas. O conceito da tarifa variável é habitual no caso do pagamento doutros serviços básicos para a população como podem ser o subministro de água ou de eletricidade, nos que se atribui um preço unitário a unidades como o litro ou o quilovátio-hora (de água e de eletricidade, respetivamente). Na gestão de RU, a sua aplicação está bastante estendida nos países da Europa central e setentrional (Áustria, Chéquia, Suécia, Suíça,...) mas, exceto a Itália, não assim tanto nos países do sul da Europa – nomeadamente em Portugal, onde se desenvolveu o trabalho prático desta tese – sendo nestes países mais comum a existência de, quer *tarifas planas* – ou seja, de valor fixo e independente da quantidade de resíduos produzida – quer tarifas indexadas com base noutras variáveis diferentes e independentes da própria produção de resíduos, como pode ser o consumo de água. Dar a conhecer os potenciais benefícios ambientais desta política alternativa é um propósito desta tese que poderá contribuir para superar as barreiras ainda existentes para a extensão do seu uso efetivo nestes países.

A ideia da aplicação da tarifa variável ao pagamento do serviço de gestão de RU está atualmente a ganhar interesse dentro do contexto atual no que se tende a promover uma evolução em direção à economia circular, tendo como horizonte teórico atingir uma sociedade de “resíduo zero”. Este novo paradigma exige a incorporação da sustentabilidade – a ambiental fundamentalmente, mas também da económica e social – como uma dimensão necessariamente a considerar quando se trata de avaliar a eficácia dos sistemas de gestão de RU no desempenho das suas funções. Neste sentido, considera-se que a adoção de uma determinada política económica a respeito do preço que deve ser pago pela gestão dos resíduos pode ter uma capacidade de influência decisiva sobre as características da produção dos mesmos. Noutras palavras: o que se está a propor é utilizar a tarifação do serviço de gestão de RU como meio para conseguir um objetivo determinado de desempenho ambiental. Tal propósito justifica-se pela necessidade de desligar o incremento da produção de resíduos do crescimento económico e assim reduzir a quantidade de RU efetivamente produzida, como única via para alcançar a sustentabilidade ambiental. Esta é precisamente a principal finalidade dos sistemas de tarifação variável baseados no pagamento por produção, alinhando assim com o princípio ético geral da responsabilidade ambiental, chamado “principio do poluidor-pagador”: «quem polui, paga».

Porém, é preciso definir antes o que se entende na prática por sustentabilidade antes de marcar objetivos específicos a atingir. Isto é: estabelecer quais são as variáveis concretas nas que se entende que um sistema de gestão de RU (no âmbito dum município, tal como foi entendido neste trabalho), deve progredir se efetivamente pretende avançar no roteiro da sustentabilidade. Este é precisamente o propósito do primeiro trabalho abordado nesta tese (no capítulo 3), no que se definiu um conjunto de seis indicadores dinâmicos que avaliam o progresso interanual da entidade em direção à sustentabilidade – ou seja: comparam o valor da variável no ano de referência com o valor do ano anterior –, estando representada cada uma das três dimensões fundamentais (ambiental, económica e social) por dois dos seis indicadores. O conjunto completo corresponde a estes indicadores:

1. Progresso na redução de RU enviados ao aterro como destino final.
2. Progresso na redução na produção de RU.
3. Progresso em direção ao equilíbrio financeiro.
4. Progresso na redução de custos.
5. Progresso na acessibilidade à recolha seletiva.
6. Progresso na redução do número de queixas.

Os indicadores não tomam valores absolutos, senão que são sempre expressados em forma relativa: o progresso registado em relação com o ano anterior. Isto fará sentido, entretanto o horizonte final não for atingido, o que pode não ser viável nalguns dos indicadores – por exemplo: produzir zero resíduos ou ter zero custos – mas noutros sim: conseguir um equilíbrio financeiro ou não receber nenhuma queixa. Uma vez obtidos os resultados individuais de cada indicador, torna-se possível combinar todos para os condensar num único valor, a modo de "índice de sustentabilidade na gestão de RU". Para esta última operação, optou-se por não aplicar fator de ponderação algum, dando a todos os indicadores o mesmo peso no resultado global; mas a conveniência de aplicar ou não alguma ponderação é uma questão que fica aberta para futuros desenvolvimentos.

Mesmo que generalizável a outras entidades, o conjunto de indicadores foi concebido tendo em conta sobre todo o contexto dos municípios em Portugal, onde todas as informações requeridas para o cálculo de cada indicador estão disponíveis para o público em cada concelho. Assim, decidiu-se experimentar a aplicação do conjunto de indicadores aos casos particulares de dois municípios portugueses A e B, dando como resultado que as políticas de gestão de RU no concelho A permitiram-lhe avançar em direção à sustentabilidade em mais dum 10%, no entanto as do concelho B fizeram que recuasse quase um 3%. Porém, se bem o conjunto de indicadores permite em principio a comparação entre concelhos por usar dados homogéneos, resulta evidente que o resultado particular de cada município vai estar afetado por circunstâncias específicas (características geográficas, demográficas, etc.) que influem no seu resultado, de aí que seja preciso tomar com cautela essa comparação e por isso é que se preferiu não identificar os municípios escolhidos para não espertar suscetibilidades. Não por isso o quadro de indicadores proposto deixa de ter utilidade como ferramenta de avaliação e comunicativa.

Uma vez definidas as variáveis de relevância em relação ao objetivo global da sustentabilidade – especialmente a ambiental –, é preciso estabelecer quais serão as metodologias analíticas mais adequadas para ligar essas variáveis com os impactos ambientais por elas geradas. No trabalho desta tese foi dado especial protagonismo como ferramenta de análise do desempenho ambiental da gestão de RU à Análise de Ciclo de Vida (ACV), que na atualidade constitui um método comum e solidamente estabelecido de avaliação dos impactos ambientais de produtos e serviços. Na sua modalidade *atribucional*, a ACV, pode ser empregada para a análise descritiva em termos de impactos ambientais dum determinado conjunto de processos, como pode ser um sistema de gestão de resíduos. Porém, quando se trata de fazer servir a ACV como uma ferramenta de suporte e guia para a toma de decisões, envolvendo a comparação ambiental entre diferentes alternativas, a modalidade *consequencial* pode resultar mais adequada, dado que esta põe o foco da análise nas consequências ambientais que um determinado sistema experimenta como resultado de uma alteração na demanda no produto ou serviço em questão. Esta alteração pode ser provocada por alguma mudança nos fatores externos que governam o sistema estudado, como por exemplo neste caso: uma modificação na política tarifária da gestão de RU. Concretamente, nesta tese aplicou-se a técnica da ACV para a avaliação ambiental de uma experiência piloto de posta em prática dum sistema de recolha de RU associado a uma tarifa variável com o fim de determinar se a mudança no sistema tarifário conduziu efetivamente a um cenário ambientalmente mais favorável do que o anterior e mais próximo da sustentabilidade.

A referida experiência piloto foi realizada dentro do âmbito do projeto LIFE PAYT, que precisamente pretendeu promover a introdução das tarifas variáveis indexadas à produção de RU em três países do sul da União Europeia (Chipre, Grécia e Portugal). Esta experiência teve lugar num bairro da cidade portuguesa de Aveiro, especificamente selecionado para este propósito. O trabalho prático de campo consistiu numa compilação de informação relativa à produção e composição dos RU no bairro – realizada com recurso a campanhas de caracterização física e pesagens – e, paralelamente, uma caracterização sociológica das atitudes e opiniões da população residente a respeito da problemática dos resíduos, realizada por meio dum inquérito à população residente. Estas tarefas foram realizadas por duplicado em duas fases temporais diferentes: antes da implementação do novo sistema, para conhecer a situação de base inicial, e posteriormente após o final da experiência, para avaliar os resultados obtidos e os efeitos observados. Os dados obtidos nos trabalhos de campo foram os que serviram para elaborar duas análises de ciclo de vida: uma primeira que adotou uma perspetiva atribucional para descrever a situação ambiental inicial da gestão de RU no bairro, e uma segunda, agora sob uma perspetiva consequencial, que se justifica precisamente pela intenção declarada de avaliar as consequências ambientais derivadas da mudança no sistema tarifário.

A análise atribucional corresponde à Secção II desta tese, que foi dividida numa análise do sistema de recolha de RU inicialmente existente (no capítulo 4), seguida duma análise global do sistema de gestão, com ênfase na pegada de carbono, ligada a uma análise de custos (no capítulo 5).

Na análise atribucional do sistema de recolha, elaborou-se um inventário de ciclo de vida contabilizando as entradas e saídas – relativas a material, energia e emissões poluentes – que podem ser imputadas a todos os elementos existentes num sistema de recolha de RU, que para o caso do bairro analisado resumem-se em:

- <u>Sacos plásticos (de polietileno)</u>: para o armazenamento e transporte inicial de resíduos.
- <u>Caixotes de lixo</u>: onde colocar os sacos entanto não estão cheios.
- <u>Contentores de rua</u>: para o armazenamento dos resíduos previamente a serem recolhidos.
- <u>Veículos de recolha (camiões)</u>: necessários para realizar a recolha dos resíduos desde os contentores e o transporte até a instalação de tratamento.

Após a análise dos impactos ambientais associados a estes elementos (feito com a metodologia ReCiPe), encontrou-se que os principais impactos eram os correspondentes ao consumo de gasóleo pelos camiões de recolha naquelas categorias de impacto ambiental mais relacionadas com a qualidade do ar – acidificação terrestre, alterações climáticas, formação de matéria particulada, formação de oxidantes fotoquímicos, além da ecotoxicidade terrestre. Pelo contrário, noutras categorias relevantes, como o consumo de recursos fósseis e a ecotoxicidade da água doce, os maiores impactos vinham dados pelo uso dos sacos de plástico. Perante este resultado, achou-se que tinha interesse profundar na análise do impacto dos sacos para verificar se outros materiais alternativos ao polietileno conseguiriam reduzir esses impactos negativos. Porém, após experimentar comparativamente o impacto de sacos fabricados a partir de dois biopolímeros diferentes – amido termoplástico e poli-hidroxialcanoatos – e também polietileno produzido inteiramente com material reciclado, obteve-se que apenas este último modelo de saco era capaz de mitigar significativamente os impactos, comparado com os outros. Complementariamente, também foi analisada a influência da construção dos sacos no seu impacto: analisando as medidas de vários modelos de sacos de lixo de diversos tamanhos, observou-se que a quantidade de polietileno utilizada para fabricar um saco não varia de maneira proporcionalmente linear em relação ao tamanho deste, senão que tanto os sacos mais pequenos como os sacos maiores precisam duma quantidade relativamente maior de material, sendo o tamanho ótimo aquele situado por volta dos 40 litros de volume.

No relativo ao consumo de combustível, concluiu-se que, dada a grande influência que supõe no resultado global, resulta fundamental que a origem dos dados usados para calcular a quantidade consumida na recolha seja preferentemente obtida diretamente dos registos dos veículos implicados na recolha analisada – como foi feito neste trabalho –, pois qualquer outra fonte de informação alternativa vai resultar em estimações que podem ficar muito afastadas da realidade que se pretende representar.

A seguir, no capítulo 5 a análise atribucional ampliou-se para integrar toda a fase de tratamento e valorização dos RU, que no caso do bairro estudado tem lugar a nível regional por meio de uma planta de tratamento mecânico-biológico (TMB), destinada a receber a fração dos RU não separada previamente a través da recolha seletiva. Nas instalações TMB, a fração orgânica dos RU é separada doutros materiais para ser degradada num processo de digestão anaeróbia onde é produzido biogás rico em metano, que serve como combustível para a produção de energia elétrica, entanto o resto da matéria orgânica sobrante é enviada a um processo de compostagem. Do resto dos RU, separam-se aqueles materiais mais valiosos para a reciclagem (metais e alguns plásticos), que se juntam a aqueles já recuperados por meio da recolha seletiva (metais, papel/cartão, plásticos e vidro). A todos estes materiais reciclados, mais à energia renovável obtida do biogás e ao composto, é-lhes descontado nos seus valores de impacto ambiental um “crédito” assignado em função da quantidade de produção de materiais e energia primários que contribuíram a evitar. O refugo não apto para a reciclagem é depositado numa lixeira controlada ou aterro, sendo os seus impactos ambientais modelados com base na composição obtida nos balanços de matéria elaborados a partir da composição inicial dos RU, que é conhecida pelas campanhas de caracterização. Da análise deste conjunto de processos, concluiu-se que, com diferença, são as emissões poluentes causadas pelo aterro o maior impacto ambiental registado, confirmando o protagonismo dado a esta forma de tratamento nos indicadores de sustentabilidade desenvolvidos no capítulo 3. Isto também é assim no caso da pegada de carbono, que pela sua relevância atual foi a categoria selecionada para gerar uma série de cenários alternativos, nos que se tratou de descobrir quais seriam os parâmetros a melhorar dentro da instalação TMB para alcançar uma pegada de carbono, se não neutra, pelo menos o mais próxima possível da neutralidade. Finalmente, esta neutralidade carbónica foi atingida no mais ideal dos cenários teóricos projetados, no que a separação em origem dos materiais recicláveis seria a máxima possível, a separação da fração orgânica na TMB seria também a melhor possível e facilitaria um rendimento ótimo da produção de biogás e composto, e por último haveria pelo menos parte do refugo que ainda seria aproveitado em forma de combustível derivado de resíduos (CDR), apto por exemplo para a indústria do cimento.

No relativo à análise de custos, tem-se que, partindo da situação deficitária encontrada no momento da primeira análise, os sucessivos cenários de melhoria projetados permitem melhorar o balanço global entre custos e benefícios da gestão de RU, mas neste caso sem chegar a atingir o equilíbrio – ainda que não por muita diferença. Em qualquer caso, é preciso lembrar que tão-pouco o equilíbrio entre os benefícios e custos próprios da gestão de RU é uma prioridade desta atividade, sendo coberta a diferença precisamente graças ao valor pago pela cidadania na tarifa de resíduos.

Como já foi dito anteriormente, no capítulo 6 a análise aplicada ao caso de estudo adota uma perspetiva diferente e complementária da anterior atribucional, passando agora a ser consequencial. Esta abordagem permitiu a avaliação dos efeitos que sobre o impacto ambiental dos RU podem provocar as alterações no tarifário desde um ponto de vista mais amplo, não limitado só aos efeitos diretamente provocados pela gestão de RU, mas abrangendo também outros sistemas externos a esta, porém com os quais esta indiretamente interage. Em relação com esta implicação, modificou-se, relativamente aos capítulos 4 e 5, a forma em que são contabilizados na ACV os benefícios derivados da substituição de fontes de energia primária e matérias primas extraídas da natureza pela energia renovável e os materiais reciclados obtidos a partir da valorização de RU, para incluir agora também os processos de fundo que serão marginalmente afetados pelas alterações experimentadas nos produtos de saída da gestão de RU, nomeadamente:

- Eletricidade a partir de biogás: substitui a combinação de fontes de eletricidade prevista em Portugal no período 2020 – 2030.
- Composto: substitui o estrume como agente estruturante do solo, que em alternativa pode ser tratado por digestão anaeróbia para obter biogás, que por sua vez substitui gás natural como combustível para produzir eletricidade e calor.
- Papel reciclado: substitui papel não reciclado, o que deixa disponível biomassa florestal para ser utilizada em alternativa como combustível para produzir eletricidade e calor, substituindo por sua vez gás natural.
- Metais, plásticos e vidro reciclados: substituem os seus respetivos equivalentes não reciclados.

A intenção primeira foi empregar esta análise consequencial para estabelecer uma comparação entre os dois momentos inicial e final da experiência piloto com a tarifa variável. Os dados de campo obtidos nas campanhas mostraram uma redução global da produção de RU superior ao 17%, dentro da qual os RU misturados sen separar (fração “resto”) diminuíram mais dum 30%, no entanto os RU encaminhados à recolha seletiva aumentaram num 48%. Na tradução destes números nos impactos ambientais correspondentes por meio da ACV, o resultado foi que, sendo o impacto da deposição no aterro tão dominante sobre os demais – especialmente nas categorias de alterações climáticas, ecotoxicidade para a água doce e toxicidade para a humanidade –, qualquer diminuição na produção de RU tal como a observada, vai resultar numa redução neta do impacto global do sistema, se bem que o impacto noutras categorias menos dependentes do aterro aumenta, por causa do maior esforço requerido para uma recolha seletiva em expansão e também pelo impacto dos próprios processos de reciclagem, que o benefício descontado pela substituição de materiais e energia não chega a compensar totalmente.

Porém, resultou ser do máximo interesse aproveitar a potencialidade da ACV consequencial em representar a variação dos impactos ambientais no longo prazo para tentar construir um modelo que permitisse dalguma maneira predizer a evolução da produção de RU em função do comportamento da população afetada pela mudança na tarifa. Para isto foi feito uso da metodologia de modelização baseada em agentes (MBA). Esta técnica considera a população envolvida na experiência como um conjunto de agentes, caracterizados por uma serie de propriedades distintivas, que em função de variáveis que lhes são internamente atribuídas, são capazes de adaptar o seu comportamento por meio da toma de decisões – supostamente racionais – cando se lhes submete a uma alteração das circunstâncias nas que atuam.

Para este caso de estudo, construiu-se numa plataforma informática de MBA uma população de agentes que reproduze as características demográficas do bairro analisado: os agentes equivalem a famílias (entendidas como comunidade que habita uma morada), tendo em conta se têm membros menores de 15 anos ou maiores de 65 e se a habitação é um apartamento ou uma casa isolada com jardim. Para tratarem dos seus resíduos, os agentes escolhem entre quatro níveis de implicação, sendo o primeiro não se preocupar nada em relação aos resíduos, e consistindo os outros três em adotarem acumulativamente práticas cada vez mais sustentáveis, nomeadamente:

- Separar matérias recicláveis para a recolha seletiva.
- Modificar os hábitos de consumo para reduzir a produção de resíduos: quer adquirindo menos produtos, quer passando a adquirir produtos mais sustentáveis (exemplo: com menos embalagem).
- Praticar a compostagem doméstica da fração orgânica: ajeitada para tratar os resíduos alimentares e de atividades agrícolas ou jardinaria.

Os agentes escolhem qual será o seu nível de implicação em função das suas características demográficas e com base no valor (atribuído aleatoriamente) que adotem duas variáveis internas que cada agente possui, representando respetivamente a «força de vontade», requerida para adotar práticas mais sustentáveis e mais a «sensibilidade perante o preço», que representa a disponibilidade para pagar mais ou menos pela gestão de RU. É claro que unicamente a segunda variável será diretamente afetada por uma mudança no sistema tarifário, mas isso não diminui a importância da outra. De facto, diversos estudos publicados demostraram já como a preocupação com o preço em si próprio não é o único fator que influi na maior ou menor implicação pessoal com uma gestão de resíduos e um estilo de vida mais sustentáveis – até porque o valor a pagar na fatura do lixo nunca será tão considerável como o pago por outros serviços básicos –, senão que há outros fatores – como a consciência ambiental ou o sentido cívico e de responsabilidade com a comunidade – que são igual ou até mais importantes na decisão a tomar. A respeito disto, deve ser lembrado que uma das chaves para explicar o êxito das tarifas variáveis de RU é a perceção de que são uma forma mais justa de distribuir os custos, pois recompensa a quem tenta produzir menos resíduos e penaliza a quem não o faz – de novo, o principio do poluidor-pagador.

Porém, atendendo às respostas dadas pela população nos inquéritos, considerou-se que nem todos os agentes reagem igual às alterações nas variáveis. Para além das diferenças demográficas, foi proposta uma distribuição dos agentes em quatro perfis de personalidade:

- <u>Insensíveis</u>: não reagem a nenhuma das duas variáveis (ou seja, não têm disposição a mudar nada).
- <u>Conscientes</u>: atuam com base na sua consciência, seja esta dirigida por uma preocupação pelo ambiente ou por um sentido cívico de responsabilidade. São sensíveis à variável da força de vontade.
- <u>Poupadores</u>: atuam com base na preocupação pelo dinheiro que gastam. São sensíveis à variável da preocupação pelo preço.
- <u>Sensíveis a tudo</u>: combinação das duas anteriores. Atuam atendendo às duas variáveis de força de vontade e preocupação pelo preço.

Existe, ainda, uma terceira variável a influenciar a toma de decisões: a pressão social – aquela que influi na toma de decisões não por uma convicção moral senão com o argumento de que «é o que faz o resto» e o desejo por evitar conflitos. Neste modelo representou-se a pressão social como uma possibilidade aleatória de que os agentes situados nas atitudes mais comprometidas com a sustentabilidade possam persuadir aos menos comprometidos para mudarem o seu comportamento.

Finalmente, o modelo assim configurado foi calibrado com os dados de campo obtidos da análise dos inquéritos feitas à população do bairro. Este passo possibilitou o uso posterior do modelo para gerar hipotéticos cenários futuros, alterando a pressão exercida pela tarifa sobre os agentes. Os resultados mostram que, com base na informação real recolhida na experiência piloto, se no início a grande maioria dos agentes – ou seja, as famílias do bairro – situavam-se no grupo unicamente comprometido com a separação de materiais para a recolha seletiva (66% dos agentes), após o fim da experiência formavam uma leve maioria aqueles que além do anterior tentaram reduzir o seu consumo (54% dos agentes). Esta maioria viu-se ampliada no cenário – este, já hipotético – de que a pressão exercida pela tarifa fosse aumentada num fator de 1,5: então, os agentes que optariam por reduzir o seu consumo somariam o 85%. Traduzido a impactos ambientais por meio da ACV consequencial, este último resultado supõe uma diminuição do impacto global graças à redução dos resíduos depositados no aterro, mas não assim – como já explicado anteriormente – naqueles impactos referidos à recolha e aos processos de reciclagem.

Os outros dois grupos, mais extremos, foram sempre minoritários, mas o interesse em descobrir quanto é que seria preciso aumentar a pressão tarifária para que mais agentes/famílias escolhessem praticar a compostagem doméstica, projetou-se um novo cenário com a tarifa multiplicada por 2, e se bem nesta situação o nível comprometido até o ponto de reduzir o consumo continuou a ser o maioritário – agora com o 65% dos agentes –, o seguinte nível comprometido também com a compostagem doméstica atingiu a segunda posição com o 31% dos agentes. Porém, o impacto ambiental correspondente mostrou um resultado em parte paradoxal: se bem os impactos ambientais diminuíram ainda mais em relação ao cenário anterior, houve categorias – como a da contribuição para as alterações climáticas – nas que esta diminuição não foi muito relevante. Isto pode ser explicado porque a retirada na composição dos RU da fração orgânica, agora desviada para a compostagem doméstica, prejudicou a obtenção de energia renovável que com ela tinha lugar na instalação TMB, no aterro e nos processos secundários derivados. Isto quer dizer que, chegado um certo ponto de aumento da pressão tarifária, o esforço requerido para melhorar ainda mais a avaliação ambiental começa a ser proporcionalmente maior que antes. Dito doutra forma: a medida que o avanço em direção à sustentabilidade vai progredindo mais, a margem de melhoria disponível para conseguir novos avanços vai ser menor, assim como o esforço para obter esses novos avanços será comparativamente maior. Uma conclusão que serve para melhor ponderar o contributo das tarifas variáveis nessa progressão em direção à sustentabilidade da gestão de resíduos, servindo como fechamento final do trabalho desenvolvido nesta tese.

Summary

This doctoral thesis addresses the environmental implications that can be expected from the application in the field of urban or municipal solid waste management (MSW) of variable pricing systems for payment by citizens of this service, which is usually a responsibility of municipal entities (municipalities). A variable charge consists in assigning a unit price referred to the usual unit of measurement of the good or service provided, calculating the total value to be paid by the user based on the number of units purchased. The concept of the variable tariff is common in the case of the payment of other basic services for the population such as the supply of water or electricity, in which a unit price is assigned to units such as litre or kilowatt-hour (of water and electricity, respectively). Within MSW management, its application is quite widespread in the countries of Central and Northern Europe (Austria, Czech Republic, Sweden, Switzerland, ...) but, except for Italy, not so much in the countries of southern Europe - namely in Portugal, where the practical work of this thesis has been developed - the existence of *flat tariffs* being more common in these countries – that is, a fixed value, independent of the amount of waste produced – or indexed tariffs based on other different variables and independent of the waste generation itself , such as water consumption. Disseminating the potential environmental benefits of this alternative policy is a purpose of this thesis which may contribute to overcoming the barriers that still exist for the extension of its effective use in these countries.

The idea of applying the variable charge to the payment of the MSW management service is currently gaining interest in the current context in which there is a tendency to promote an evolution towards the circular economy, with the theoretical horizon of achieving a “zero waste” society. This new paradigm requires the incorporation of sustainability – fundamentally environmental, but also economic and social – as a necessary dimension to consider when assessing the effectiveness of MSW management systems in the performance of their functions. In this sense, it is considered that the adoption of a certain economic policy regarding the price to be paid for waste management may have the capacity of decisively influence the characteristics of its generation. In other words, what is being proposed is to use MSW management service pricing as a means to achieve a given environmental performance target. Such a purpose is justified by the need to disconnect the increase in waste generation from economic growth and thus reduce the amount of MSW actually produced, as the only way to achieve environmental sustainability. This is precisely the main purpose of variable pricing systems based on pay-per-generation, thus aligning with the general ethical principle of environmental liability, called the “polluter-pays principle”: “the one who pollutes, must pay”.

However, it is necessary to first define what is meant in practice by sustainability before setting specific goals to be achieved. That is, to establish which are the specific variables in which a MSW management system (within a municipality, as understood in this paper) is supposed to progress if it pretends to effectively move forward on the sustainability path. This is precisely the purpose of the first work addressed in this thesis (in Chapter 3), which defined a set of six dynamic indicators that assess the year-on-year progress of the entity towards sustainability – that is: the value of the variable in the reference year is compared with the value of the previous year –, each of the three fundamental environmental, economic and social dimensions being represented by two of the six indicators. The complete set corresponds to these indicators:

1. Progress in reducing MSW landfilled as final destination.
2. Progress in reducing MSW production.
3. Progress towards financial balance.
4. Progress in costs reduction.
5. Progress in accessibility to separate collection.
6. Progress in reducing the number of complaints.

The indicators do not take absolute values, but are always expressed in relative terms: the progress made in relation to the previous year. This will make sense until the final horizon is reached, which may not be feasible in some indicators – for example, zero waste or zero cost – but in others it may be so: achieve a financial balance or receive no complaints. Once the individual results of each indicator have been obtained, it becomes possible to combine them all to condense them into a single value, as a "sustainability index in MSW management". For this last operation, it was decided not to apply any weighting factor, giving all the indicators the same weight in the overall result; but whether or not to apply some weighting is an issue that remains open for future developments.

Although generalisable to other entities, the set of indicators has been designed taking into account the context of the municipalities in Portugal, where all the information required for the calculation of each indicator is available to the public in each municipality. Thus, it was decided to experiment with the application of the set of indicators to the particular cases of two Portuguese municipalities A and B, resulting in MSW management policies in municipality A allowing it to move towards sustainability by more than 10%, however those of the municipality B made it to go back by almost 3%. However, although the set of indicators allows in principle the comparison between municipalities by using homogeneous data, it is clear that the particular result of each municipality will be affected by specific circumstances (geographical characteristics, demographics, etc.) that influence its outcome, hence it is necessary to take this comparison with caution and that is why it was preferred not to identify the chosen municipalities so as not to arouse susceptibilities. That is not a reason, though, to the proposed scoreboard being no longer useful as an evaluation and communication tool.

Once the relevant variables have been defined in relation to the overarching objective of sustainability – especially the environmental one –, it is necessary to establish which will be the most appropriate analytical methodologies to link these variables with the environmental impacts generated by them. The work of this thesis has given special prominence as a tool for analysing the environmental performance of MSW management to Life Cycle Assessment (LCA), which is currently a common and well-established method of assessing the environmental impacts of products and services. In its *attributional* modality, LCA can be used for descriptive analysis in terms of environmental impacts of a given set of processes, such as a waste management system. However, when it comes to using LCA as a support and guidance tool for decision making, involving the environmental comparison between different alternatives, the *consequential* modality may be more appropriate, as it focuses the analysis on the environmental consequences that a particular system experiences as a result of a change in demand for the product or service in question. This alteration may be caused by a change in the external factors that govern the system under study, such as in this case: a change in the tariff policy of the MSW management. Specifically, in this thesis the LCA technique was applied for the environmental assessment of a pilot experience for implementing a MSW collection system associated with a variable tariff in order to determine whether the change in the pricing system effectively led to a more environmentally favourable scenario than the previous one, and closer to sustainability.

This pilot experiment was carried out within the scope of the LIFE PAYT project, which aimed to promote the introduction of variable tariffs indexed to the MSW generation in three southern European Union countries (Cyprus, Greece and Portugal). This experience took place in a neighbourhood of the Portuguese city of Aveiro, specifically selected for this purpose. The practical fieldwork consisted of a collection of information on the generation and composition of the UK in the neighbourhood – performed through physical characterisation and weighing campaigns – and, at the same time, a sociological characterisation of the attitudes and opinions of the resident population with respect to the waste problems, conducted through a survey to the resident population. These tasks were performed in duplicate at two different time phases: before the implementation of the new system, to know the initial baseline situation, and later after the end of the experiment, to evaluate the results obtained and the effects observed. The data obtained in the fieldwork were used to prepare two life cycle analyses: a first one, that adopted an attributional perspective to describe the initial environmental situation of the MSW management in the neighbourhood, and a second one, this time with a consequential perspective, which is justified precisely by the stated intention to assess the environmental consequences arising from the change in the tariff scheme.

The attributional analysis corresponds to Section II of this thesis, which was divided into an analysis of the initially existing MSW collection system (in Chapter 4), followed by a global analysis of the management system, with an emphasis on the carbon footprint, linked to an analysis of costs (in Chapter 5).

In the attribution analysis of the collection system, a life cycle inventory was compiled counting the inputs and outputs – relating to polluting material, energy and emissions – which can be attributed to all existing elements in a UK collection system, which for the case of neighbourhood analysed are summarised in:

- <u>Plastic bags (of polyethylene)</u>: for the initial storage and transport of waste.
- <u>Rubbish bins</u>: where to put the bags while they are not full.
- <u>Street containers</u>: for the storage of waste prior to collection.
- <u>Collection vehicles (lorries)</u>: necessary to perform the waste collection from containers and the transport to the treatment facility.

After analysing the environmental impacts associated with these elements (done with the ReCiPe methodology), it was found that the main impacts were those corresponding to the consumption of fuel by collection lorries in those categories of environmental impact most related to air quality – land acidification, climate change, formation of particulate matter, formation of photochemical oxidants, in addition to terrestrial ecotoxicity. In contrast, in other relevant categories such as fossil fuel consumption and freshwater ecotoxicity, the greatest impacts were due to the use of plastic bags. Given this result, it was found that it was of interest to deepen the analysis of the bags impact to see if other materials alternative to polyethylene would be able to reduce these negative impacts. However, after comparatively testing the impact of bags made from two different biopolymers – thermoplastic starch and poly-hydroxyalkanoates – and polyethylene produced entirely from recycled material, it was found that only the latter bag model was able to significantly mitigate the impacts, compared with the other. In addition, the influence of bag design on its impact was also analysed: analysing the measurements of various models of rubbish bags of various sizes, it was observed that the amount of polyethylene used to make a bag does not vary proportionally linearly to the size, but both smaller and larger bags require a relatively larger amount of material, with the optimum size being around 40 litres in volume.

With regard to fuel consumption, it was concluded that, given the great influence it has on the overall result, it is essential that the source of the data used to calculate the amount consumed in the collection be preferably obtained directly from the vehicle records involved in the analysis – as done in this work – because any other alternative source of information will result in estimates that might fall far away of the reality that is intended to be represented.

Next, in Chapter 5, the attributional analysis was extended to integrate the whole phase of treatment and recovery of MSW, which in the case of the neighbourhood studied takes place at regional level through a mechanical-biological treatment (MBT) plant, intended to receive the fraction of MSW not previously separated through selective collection. At MBT facilities, the MSW organic fraction is separated from other materials to be degraded in an anaerobic digestion process where methane-rich biogas is produced, which serves as fuel for electricity generation, while the rest of the remaining organic matter is sent to a composting process. From the rest of MSW, the most valuable materials for recycling (metals and some plastics) are separated and added to those already recovered through separate collection (metals, paper/cardboard, plastics and glass). All these recycled materials, in addition to the renewable energy obtained from biogas and compost, receive in their environmental impact values a "credit", based on the amount of production of primary materials and energy that they have contributed to prevent. The refused waste not suitable for recycling is disposed of in a controlled (sanitary) landfill, its environmental impacts being modelled basing on the composition obtained with the mass balances elaborated from the initial composition of MSW, which is known from the characterisation campaigns. From the analysis of this set of processes, it was concluded that, by far, the pollutant emissions caused by landfilling are the largest environmental impact recorded, confirming the prominence given to this form of treatment in the sustainability indicators developed in Chapter 3. This is also the case of the carbon footprint, which due to its current relevance was the category selected to generate a series of alternative scenarios, in which it was tried to find out which parameters would be improved within the MBT installation to achieve a carbon footprint, if not neutral, at least as close to neutrality as possible. Finally, this carbon neutrality was achieved in the most ideal of the projected theoretical scenarios, in which the separation at source of recyclable materials would be the maximum possible, the separation of the organic fraction in the MBT would also be the best possible and would facilitate an optimal yield of biogas production and compost, and finally there would be at least part of the waste that would still be used in the form of waste-derived fuel (RDF), suitable for the cement industry, for example.

Regarding the cost analysis, it should be noted that, based on the deficit situation found at the time of the first analysis, the successive projected improved scenarios allow to improve the overall balance between costs and benefits of MSW management, but in this case without reaching equilibrium – although not by far. In any case, it should be remembered that the balance between the benefits and costs of MSW management is not a priority of this activity, the difference being covered precisely thanks to the value paid by the public in the waste tariff.

As stated above, in Chapter 6 the analysis applied to the case study takes a different and complementary perspective from the previous attributional one, now becoming consequential. This approach allowed the assessment on the MSW environmental impact of the effects of tariff changes from a broader point of view, not limited to the effects directly caused by MSW management, but also covering other external systems. with which this indirectly interacts. In relation to this implication, it was modified, relatively to Chapters 4 and 5, the way in which the benefits accruing from the replacement of primary energy sources and raw materials extracted from nature, respectively by renewable energy and recycled materials obtained from valorisation of MSW, to include now also the underlying processes that will be marginally affected by the changes experienced in the output products of MSW management, namely:

- Electricity from biogas: replaces the combination of electricity sources planned in Portugal in the period 2020 - 2030.
- Compost: replaces manure as a soil structuring agent, which alternatively can be treated by anaerobic digestion to obtain biogas, which in turn replaces natural gas as fuel to generate electricity and heat.
- Recycled paper: replaces non-recycled paper, which leaves forest biomass available to be used as an alternative as a fuel to generate electricity and heat, in turn replacing natural gas.
- Recycled metals, plastics and glass: they replace their respective non-recycled equivalents.

The first intention was to use this consequential analysis to establish a comparison between the two initial and final moments of the pilot experience with the variable charge. The field data obtained in the campaigns showed an overall reduction in MSW generation of more than 17%, within which the mixed MSW without separation ("rest" fraction) decreased by more than 30%, while the MSW for separate collection increased by 48%. In the translation of these numbers into the corresponding environmental impacts through LCA, the result was that, with the impact of landfill disposal being so dominant upon the other – especially in the categories of climate change, ecotoxicity to freshwater and human toxicity –, any decrease in MSW production as the one observed will result in a net reduction of the overall impact of the system, although the impact in other categories less dependent on landfilling increases, due to the increased effort required for an expanded separate collection and also due to the impact of the recycling processes themselves, which the benefit discounted by the replacement of materials and energy does not fully compensate.

However, it turned out to be of the utmost interest to take advantage of the potential of consequential LCA to represent the variation of long-term environmental impacts for trying to build a model that would somehow predict the evolution of MSW generation, basing on the behaviour of the population affected by the tariff change. For this, the agent-based modelling (ABM) methodology was used. This technique considers the population involved in the experience as a set of agents, characterised by a series of distinctive properties, which depending on variables that are internally attributed to them, are able to adapt their behaviour through decision-making – supposedly rational – when they are subjected to an alteration of the circumstances upon which they act. For this case study, a population of agents that reproduces the demographic characteristics of the neighbourhood analysed was designed on an ABM computer software: the agents are equivalent to families (understood as a community living in a household), taking into account if they have members under 15 or over 65 and if the dwelling is an apartment or a detached house with garden. In order to deal with their waste, the agents choose between four levels of involvement, the first being not to worry about the waste, and the other three being to cumulatively adopt increasingly sustainable practices, namely:

- Separate recyclable materials for separate collection.
- Modify consumption habits to reduce waste production: either by purchasing fewer products or by switching to more sustainable products (example: with less packaging).
- Practice domestic composting of the biowaste fraction: suitable for treating food waste and agricultural or gardening activities.

Agents choose their level of involvement based on their demographic characteristics and based on the value (randomly assigned) adopted by two internal variables that each agent possesses, representing respectively the “willpower”, required to adopt more sustainable and more sustainable practices and the “price sensitivity”, which represents the willingness to pay more or less for MSW management. It is clear that only the second variable will be directly affected by a change in the tariff system, but that does not diminish the importance of the other. In fact, several published studies have already shown that concern about price itself is not the only factor influencing greater or lesser personal involvement with a more sustainable waste management and lifestyle – indeed, because the value to pay on the waste bill it will never be as significant as paying for other basic services –, but there are other factors – such as environmental awareness or a sense of civic responsibility and community responsibility – that are equally or even more important in the decision to make. Regarding this, it should be remembered that one of the keys to explaining the success of MSW variable pricing is the perception that they are a fairer way to distribute costs, as it rewards those who try to produce less waste and penalises those who do not – again, the polluter-pays principle.

However, in view of the responses given by the population in the surveys, it was considered that not all agents react in the same way to the alterations in the variables. In addition to demographic differences, a distribution of agents into four personality profiles has been proposed:

- Insensitive: They do not react to either variable (that is, they are unwilling to change anything).
- Civic-minded: They act on the basis of their conscience, whether it is driven by a concern for the environment or a civic sense of responsibility. They are sensitive to the variable of willpower.
- Saver: They act out of concern for the money they spend. They are sensitive to the variable of price concern.
- All-sensitive: combination of the two above. They act according to the two variables of willpower and concern for price.

There is still a third variable that influences decision-making: social pressure – which influences decisions not by moral conviction but by the argument that “everybody does it like that” and the desire to avoid conflict. In this model, social pressure was represented as a random possibility that the most committed agents to sustainability attitudes could persuade the least committed to change their behaviour.

Finally, the model thus configured was calibrated with field data obtained from the analysis of surveys conducted on the population of the neighbourhood. This step made possible the later use of the model to generate hypothetical future scenarios, altering the pressure exerted by the tariff on the agents. The results show that, based on the actual information gathered in the pilot experiment, while in the beginning the vast majority of agents – that is, families in the neighbourhood – were in the group solely committed to the separation of materials for separate collection (66% of agents), after the end of the experience majority those who also tried to reduce their consumption (54% of agents) formed a slight majority. This majority was extended in the scenario – this one, already hypothetical – that the pressure exerted by the tariff was increased by a factor of 1.5: then, the agents who would choose to reduce their consumption would add up to 85%. In terms of environmental impacts through consequential LCA, the latter result corresponds to a reduction in the overall impact thanks to the reduction of landfilled waste, but not so – as explained above – in those impacts related to collection and recycling processes.

The other two, most extreme, groups were always a minority, but due to the interest in finding out how much it would take to increase the tariff pressure for more agents/families choosing to practice home composting, a new scenario was projected, with the tariff charge multiplied by 2, and although in this situation the level committed to the point of reducing consumption remained the majority – now with 65% of agents – the next level committed in addition to home composting reached the second position with 31% of agents. However, the corresponding environmental impact showed a somewhat paradoxical result: although the environmental impacts decreased even more in relation to the previous scenario, there were categories – such as the contribution to climate change – in which this decrease was not very relevant. This can be explained by the fact that the withdrawal in the MSW composition of the biowaste fraction, now diverted to domestic composting, has affected the renewable energy obtained in the MBT facility, in the landfill and in the secondary processes derived. This means that, at a certain point in the increase in tariff pressure, the effort required to further improve the environmental assessment is starting to be proportionally higher than before. In other words, as progress towards sustainability advances deeper, the room for improvement available for further improvements will be smaller, and the effort to make those new progresses will be comparatively greater. A conclusion that serves to better assess the contribution of variable charges in this progression towards the sustainability of waste management, as closing mark of the work developed in this thesis.

Section I.
Contextualisation

1. Introduction and Objectives

Álvaro Fernández Braña

ABSTRACT

The increasing generation of waste as a consequence of economic development is nowadays becoming a generalised concern. The new circular economy paradigm, in accordance with the aim of achieving closer societies to environmental sustainability – the *zero waste* horizon – requires waste generation to be decoupled from economic activity. Accordingly adapting the waste pricing policy may be an effective tool to effectively reduce waste generation, in order to achieve that objective. To this purpose, variable pricing strategies indexed to the amount of municipal solid waste (MSW) effectively generated – known as unit-based pricing (UBP) or *pay-as-you-throw* (PAYT) – are in line with the polluter-pays principle, and demonstrated success in reducing the generation of mixed (unsorted) MSW, basing on these possible pathways: increasing MSW diverted to separate collection of recyclables, reducing the overall MSW generation by decreasing consumption, and promoting home composting for biowaste. As UBP pricing schemes are not common in the south of Europe, the LIFE PAYT project was intended to put in place several PAYT pilot experiences in Cyprus, Greece and Portugal to test and show its benefits. The experience performed in the Portuguese city of Aveiro was selected as a case study for this thesis, in order to perform a complete assessment of the environmental effects derived from the change in the pricing policy and its contribution to sustainability.

1.1. THE WASTE ISSUE

Waste generation – comprising solid waste, but also wastewater – is an inherent consequence of human life, thus making unavoidably necessary the development of waste management systems. In rural societies, comprising territories where population density is low, waste management has historically not been a generalised problem, since waste generation in that situation is typically limited to food waste and agricultural waste – i.e. biowaste – while materials such as ceramics, glass or metals were constantly reused to the maximum extent. In the final case of disposal – either through burning or disposal in dumpsites –, they did not pose anyway a noticeable harm to the environment at that time, due to its low toxicity or the limited amount of waste which was disposed of (ACCP, 2019). The main purpose of waste management in this context – and indeed, the primary function of every waste management system – is to protect public health by preventing the spread of disease vectors and the pollution of the surrounding environment, especially water sources. Only the strengthening of ecological concerns in recent times has widened the perception of waste generation as a global problem, not just focused on localised threats to human health, but contributing to worldwide issues such as global warming. Global warming is chiefly due to greenhouse gases emissions (GHG) caused by the biodegradation of organic matter – responsible for 3% of total GHG emissions in the European Union (EU), (EEA, 2021), and particularly for 8% in Portugal (APA, 2021) – or other potential risks, such as the widespread presence of microplastics in the environment (SAPEA, 2019).

The consideration of waste management as a challenge for human development is closely associated to the urbanisation process and the increasing development of economic activities (Karak et al., 2012). Nowadays, the increasingly growing generation of waste has become a symbol intrinsically associated to consumerist societies, and one of the main concerns to be solved as part of the challenge for achieving a more sustainable relationship between humans and environment. This is especially the case of the more industrialised countries, responsible for the major part of waste generation. According to the World Bank (IEG, 2020), the average waste generation *per capita* in high-income countries reached 1.9 kg/day, while in the group of low-income countries it is less than 0.5 kg/day. But also in the developing countries waste management is rapidly becoming a relevant concern, as long as their economies and urbanisation progress: Kaza et al. (2018) estimated that waste generation will have grown by 19% in 2050 in high-income countries, but more than 40% in middle and low-income countries.

1.2. WASTE PRICING POLICIES AND THE POLLUTER-PAYS-PRINCIPLE

In most countries, the responsibility of waste management is typically assigned to municipal (local) governments, while upper instances of government are responsible for developing overarching policies and providing a regulatory framework. Given the complexity of waste management systems, the financing of this service is usually one of the great sources of expenditure for a municipality. This is true especially in the case of municipal solid waste (MSW) collection – comprising 70–80% of total MSW management costs (UN–HABITAT, 2010) –, since it must be performed throughout the inhabited territory of every municipality, whereas for the waste treatment stage it is usual a single treatment facility being able to serve a group of several associated municipalities which share together the costs. In high-income countries such as those in Western Europe, it is estimated that the management of 1 tonne MSW requires roughly 100€ (Kaza et al., 2018).

Notwithstanding, when addressing the financing approaches for MSW management practiced by municipalities, a variety of different options is found, even inside the same country

– since municipalities usually have some autonomy to decide how to finance this service. In the case of Portugal, where the work for this thesis took place, this was the situation found in the year 2005, taken from the study of Simão Pires (2013):

- 16% of municipalities financed MSW management with their own budgets, without charging any specific tax or fee from citizens.
- 24% had in place a so-called *flat fee*: a fixed price which is regularly paid by every household or establishment.
- 49% practiced an indirectly indexed fee: in this case, the value to pay for MSW management is indexed to a proxy measurable variable, which in this case corresponds to the water consumption registered in the household or establishment.
- 11% had fees indexed to other variables, either fixed such as household area, or variable, such as the collection frequency.

Only in the last case, where the tariff depended of collection frequency, there exists to some extent a relationship between the actual quantity collected and the price paid for it. But, in the rest of cases, the value paid as waste tariff is completely independent of how much waste has been produced – moreover, the indexing related to water has been criticised for being sometimes unrealistic (Simão Pires, 2013). The undesired consequence of these pricing policies is the lack of incentive for the population to move towards more sustainable lifestyles – that is: to generate less waste. This is exactly what the so called "unit-based pricing" (UBP) schemes try to tackle, by indexing the price paid for the service – waste management – to the extent of use which has been made by each user – waste quantity effectively discarded. Although the concept of UBP had been first applied to waste tariff schemes since decades ago, it has been the emergence of the "circular economy" paradigm – intended to achieve an effective decoupling of waste generation from economic growth – the driving force to encourage the idea of applying the "polluter-pays-principle" to waste collection pricing – under the name of *pay-as-you-throw* (PAYT), to the point of being favoured by EU authorities as a feasible instrument for achieving a reduction of MSW generation (Bio Intelligence Service, 2012).

1.3. Variable waste pricing: PAYT/UBP

The key technical aspect for the effective implementation of a PAYT/UBP waste tariff scheme consists on the identification of the user who is discarding his/her waste into the collection system. This can be fulfilled by means of several technical alternatives, which can be summarily classified in two main groups: on the one hand, those based on collective collection system where the user identification is done by limiting the access to the collection equipment – e.g. locked waste bins with personalised access through cards, keys, chips or similar – and on the other hand, those based on an individualised collection service where the user – or more exactly: her/his household – is directly identified by the collection operators: this corresponds to the wide range of door-to-door (DtD) collection schemes. The latter is more intended for lower population density areas, based on single housing – i.e. detached houses –, while the former is more adequate for high population density areas, based on multi-household buildings. Thorough descriptions of PAYT systems are those published by Bilitewski (2004) and Reichenbach (2008).

Although widespread in developed countries (Sakai et al., 2008; Skumatz, 2008; Usui and Takeuchi, 2014; Van der Werf et al., 2020; Huang et al., 2021) and particularly successful in Central Europe (Bilitewski, 2008; Reichenbach, 2008; Šauer et al., 2008; Allers and Hoeben, 2010; Morlok et al., 2017; Slavík et al., 2020; Slučiaková, 2021), PAYT policies have not yet been massively introduced in Southern Europe – with the notable exception of Italy (Bio Intelligence Service, 2012; Compagnoni, 2020). Within this context, the EU-funded LIFE PAYT project[1] was intended to promote the introduction of UBP in Southern Europe in five locations of three southern European countries – namely: Cyprus, Greece and Portugal –, by implementing and monitoring a PAYT pilot experience in each of the five selected locations. The field work which served as a basis for the present thesis corresponded to one of the five pilot experiences, which took place in a residential neighbourhood located on the Portuguese city of Aveiro. Further detailed information on the experience is to be found in Chapter 2.

1.4. PAYT/UBP AS A TOOL FOR SUSTAINABILITY

The main ultimate goal which is expected to be attained with the adoption of a PAYT pricing system consists on an effective reduction of the waste generated by the households or establishments affected by this kind of tariff, in comparison with the waste generated under other pricing systems not indexed to the quantity of waste. Notwithstanding, this waste reduction should not be understood as a straightforward consequence of the direct application of the PAYT tariff, but as a progressive process following several pathways, depending on the particularities of each case:

- Provided that separate collection circuits exist – at least in the European context which applies to the scope of this work – for the most common recyclable materials which can be separated from residual mixed MSW at the source of consumption – namely: the three fractions corresponding to glass, paper/cardboard and co-mingled packaging materials made of metals and plastics – and, given that those separate collection fractions are usually excluded from the waste tariff – since their management is financed by the producers of goods via the Extended Producer Responsibility (EPR) schemes –, a quite expectable consequence of PAYT/UBP implementation is the increase of recyclable materials discarded by participating citizens into the separate collection circuits. This appears as the easiest way to reduce the MSW amount quantified for billing, at least for those people who did not separate recyclable materials before.
- However, the previous action does not imply a real decrease of the MSW generated, but a diversion of materials towards recycling, aimed at lowering the price paid for management. Although it is arguably a more responsible and sustainable behaviour, it still does not fit the ultimate objective of waste decrease. To this purpose, the only manner to prevent waste generation consists in modifying the consumption of products, that is: either to effectively consume less products, or alternatively, consume products associated to a lower waste generation – e.g. products with less packaging material.
- There is still a third possible response encouraged by PAYT: to keep consumption in the same level, but not to discard the waste generated into the management system, so that it is not accounted for the tariff to be paid. This might happen in an illegitimate manner – i.e. disposing of waste in a not allowed manner: dumping waste out of the collection devices (“free riding”), burning it or carrying it to other municipalities without open collection devices (“waste tourism”) – but also in a legitimate manner, if citizens are able to manage their own waste by themselves with sustainable methods. This second situation is the case of home-composting, a practice which allows the treatment of biowaste without discarding

[1] www.life-payt.eu

it in the collection system. From the point of view of the MSW management operators, this is equivalent to waste prevention: although the waste has been effectively *generated*, it has never been *collected* and therefore it is not considered for calculating to price to be paid. In fact, some municipalities do indeed encourage the practice of home composting by citizens, either in an individual or collective form, as an alternative to alleviate pressure from collection systems by diverting away significant amounts of waste.

In general, municipalities where waste management pricing based on PAYT/UBP has been implemented have obtained positive results, with at least some of the previous effects described above – if not all – having effectively taken place to some extent immediately after the adoption of the new scheme. It seems undoubtful that any increase of the effects observed, in terms of waste prevented or separated for recycling, contributes to progress towards a more environmentally sustainable situation, closer to the circular economy paradigm: the *zero waste society*. However, it is not straightforwardly evident how the changes on the amounts of waste derived from the reaction of the population facing the new tariff scheme can be translated into measurable environmental impacts, and to which extent these impacts contribute or prejudice the ultimate goal of sustainability, a challenge which constitutes the main question to which this thesis has tried to give an answer.

Besides the factors which are intrinsic to the population, such as demographic and social conditions – age, income, education level, type of housing, etc. –, the relative success – or failure – of a PAYT experience is dependent on the particular situation found regarding waste management in the place where it is to be applied at the moment where the tariff change is introduced and the experience with the new system begins. Features like the type of collection implemented – collective street containers or door-to-door –, the frequency of collection, the distribution of the bins/containers, the ratio between mixed MSW and separate collection equipment placed and even the specific model of bin used, are all factors which influence the acceptability of the price paid and, hence, the environmental performance of the system (De Jaeger and Eyckmans, 2015). Municipalities without good starting conditions regarding these aspects may not be able to achieve great results from a change in the pricing policy. On the other extreme, it seems also clear that, in municipalities with already well-developed and optimised collection systems allowing high rates of source separation and perhaps giving support to home composting, linked to a population with high environmental awareness which already practises a sustainable lifestyle, a change of the tariff will not bring many further environmental benefits: their room for improvement is reduced, since they are already good performers. The evaluation criteria to establish whether a given policy concerning waste management has been successful or not, should be, therefore, adapted to these particular circumstances. A thorough analysis of the initial scenario is, thus, necessary as an exercise of benchmarking, in order to correctly assess and evaluate the later consequences of a change in the pricing policy – primarily on the environmental dimension, but also in the financial aspects (Le Bozec, 2008). This exercise has been addressed in Section II of this thesis, where the Chapters 4 and 5 are dedicated to make use of the Life Cycle Assessment (LCA) methodology to analyse the environmental implications of the waste management present in the case study addressed, focusing on exploring how particular aspects of the management process have a significant influence on the overall environmental outcome, such as the type of bags used for carrying the waste (Chapter 4) or the optimisation of the system outputs, in this case focused on carbon footprint and linked to a cost analysis (Chapter 5).

Actually, the question regarding how to evaluate the environmental performance may well be extended, not to focus only in the effects derived from the pricing policy, but to the whole environmental outcome of a waste management system, comprising also the efforts made for waste prevention – an aspect often disregarded in waste management research studies (Laurent et al., 2014; Bernstad-Saraiva et al., 2018). For instance, the environmental impact derived from an optimised waste management system which has been able to reach high rates of recycling and valorisation of waste, but located in a region where the waste generation rates are extremely high (e.g. a developed touristic region), might perhaps be not that different of the impact caused by a less optimised management system based mostly on waste landfilling, but located in a region where waste generation is low (e.g. a rural region). Again, a performance assessment should take into account these differences in the particular context of each case in order to obtain fairly, not biased comparable results. It is actually the existence of these kind of nuances when addressing the issue of waste management performance measurement what justified the inclusion in this thesis, previously to the actual assessments of the case study, of the contextualisation work developed in Chapter 3, dedicated to the development of suitable indicators for measuring waste management performance with respect to their progress towards sustainability, not just concerning its environmental dimension but also the economic and social implications as well.

There is still a further issue to be dealt with when assessing the impacts of PAYT/UBP, namely: to delimit which part of the results obtained can be effectively attributed to the change in tariff by itself. It has been already demonstrated that the economic incentive is not by itself the only influential variable for adopting a more sustainable behaviour (Heller and Vatn, 2017). Other factors, such as personal beliefs – e.g. environmental awareness – or the social pressure – i.e. respect for the norms established and accepted by the community – play also a decisive role (Thøgersen, 2003; Dunne, 2008). Still, this observation should not be understood as an underestimation of PAYT ability to change the population attitude concerning waste generation, but as the ascertainment that the economic dimension is not the only acting vector of change. If compared to the price of other services and commodities commonly paid by households (electricity, gas, water), the monetary value of the waste tariff does not usually put great pressure on a household income – specifically in the city of Aveiro, the average *per capita* value paid by every city corresponded to 34.5 €/year in 2020 (INE, 2021)–, so that its compelling power to induce behaviour changes remains limited. However, the results obtained in the LIFE PAYT project, as well as those from other similar experiences known by the author, show that the mere adoption of a new variable charge on MSW may act as a “trigger” on the awareness of citizens, even if the new monetary value to pay does not result in a significant real change in price. This can be explained, either due to the variable charge acting as a symbolic reminder of the negative environmental impacts caused by waste generation, or as a consequence of the fear sensation induced in some citizens, worried about their household waste being controlled and inspected by authorities in pursue of norm violations – regardless of this “threat” being real or not.

The issue requires, along with the other aspects commented before, a sound and quantified assessment, which was addressed in Chapter 6 of this thesis, basing on the field data obtained in the practical pilot experience developed in Aveiro and making a combined use of two modelling techniques – Agent Based Modelling (ABM) and the consequential approach of LCA (C-LCA). Altogether, it is expected that the works developed within the scope of this thesis will bring a relevant contribute to the better understanding of the actual environmental benefits brought by PAYT/UBP or variable charging systems, still regarded as a pending matter for research in the field of waste management.

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2. Case study and Methods

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ABSTRACT

The practical work performed in this thesis was mainly based in a case study, which consisted in a pilot experience for implementing a PAYT pricing system that took place in a specifically selected neighbourhood of the Portuguese city of Aveiro. The experience lasted for several months, during which the conventional MSW collection of open collective street containers previously in place was substituted by new containers with electronically controlled access through user cards distributed to the population. The number of discards made by citizens were registered as a basis for calculating the tariff value to be theoretically paid. The situation found in the neighbourhood relatively to MSW management was evaluated before the beginning of the pilot experience and after its end by means of field waste characterisation campaigns and surveys made to the population. The data obtained were later used as input for developing a Life Cycle Assessment (LCA). For the making of LCA, advantage was taken from its two possible modelling approaches: attributional and consequential: an attributional perspective was chosen for describing the situation found before the project, whereas consequential modelling was found to be more suited for evaluating the systemic effects induced by the change of policy.

[2] Further details provided in the List of publications included (page xxxvii).

2.1. CASE STUDY

The city of Aveiro is located on the Atlantic coast (Figure 2.1), and with its nearly 80,000 inhabitants forms the fourth largest urban area in the Portuguese Centre Region. The city has developed itself to become a regional industrial, trade, academic and touristic hub. Given these features, Aveiro is considered as a representative example of a typical European medium-sized city. More than half of urban European population live in cities below 1,000,000 inhabitants, and roughly a 28% in cities under 250,000 inhabitants (EC & UNHABITAT, 2016). In consequence, there currently exists in Europe great interest in gaining a better understanding of environmental issues in this kind of cities, ultimately related to the quality of life of its population.

This study has been run under the scope of the LIFE PAYT project – funded by the European Commission and aimed at the implementation of variable PAYT ("pay-as-you-throw") pricing schemes for MSW tariffs in Southern European municipalities. Although separate collection schemes are already well established within the Aveiro municipality, the amount of recyclable materials which is separately collected remains still low – roughly a 6% of the total MSW collected (ERSUC 2016). Moreover, this percentage of recyclables source separation remains stagnated during last years, while on the other hand the overall MSW generation is growing again along with the recovery of economic activities after the financial crisis in 2010-2014 (INE 2019). This situation is similar to the rest of Portugal, hence the interest of the municipalities for the introduction of new incentives – namely: PAYT schemes – in an attempt to reduce the generation of mixed MSW, and consequently, the costs associated to its collection.

In the case of Aveiro, the municipality showed interest on first testing the implementation of the PAYT system in a given neighbourhood of the city – named Forca Vouga (Figure 2.2) – thus the neighbourhood was designated as pilot area for the LIFE PAYT project.

Figure 2.1. Location of Aveiro in Portugal and Western Europe. Adapted from Wikipedia (public domain).

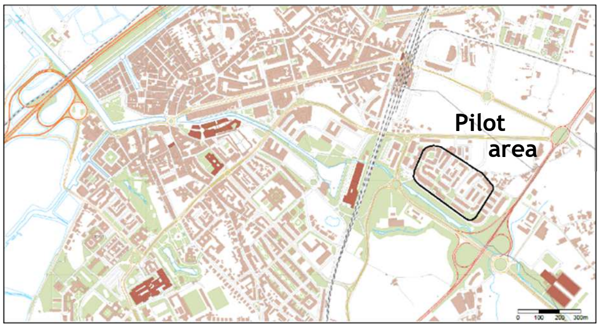

Figure 2.2. Location of the studied neighbourhood within the city. Adapted from Aveiro City Council (public domain).

The selected neighbourhood is situated to the east of the city centre (Figure 2.2) and consists on both apartment buildings as well as single-family houses. There are also some offices, bars and shops. The area has been recently developed with new constructions, but still lies on the edge of the urbanised surface and is mostly surrounded by open terrain. This relative isolation from the main urban core poses an advantage for PAYT experimentation, since unwanted deviations out of the area of household waste trying to escape the registration system are expected to be minimised. Due to its recent development, the neighbourhood is mostly populated by young, medium-high income families.

2.2. PAYT PILOT EXPERIENCE

Within the framework of the LIFE PAYT project, the MSW open access collection system previously existent in the neighbourhood was substituted by a limited access system based on user cards (Figure 2.3). To this purpose, the 1100L collective street containers open access were substituted by new containers of the same type, but equipped with an opening lid attached to a rotating drum which allowed the introduction of approximately one 30L bag full of waste at each time. The opening lid is controlled by an electronic lock, connected to the electronic card reader.

Figure 2.3. Old container (right) replaced by new PAYT containers (left). Picture by LIFE PAYT team.

A user electronic card to access the container was assigned to every household in the neighbourhood. Every time a citizen unlocks the container with a user card, the electronic device in the container registers the event as one waste discard. The register is submitted to an online database via GPRS mobile data transmission, enabling then the billing process according to the tariff established. The user is able to access the register of discards of her/his correspondent household through a personal account hosted in an online portal linked to the database, and subsequently check the values to be paid to the municipality, as well as verifying how the tracked record of the household MSW discarded has evolved with time. The data stored within the platform were later employed in anonymised manner in this thesis as a source of information for determining MSW generation trends in the neighbourhood.

The electronic containers were purposely developed by a Portuguese consortium of companies as prototypes, i.e. as an entirely new design. Even though this is regarded as a positive outcome in terms of the project in terms of innovative technology development, that circumstance resulted in some malfunctions and design flaws being detected during field testing. For that reason, the field experience was split in two phases: a first testing period ran from May 2018 to September 2018 with the first group of prototype containers; thereafter a pause followed in order to solve all the faults detected and finally, a second generation of improved containers was put in place from September 2019 until November 2020.

Information and training was offered to citizens in the beginning of the testing periods. As complementary measures to promote a more sustainable waste management, the number of bring banks for separate collection of glass, paper/cardboard and metal/plastic packaging was increased from 5 to 10 and, in parallel, home composters were distributed to the interested households, along with specific training given for home composting.

2.3. FIELD METHODS OF ASSESSMENT

2.3.1. Waste characterisation campaigns and weighings

As part of LIFE PAYT project tasks, field characterisations campaign of the mixed MSW fraction generated in the Forca Vouga neighbourhood were performed (Figure 2.4), with the aim of determining its composition. The characterisation followed the procedure established by European reference SWA-Tool methodology (EC, 2004). Two campaigns took actually place: the first before the implementation of the project, in order to assess the initial baseline scenario, whereas the second was scheduled shortly after the end of the testing periods, to evaluate how the waste generation pattern in the neighbourhood had changed with the adoption of the PAYT system.

Figure 2.4. Waste characterisation campaign. Picture by LIFE PAYT team.

Besides characterisation campaigns, a series of regular weighings of the daily amount of mixed MSW collected only in the target neighbourhood were done in selected dates of the year at the beginning of the project. Thereafter, these field data were compared with the historical records of collected MSW quantities kept by the Aveiro municipality to detect generation patterns and take into account the seasonal variation through statistic procedures which allowed to determine the total amount of mixed MSW generated in the neighbourhood. This particular work was published as Fernández-Braña et al. (2021). This result was later compared with data gathered from the PAYT online platform mentioned before to check the variations in MSW generated.

In the case of separate collection, the collected amounts were estimated basing on the regular controls of containers filling degree at collection made by the company responsible for separate collection.

2.3.2. Surveys to the population

In parallel to the activities dedicated to the physical characterisation and quantification of MSW, a sociologic characterisation of the population resident in the neighbourhood was performed, with the goal of better understanding their global reaction to the project and particularly their predisposition to adopt a friendlier attitude with sustainability concerning waste generation. In analogous manner to the waste characterisation described above, the survey was also twice performed: before and after the pilot experience. Both surveys were distributed to the population in two modes: either through a personal spontaneous interview with members of the project team or via an online questionnaire.

Besides general characterisation questions – age, gender, number of household members – the questionnaires contained some relevant questions for the project assessment, regarding the degree of commitment with source separation of recyclable materials and the opinion about the tariff system and the fairness of PAYT/UBP. In the second questionnaire, besides participants being asked about their opinion on the experience, some specific questions were added asking for the effects personally felt because of the project actions: whether they tried to somehow reduce their waste generation by collaborating with separate collection, adapting their consumption habits to the new situation or trying home composting.

In the first survey, 53 valid respondents were obtained, whereas in the second survey the number reached 51 respondents. A larger sample of population would have been desirable, but then the personnel and resources required would have beyond the available means for this project.

2.4. Analytical tools: Life Cycle Assessment

According to the ILCD Handbook (JRC – IES, 2010), Life Cycle Assessment (LCA) is a structured and comprehensive methodology to quantify the environmental impacts of a product or service along its entire life, from extraction of raw materials to final disposal. The procedure for elaborating a LCA has been codified by the International Organization for Standardization (ISO) in two norms: ISO 14040 and ISO 14044. Summarising, the structure of LCA is always organised into four main stages, namely:

- <u>Goal and scope</u>: initial phase for stating the reasons and purpose of the study and defining the system to be studied, along with its boundaries, functional unit, modelling approach and other methodological considerations.
- <u>Life Cycle Inventory (LCI)</u>: quantification of all inputs and outputs considered in the system to be analysed, comprising energy, materials, pollutant emissions released to the environment and waste.
- <u>Life Cycle Impacts Assessment (LCIA)</u>: linking every of the items considered in LCI to a defined environmental impact category (carbon footprint, ecotoxicity, ressources depletion, etc.).
- <u>Interpretation of results</u>: discussion of the data obtained and conclusions of the assessment.

The application of LCA to waste management was developed somewhat later after the generalisation of LCA as an environmental assessment tool, but it was nevertheless already firmly established by year 2000 (Finnveden et al., 1995; Finnveden, 1999; Clift et al., 2000; Ekvall and Finnveden, 2000). Since then, LCA, as a methodology closely associated to the concept of sustainable development, has found in the field of waste management one key area of interest for application, as demonstrated by the large number of studies published (Laurent et al., 2014). Its use for giving an answer to the objectives pretended by this thesis was, hence, fully justified. But not without addressing before some particular methodological choices which should be clarified.

Firstly, it should be noted that the functional unit selected for all the assessments presented in the next chapters was not a fixed number, representing a given quantity or number of units of the process to be analysed. In this case this would correspond to a given amount of municipal waste being managed. However, since the intention was always to make a comparison between the situation found before and after the change in the waste tariff scheme, it seemed clear that the amount of MSW received by the system would not be the same between the two moments – i.e. it was not expected to be the same. Hence, the functional unit could not be fixed, but it should be a variable concept: yearly MSW collected; remark: collected instead of generated because there is no other information available than that referred to the amount effectively collected by the system, other amounts of waste generated but not collected – such as biowaste composted at home – are not known and thus excluded from this assessment.

A second discussion is that referred to the modelling framework: attributional or consequential? Attributional modelling allocates the impacts to the functional unit by partitioning the unit processes of the system analysed where necessary, while in consequential modelling, impacts are totally linked to the activities that are expected to be affected as a consequence of demand changes for the functional unit, even if these activities are outside of the boundaries originally set for the system analysed – in other words: the system is expanded. According to the reference ILCD Handbook (JRC – IES, 2010), three possible contexts for LCA are considered, attending to its specific purpose: A (small-scale or local decision taking), B (large-scale or systemic decision taking) and C (no decision taking: a purely descriptive assessment). An attributional framework is recommended for situation C, while consequential is recommended for situation B and, lastly, both may be appropriate for situation A, depending on each particular case.

In the particular situation of waste management LCA assessments, even when an attributional perspective is adopted, is not uncommon to apply a punctual system expansion in order to include within the system the environmental benefits obtained from recycling and valorisation processes which obtain resources – materials and energy – from waste, preventing the consumption of other resources extracted from nature (Vadenbo et al., 2016). In the case here addressed, this attributional approach with punctual system expansion was chosen for the initial assessment previous to PAYT application, since the main purpose was to simply describe the situation found at that moment – even if some comparisons with hypothetical future scenarios were also included. However, given that the introduction of PAYT/UBP constitutes a significant change in the policies governing the system, a consequential approach was selected for the assessment comparing the PAYT scenarios with the previous one. In spite of the case studied being of small scale – just a pilot experience in a neighbourhood –, the change in the pricing policy is without doubt systemic, and would result in very significant consequences if it were to be applied, for example at the level of an entire city or at national level.

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3. A framework to assess and communicate progresses in sustainability

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ABSTRACT

A framework of indicators to assess the progress towards sustainability of municipal waste management utilities was developed. Its purpose is to fulfil the need for assessing the performance of municipal waste (MSW) management in a simple but comprehensive way – unlike indicators based on individual aspects such as recycling – and including aspects not well considered before, such as waste prevention. The framework is composed by a set of six single indicators, concerning the three dimensions of sustainability: reduction of effectively landfilled MSW and reduction of MW generation (environmental component), balance between expenses and revenues and reduction of costs (economic component), accessibility to separate collection and number of complaints (social component). Each indicator consists on an evaluation of the current status of the variable in contrast to a previous situation, with positive value in case of improvement or negative in case of decline. Then the values of the individual indicators are combined to obtain a global result. This approach focuses on dynamic progress towards sustainability, complementing the common static indicators. Contrarily to the existing performance indicators schemes, the proposed framework aims at measuring the progress and not the absolute or relative achievement of a waste management utility. The framework was tested for two Portuguese municipalities, proving to be of straightforward application and reliable in guiding stakeholders. Results for the case study showed good performance on economic sustainability, while environmental and social performance were lower due to a lack of strategies for waste prevention and low source separation, affected by poor accessibility to separate collection.

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[3] Further details provided in the List of publications included (page xxxvii).

3.1. INTRODUCTION

The incorporation of environmental sustainability as a relevant dimension is increasingly becoming an essential practice when addressing the integrate management of municipal waste (MSW) as a whole system, in the context of shift towards a more circular economy (Pires et al., 2011). For instance, the policies promoted by European Union (EU) on waste management have been orientated to meet this requirement (European Commission, 2014, European Parliament and Council, 2008). As a consequence, the need for analysing and evaluating the performance of the MSW management systems has been gaining significance (Pires et al., 2011). Outside of the EU and other high income nations, developing countries are facing an even more urgent challenge to assess their waste management practices, due to the fast growing urbanisation processes taking place in those countries (Karak et al., 2012).

These performance analyses have been typically linked with the use of indicators in order to communicate results (Ferreira et al., 2017). The use of indicators was long ago established in the economic and social fields and was later extended (after the 1992 Rio de Janeiro Conference on Environment and Development) to the environmental studies and, particularly, waste management (Ristić, 2005). Several innovative proposals on the use of performance indicators for evaluation of MSW management systems have been published. Guimarães et al. (2010) suggested for Portugal an approach based on Balanced Scorecard methodology, later tested by Mendes et al. (2012, 2013). Armijo et al. (2011) proposed a set of indicators fitting in a DPSIR model (*Driving force-Pressure-State-Impact-Response*). Greene and Tonjes (2014) and Liu et al. (2017) developed sets of indicators and applied them to the particular contexts of USA and China, respectively; the former also developed a scoring system to evaluate the quality of the indicators. Wilson et al. (2012) analysed waste management performance and governance in twenty cities around the world, a work further updated to become the '*Wasteaware*' benchmark indicators (Wilson et al., 2015). More examples are found in a revision on the issue presented by Sanjeevi and Shahabudeen (2015).

Indicators can be understood as the result of "compacting" a large amount of information in order to handle it in a manageable format, which allows to quickly realising whether or not a given objective within a project is being reached as expected, and to which extent. In fact, indicators are often the way to transfer to a general public –regardless of technical background– the results supplied by more complex tools. Several of these tools have been used to study MSW management, with a significant prevalence of Life Cycle Assessment (LCA) (Pires et al., 2011), and also Life Cycle Cost (LCC) (Sousa et al., 2018). LCA has currently become a common, well-established approach to assess the performance of waste management, primarily regarding the environmental dimension – a thorough review of applications in this field is given by Laurent et al. (2014a, b). However, it requires a costly process of elaboration and, as pointed out by Blengini et al. (2012), it is not easy to understand for non-experts. Therefore, the diverse group of stakeholders involved in the waste management planning is not always fully aware of the potential of this analysing tool. This is especially relevant for administrators responsible for taking decisions, who tend to rely more on financial considerations than in environmental benefits (Blengini et al., 2012).

Without overcoming the importance of more complex analysis such as LCA, the incorporation of an integrated indicator which could act as a simplified preview of conclusions about the sustainability of a given system is regarded as worth of interest. In recent times, there has been some contributions to such an idea of a simple and comprehensive waste management sustainability indicator, including the methodology based on metabolic perspective developed by Fragkou et al. (2010), the "Zero Waste Index" created by Zaman and Lehmann (2013) and tested by Zaman (2014) and the composite indicator proposed by Rigamonti et al. (2016). However, these valuable approaches still leave some questions open to further discussion, one among those being how to take into account the waste prevention efforts. Environmental sustainability can be only fully achieved if generation of waste is decoupled from economic activity; the mere existence of the "waste" concept is actually a sign of inefficiency. Also, this kind of global sustainability indicators should consequently pay attention too to the economic considerations and social issues related to MSW generation and management, even if they are only addressed in a brief manner. In this work a simplified sustainability set of indicators combining the three aspects environmental, economic and social is developed, with the aim of assessing the progress towards sustainability of the waste management system in a given municipality. This methodology was then applied to two Portuguese municipalities selected for study.

3.2. BUILDING THE FRAMEWORK

3.2.1. Overview

The proposed set of indicators was developed taking into account the three essential dimensions of sustainability: environmental, economic and social. Furthermore, instead of more common performance indicators reflecting a static picture of the actual situation of the analysed system – which are more intended to compare different alternative waste treatment options or for benchmarking purposes –, the approach followed was more focused on the conceptual view of dynamic progress towards sustainability of a waste management utility. Sustainability should be regarded as an ideal situation to be achieved through continuous improvement in every of the three dimensions. Therefore, the framework is organised in three components of progress: i) environmental; ii) economic; and iii) social. To represent each component, two single indicators were selected, resulting in a set of six indicators to compose the whole framework (Figure 3.1). The scores on the individual indicators were later aggregated to obtain a single result of overall progress. The framework is generalizable to any context, but it was designed thinking on low performing waste management utilities eager to improve. In this context, there will be a higher potential for improvement and the proposed framework will be more useful. For high performing waste management utilities, the framework will provide little information, unless a significant decline is observed. For low performing utilities with no effort to improve, the framework will be of little or no use.

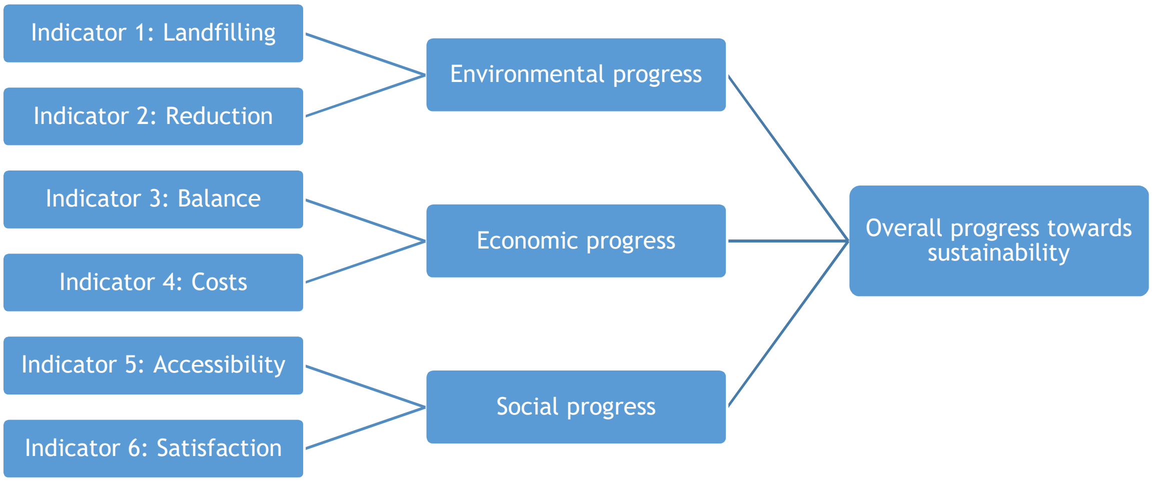

Figure 3.1. Scheme for the framework

The definition of each indicator is based on the same logic of comparison between current and previous states, which is shown, in a general form, by Equation 3.1:

$$\%\ \text{sustainability progress} = \frac{(\text{situation}_{\text{year i}-1} - \text{situation}_{\text{year i}})}{\text{situation}_{\text{year i}-1}} \cdot 100 \qquad \textbf{Equation 3.1}$$

In the proposed form, and also in the proposed indicators explained in the following sections, the comparison is made between the "current" year – actually, the last year with available data – and the previous one – actually, the previous year with available data, thus measuring a progress in time. The development presented herein adopts a yearly time scale, since this is a typical time interval for the organisations within the waste management sector to report results, but the user is free to adapt the indicators to other time periods, provided that the required data are available. As seen in Equation 3.1, the sign of the indicator provides a simple test to check if the system is progressing towards sustainability (if positive) or, on the contrary, moving away from it (if negative), along with a relative measure of how great the progress (or regression) has been. The variables for each indicator should be used in form of *per capita* values when this is applicable, to exclude the effect of population changes between different years. In mathematical terms, any performance indicator in the form of a ratio fails if the value in the denominator is zero ($\text{situation}_{\text{year i-1}} = 0$). If this happens, the indicator needs to be computed by one of the following options: i) if $\text{situation}_{\text{year i}}$ also equals to 0, then there is no progress and the indicator result is 0; or ii) if $\text{situation}_{\text{year i}}$ is not 0, then the indicator should take the value of -100%. The possibility of this situation will be assessed and discussed for each of the proposed indicators.

The ability of the framework to reflect the actual changes taking place in MSW generation and management essentially depends on the variables chosen to constitute the indicators making part of it. The variables were selected attending to: i) the feasibility of being calculated in the form of comparison proposed in Equation 3.1; ii) the availability of readily accessible data to calculate their correspondent values; and iii) their representativeness of the effects derived from the implementation of a given waste management policy. In this way, the indicators allow to evaluate the influence of these policies in altering waste flows or consumer behaviour. The selected variables and the indicators derived from them are described in detail in the following sections, for each of the three components of sustainability – environmental, economic and social.

3.2.2. Environmental component

A significant focus for the waste management utilities has been in enhancing separate collection and source separation of recyclable materials. However, within the field of environmental sustainability, it has been already pointed out that the ability to effectively decrease the amount of waste generated independently of the economic activity and wealth is also necessary to achieve the goal of true sustainability. Reduction of the environmental impacts associated with the production and consumption of goods and services will imply the adoption of more environmentally friendly practices by all the supply chain (manufacturers, distributors, retailers, consumers, waste managers, raw/recycled materials suppliers) as well as the consumers. Strengthening of Extended Responsibility schemes (regarding the use of packaging materials), and change of consumer patterns – for instance, through application of PAYT (*pay-as-you-throw*) strategies – may contribute to that change actually happening.

On the other hand, waste management entities have the responsibility of implementing the best available practices in accordance with environmental sustainability. Currently, the main guideline for efficient waste management in Europe is given by the waste management hierarchy. Despite some limitations (Van Ewijk and Stegemann, 2016), the waste management hierarchy has been widely accepted as a valid scheme of priorities between waste management options and was officially adopted by EU through the Waste Directive of 2008 (European Parliament and Council, 2008). Based on this hierarchy, which places landfilling as the worst option, in the last decades authorities have focused in improving their performance through the closure of landfills and the diversion of MSW to newly established facilities for recovery and valorisation: sorting facilities for recycling, mechanical-biological treatment (MBT) units, or Waste-to-Energy (WtE) plants. This has displaced the waste upwards in the waste hierarchy (EEA, 2013). Actually, except the upper categories of the waste hierarchy referred to prevention and reuse of waste materials, every other waste management option usually involves to some extent landfilling of non-recovered materials, either as direct disposal or as secondary disposal of materials rejected in the previous recovery activities. This indirect relationship of the waste hierarchy with landfilling activities is reflected in Figure 3.2:

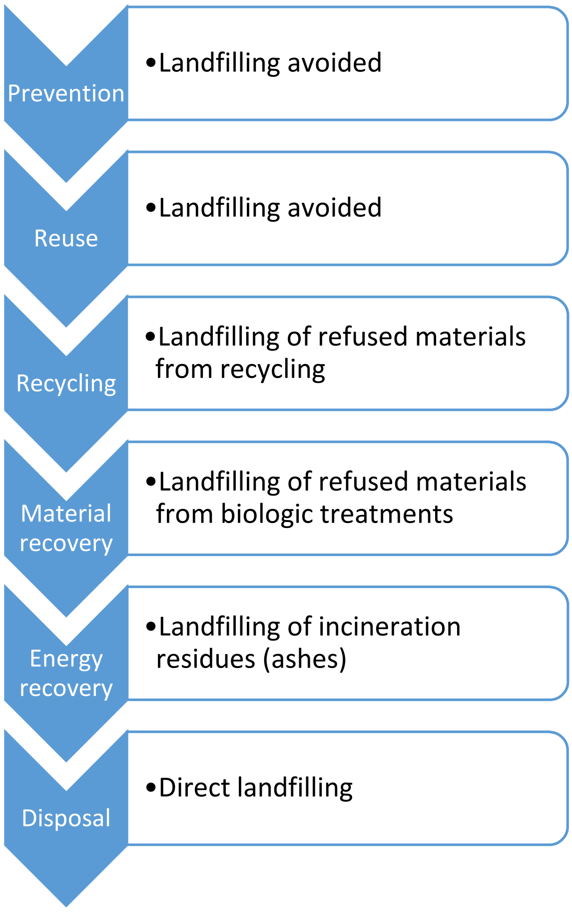

Figure 3.2. Relationship of waste hierarchy with landfilling

These progresses are usually evaluated by means of indicators which measure either the recycling rate or the waste diversion from landfill (Zaman and Lehmann, 2013). However, the use of such indicators can lead to some misconceptions. The use of recycling rates has been criticised for not being sensitive to the impacts of other treatment options than recycling (Greene and Tonjes, 2014). On the other hand, waste diversion figures are sometimes referred considering landfilling only as primary treatment option (i.e. direct disposal without any other treatment). In the case of mainland Portugal, while this *direct disposal* of MSW in landfills has been considerably reduced (from 62% in 2010 to 29% in 2016) (APA, 2017a), the *global disposal* in landfills counting also materials rejected after sorting, MBT or WtE is still high. The entities responsible for MSW management in mainland Portugal declared that in 2016 around 51% of waste treated went to landfill as final destination (APA, 2017b). This global landfilling rate is the selected variable for the first proposed environmental indicator, since it can represent not only the environmental performance of waste management systems – the more landfilling, the less sustainable – but also, to some extent, the variations in waste generation caused by reduction efforts and change on the behaviour of consumers.

Therefore, making use of the general formulation presented in Equation 3.1, an indicator about progress in reduction of final landfilling (Indicator 1) is presented in Equation 3.2:

$$\%\ \text{progress in landfilling reduction} = \frac{\left(\text{global MSW landfilled}_{\text{year i}-1} - \text{global MSW landfilled}_{\text{year i}}\right)}{\text{global MSW landfilled}_{\text{year i}-1}} \cdot 100 \qquad \textbf{Equation 3.2}$$

As explained in 3.2.1, a positive value in the indicator would mean less landfilling – and then, more sustainability –, while a negative value would indicate the opposite trend. As seen in Equation 3.2, a +100% of progress would be reached only if final landfilling in year i is non-existent (a “zero waste” scenario), while on the other extreme, negative progresses of -100% or even worse might be obtained if landfilling increases dramatically. Since a full circular economy implies the diversion of all waste from landfilling, this indicator may fail in mathematical terms in the future due to a zero denominator. However, this scenario means that there is no possibility for more positive progress and it is only possible to maintain (0 progress) or regress (-100% progress).

As explained before, the result of Equation 3.2 considers the possible decrease of MSW landfilled regardless of the reason behind that decrease – either MSW generation reduction efforts or MSW diverted from landfill towards recycling and valorisation processes. For instance: a system able to reduce MSW generation by 50% through reduction efforts without changing anything in treatment processes would obtain the same progress towards environmental sustainability as another system which improves by 50% its performance in waste collection and treatment – e.g. due to higher segregate waste collection ratios, better recovery of recyclables at the treatment plant, etc. – with the same MSW generation rates.

Nevertheless, the representativeness of the indicator is limited by the implicit assumption that MSW is a homogeneous flow with constant composition – something which obviously does not correspond to reality. Actually, the indicator is more sensitive to changes affecting the flow of mixed or residual MSW, since, as already mentioned, the most usual treatment options applied to this flow always involve to some extent landfilling, either as direct disposal or in indirect form. On the other hand, recyclable materials (which are separated at source, separately collected and directed to recycling processes) are not landfilled except for a small, refused fraction. For instance, in Portugal this refused fraction, made of non-recyclable materials, is estimated as roughly 5-10% of the MSW in separate collection flows. In this case the indicator will detect variations only to a limited extent. Take for instance a reduction of 100 t in the amount of mixed waste, which would result in a reduction of 51 t of the amount landfilled considering the present performance of the MSW treatment plants in Portugal. A reduction of the same 100 t in the recyclables fractions represents a reduction in landfilling of only 5-10 t. In other words: the proposed indicator is able to detect transfers of waste between the lower categories of the waste hierarchy – those more related with treatment of mixed MSW – and the upper categories, but it is less sensitive to transfers between reduction of MSW generation and the other high categories in the hierarchy – those related to reutilisation and recycling. Notwithstanding this, and taking into account that landfilling is still the predominant waste disposal method (Karak et al., 2012), the proposed indicator has still a remarkable potential to provide information about the evolution of MSW management in countries with poor separation rates.

In order to improve the ability to reflect changes between the upper categories of the waste hierarchy, a second environmental indicator (Indicator 2) was proposed as simply the progress in waste reduction based on the variation of the amounts of MSW generated, defined in analogous manner as Equation 3.2 and presented in Equation 3.3:

$$\%\ \text{progress in waste reduction} = \frac{(\text{MW generated}_{\text{year i}-1} - \text{MW generated}_{\text{year i}})}{\text{MW generated}_{\text{year i}-1}} \cdot 100 \qquad \textbf{Equation 3.3}$$

A positive value in the indicator would mean less waste generated, while a negative value would indicate the opposite trend. If the waste generated would become 0 there would be no need for waste management, so it is not logic for this indicator to fail in mathematical terms.

This second indicator complements the previous for assessing the overall goal of waste prevention, providing an indication of the waste efficiency of production, services and consumption. However, this indicator alone fails to capture a full circular economy – if a waste stream is fully recycled and forms a closed loop of production-consumption-recycling the amount of waste generated will be constant.

The two indicators proposed herein allow to address separately progresses in MSW reduction, which divert waste from all other categories towards the top of the hierarchy, and improvements in waste treatment, which simply divert waste from the lower towards the upper categories. Their combination provides a robust evaluation if a progress in the environmental dimension of sustainability is being achieved.

3.2.3. Economic component

The question of the financial sustainability of MSW management has been worthy of special attention in recent years. The current trend regarding this issue, encouraged by EU waste policy, is that financing of MSW management should rely on the "Polluter-Pays-Principle" (European Parliament and Council, 2008), thus assuring equity among citizens. In the case of Portugal, the practical consequence of this policy is that a regulatory board enforces municipalities to equilibrate all expenses and revenues related to the waste management service, making it self-sufficient in financial terms while excluding the possibility of obtaining economic benefit from waste tariff fees (Ministério do Ambiente, Ordenamento do Território e Energia, 2014). Even though this situation is still far from being accomplished, there has been a noticeable improvement from the situation reported in 2007, when Portuguese municipalities could only recover on average 30% of expenses on waste management (Simão Pires, 2013), to the situation in 2017, when the revenues accounted for 83% of the total costs (ERSAR, 2017). It should be noticed that there are large differences between municipalities: in 2017 only 25 municipalities (out of 278 in mainland Portugal) attested to have attained the required financial balance (ERSAR, 2017). In the cases where a deficit exists the remaining costs are subtracted from the municipal budget.

The same situation exists in countries where the cost of MSW management is still not recovered by charges. Nevertheless, and despite divergences in different national and local regulations, it can be stated in a general manner that the main driver in MSW financing should be to achieve this full balance between expenses and revenues, hence both deficit and profit situations are not desired. Therefore, it seems adequate to define the first indicator for assessing the economic component of sustainability (Indicator 3) as the evolution of MSW management financial balance, as presented in Equation 3.4:

$$\%\ \text{progress in financial balance} = \frac{|\text{revenue} - \text{expenses}|_{\text{year } i-1} - |\text{revenue} - \text{expenses}|_{\text{year } i}}{|\text{revenue} - \text{expenses}|_{\text{year } i-1}} \cdot 100 \qquad \textbf{Equation 3.4}$$

This equation gives a value different from zero if revenues and expenses are not equal. Therefore, by making use of absolute values, both deficit and profit situations are penalised and the indicator measures the progress towards the financial balance. Since the financial balance is possible and desirable to achieve, this indicator will most probably fail in mathematical terms due to a zero denominator, so the solutions presented in section 3.2.1 need to be used to compute it.

On the other hand, it seems clear that every reduction of costs can be understood as a positive contribution towards economic sustainability of waste management. In an analogous manner to the role of landfilling in the environmental indicator introduced by Equation 3.1, the improvements in operational efficiency can be tracked through the evolution of associated costs. This gives way to the second economic indicator (Indicator 4), presented in Equation 3.5:

$$\% \text{ progress in costs reduction} = \frac{\text{expenses}_{\text{year i}-1} - \text{expenses}_{\text{year i}}}{\text{expenses}_{\text{year i}-1}} \cdot 100 \qquad \text{Equation 3.5}$$

In a circular economy context, it is not impossible to assume that the waste management service may have no costs to the waste producers in the future, but there will always be expenses to provide the service, so the indicator does not fail in mathematical terms due to a zero denominator.

This definition presents the advantage of being sensitive to changes in pricing policy, as well as to other measures intended to improve savings or reduce costs, thus directly altering monetary flows, but it also takes into account the economic effect produced by changes on the MSW amount, i.e. a decrease in waste generation would lead to a reduction of the variable costs, e.g. in collection. In this context, Equation 3.4 measures the operational efficiency improvement in the waste management service and Equation 3.5 the operational efficiency improvement in the link between the waste management sector and the industry reusing waste. Moreover, in the approach suggested in Equation 3.5, a lower financial burden on MSW for municipalities could be interpreted as indicative of an improved efficiency in its management. This would be due either to a lower generation of waste or to a higher recovery of costs through sales of valuable materials or energy produced from waste. In fact, if waste materials are just considered as rejects with no economic value – a consequence of inefficient processes, as stated in Section 3.1 –, then the existence of an economic charge in form of a waste tariff is justified by the inevitable need for proper management and disposal of this waste. But if, on the contrary, waste is regarded as a source of valuable resources, suitable to be recovered and reused, then the cost of its management would be compensated by its economic value. So, instead of being subject to a charge, waste materials would be handled as goods with a market price. This already happens in the case of recyclable materials which are separated at source by consumers and sorted to be later sold to the entities responsible for the recycling. Actually, in 2011 it was estimated that recycling industry accounts for roughly a 1% of the EU Gross Domestic Product (GDP) (European Commission, 2011). In an ideal situation of zero waste and circular economy, the existence of waste tariffs would thus be unnecessary.

3.2.4. Social component

Social implications of waste management are probably the most difficult to clearly assess and, as such, still not widely researched (Yıldız-Geyhan et al., 2017). Municipal waste management is typically a complex system which involves different stakeholders and, not rarely, with opposite views between them (Blengini et al., 2012). Therefore, the link between processes and impacts is not straightforward – which, actually, is the usual situation in social impact assessments (Dreyer et al., 2006). Nevertheless, in the case of recycling there exists an agreement in considering that an increased recycling level is socially beneficial (Bio Intelligence Service, 2011, 2015, CE Delft, 2013). These social benefits derived from recycling activities are mainly focused on the development of recycling industry with a subsequent creation of jobs and contribution to economy. This focus on employment opportunities is a common approach to characterise social benefit (European Commission, 2018, Ferrão et al., 2014). However, reliable data on number of real employments created by waste management policies are not always available. In fact, depending on where the recycling facilities are located, these new jobs may be created in geographical areas far from the waste is being generated and, as such, may be out of the scope of the study (e.g. transfers of waste from Europe to East and South Asia, or to other developing countries). Moreover, jobs creation is just only one of the variables of concern when assessing social performance. Other less quantifiable dimensions, such as satisfaction of the population with the waste management system or engagement of people with more environmentally friendly attitudes and practices, play a key role for the success of a given waste management policy (Dunne et al., 2008; Jones et al., 2010; Triguero et al., 2016).

In view of this complicated situation, it is necessary to establish priorities among the many existent social perspectives and variables. According to the UNEP & SETAC guidelines for social life-cycle assessment, this prioritising is a necessary step when assessing social impacts (UNEP/SETAC Life Cycle Initiative, 2009). As already mentioned, the involvement of the local community is essential for tackling the waste problem, thus assuming it as a collective issue rather than a "problem of no one". Therefore, the preference in this work was to focus on the perspective of the population – the users of the system. These users interact with the waste management system mainly through their access to the waste collection service in order to dispose of the waste they produce. Hence, the indicators selected herein to represent the social component of sustainability refer to this interaction. Aiming at facilitating the data collection and enabling the practical use of the proposed indicators, it was decided to make use of information which, in Portugal as in other European countries, is readily available either through the environmental authorities reports or directly from municipalities.

Firstly, as indicated by Wilson et al. (2015), the existence of the MSW collection service is mainly justified to prevent public health problems derived from waste presence. Thus, in order to be really effective in meeting this target, the service must reach all citizens. The waste collection coverage, defined as the percentage of households which are adequately served by the waste collection service, is a common social performance indicator of MSW management systems (Wilson et al., 2015). For instance, in Portugal is regularly reported by every municipality. According to Portuguese environmental authorities, to be "adequately served by the waste collection service" means in practice that the waste bins must be placed at a distance of not more than 100 m from every household in urban locations, or 200 m in rural ones. In 2017, the average level of coverage in mainland Portugal was 87% for the mixed MSW collection service, but only 58% for separate collection schemes (ERSAR, 2017). Considering the room for improvement which is still needed to fulfil the 100% target, particularly in terms of expanding access to separate collection service, the progress of this coverage is selected as the first social single indicator (Indicator 5; Equation 3.6):

$$\text{\% progress in acessibility to separate collection} = \frac{(\text{coverage}_{\text{year i}} - \text{coverage}_{\text{year i}-1})}{\text{coverage}_{\text{year i}-1}} \cdot 100 \qquad \textbf{Equation 3.6}$$

It is noteworthy to remark that in this case, the order of operands in numerator of Equation 3.6 was reversed relative to the other indicators, in order to keep the same criterion for signs: positive for improvements, negative for decline. Moreover: by definition, if there is a waste management service its coverage should not be zero. Therefore, this indicator does not fail in mathematical terms due to a zero denominator.

Aiming at a possible generalisation of this framework to other geographical contexts, it is likely that the accessibility to the collection service might be evaluated in a different manner for every place: in the worst case, the question would to which extent a regular MSW collection service exists. Therefore, the precise definition of accessibility and thus the way of calculating the indicator, should be adapted to each particular situation.

Secondly, in line with the previous explanations, it was deemed convenient to somehow evaluate the satisfaction of users with the MSW collection service. In Portugal, this information is currently not directly available, since no regular surveys are performed and a satisfaction indicator as such is not part of the published performance reports. However, municipalities keep records of all complaints and suggestions posed by citizens. Hence, the number of complaints related to MSW collection was thought to be a reasonable general proxy indicator of the satisfaction with the system, so that a lower number of complaints would imply a higher level of satisfaction. In this way, the second social single indicator (Indicator 6) is defined in Equation 3.7 as the progress in reduction of the number of registered complaints:

$$\%\ \text{progress in complaints reduction} = \frac{(\text{complaints}_{\text{year i}} - \text{complaints}_{\text{year i}-1})}{\text{complaints}_{\text{year i}-1}} \cdot 100 \qquad \textbf{Equation 3.7}$$

Considering that the goal of any organisation is to have zero complaints, this indicator is highly prone to fail in mathematical terms due to a zero denominator. Again, the solutions presented in section 3.2.1 need to be used to compute it.

With these two indicators, it becomes possible to assess the progress towards social sustainability for the local waste management system through the two evaluated aspects: if the waste management service is advancing on the accomplishment of its primary social function of preserving public health and safety, and if it is doing so in a satisfactory manner.

3.3. Practical application

3.3.1. Case study

Two Portuguese municipalities – purposely named A and B – of different size and characteristics were selected as case of studies for demonstrating the of application of the proposed framework for evaluating sustainability progress. Municipality A is located on the Atlantic coast and is a significant industrial, trade, academic and touristic hub. It can be considered a medium size city. Municipality B is a rural town in the inland, part of a bigger urban agglomeration. This municipality is mainly set on a rural environment, while the town itself acts as a local centre of services and possess also some small industries. Both municipalities are currently interested in establishing PAYT systems, hence their interest for evaluating the sustainability of their MSW management situation.

Both municipalities share the same MSW management structure. The mixed fraction of MSW is collected and treated on a mechanical-biological treatment (MBT) facility, where after recovery of recyclable materials the remaining organic-rich fraction is first anaerobically digested to produce biogas and then composted. There are also sorting centres for classification of recyclable materials from separate collection.

Data provided by the municipalities of MSW generation in last years are shown in Figure 3.3, for both mixed MSW and separate collected recyclable materials. In the previous years a significant decrease of the waste generation was observed from 2010 until a slight recovery from 2014 onwards. This trend is the same registered for the whole country and is explained by the 2008 economic crisis and beginning of recovery after 2014. It is also observed that separate collection is at a much lower level than mixed collection, reaching only roughly a 7% of the total amount in both municipalities A and B. This is a poor yield, even if compared with the national average (14.4%, see 3.2.2). Moreover, this value has remained mostly stagnated over the last years. Both negative figures suggest that the municipalities still have to do more efforts to face this problem.

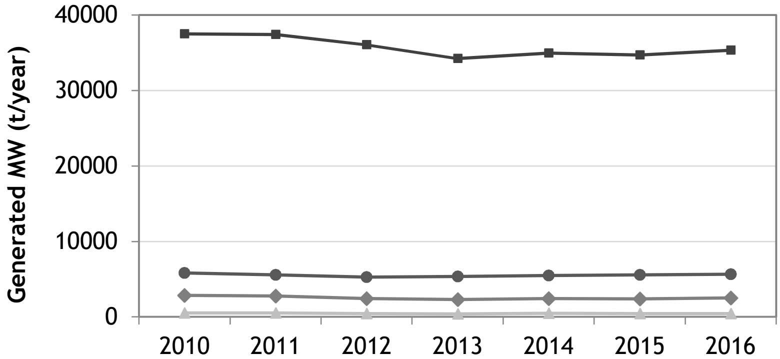

Figure 3.3. MSW generation in tonnes per year
(■ mixed MSW Municipality A; ♦ separate collection Municipality A; ● mixed MSW Municipality B; ▲ separate collection Municipality B)

3.4. RESULTS

As previously explained, the values for the six indicators of the framework were determined making use of the available information for the two selected municipalities.

3.4.1. Environmental results

For the indicators in environmental component, the results are shown in Table 3.1 for the period 2015–2016 and based on *per capita* values. The quantities of MSW are the ones annually reported by the managing entity; the share of global landfilling is reported only for the entire management system, serving 36 municipalities on the same region. Thus, the percentage of landfilling declared for the whole system was assumed to be the same for each municipality. Regarding the landfilling reduction indicator (Equation 3.2), for both municipalities the amount of MSW effectively landfilled increased, hence the indicator shows negative values. Also, a negative result – moving away from sustainability – is verified for the reduction indicator (Equation 3.3), since the total waste generation also increased.

Table 3.1. Results for environmental sustainability indicators

Municipality	Municipality A		Municipality B	
Year	2015	2016	2015	2016
MSW collected (kg per capita)	482.4	490.3	350.8	354.8
% of MSW globally landfilled	51.2	53.8	51.2	53.8
MSW globally landfilled (kg per capita)	247.0	263.8	179.6	190.9
Indicator 1: % progress in landfilling reduction	----	-6.8%	----	-6.3%
Indicator 2: % progress in waste reduction	----	-1.6%	----	-1.1%

According to the objectives set by Portuguese Government (Ministério do Ambiente, Ordenamento do Território e Energia, 2014), by 2020 the *per capita* generation of MSW should have decreased at least a 10% with respect to 2012. However, this objective is currently far from being met, since *per capita* municipal waste generation rate has increased 4.6% between 2012 and 2016 (INE, 2016). MSW generation has increased again in the last years, accompanying the recent economic growth, thus confirming that waste generation is still directly related to economic activity. A second circumstance might be also the inefficient performance of MBT facilities, which might be partly attributed to the low "quality" of the incoming waste, consequence of the poor degree of source separation and the lack of customers for the sale of refuse-derived fuel (RDF).

If economic recovery continues in the future, this increasing trend of generation will probably follow the same path, therefore moving away from the desired goal of sustainability.

3.4.2. Economic results

In the case of economic sustainability, the results are presented in Table 3.2. Data for expenses and revenues were retrieved from Portuguese National Statistics Institute (INE, 2016). In both municipalities the results of the indicators show that costs have been reduced, despite the increase in MSW generation. Hence, it can be concluded that economic efficiency has improved in the period considered.

Table 3.2. Results for economic sustainability

Municipality	Municipality A		Municipality B	
Year	2015	2016	2015	2016
Expenses (€ per capita)	54.0	49.6	24.4	23.8
Revenues (€ per capita)	63.4	47.7	25.8	23.8
Indicator 3: % progress in financial balance	----	+79.8%	----	+100.0%
Indicator 4: % progress in costs reduction	----	+8.1%	----	+2.5%

With regard to financial balance, relevant progress has been achieved: in 2015 both municipalities found themselves in a profit situation where the revenues from waste tariff were higher than the expenses, but in 2016 Municipality A has reversed the situation reducing both expenses and revenues up to a point of slight deficit, while Municipality B has reached the objective of financial balance. This progress is reflected on the value of the correspondent indicator, which shows a result of 100% progress. According to the definition of Equation 3.4, this result means that the gap from the initial situation has been fully overcome, so that sustainability in financial balance has been attained. In this situation, no further progress could be expected in the future, other than preserving the current optimal situation. This corresponds to the particular limit situation explained in section 3.2.1.

3.4.3. Social results

Finally, for the social dimension the values for both progress in accessibility to separate collection and number of complaints are shown in Table 3.3 for Municipality A and Municipality B. The first one showed clear negative results for both municipalities. The reason behind this trend are the changes applied by the Portuguese regulatory body to the methodology for determining the degree of collection coverage, which aimed at obtaining a more realistic picture of the situation. In general, the revised values widely showed poorer accessibility to the service than previously thought. This lack of accessibility has been pointed as a relevant reason for the low performance of separate collection in Portugal and other European countries (Struk, 2017; Oliveira et al., 2018). This fact opens a room for improvement by the municipalities in the next years.

Regarding the evolution in the number of complaints, the results differ between both municipalities: while in Municipality A they have decreased, in Municipality B there was an augment between 2015 and 2016 and, as seen in the information of Table 3.3 the relative number of complaints has surpassed that of Municipality A. A large part of these complaints are about the placement of MSW road-side containers, so this indicator is partially related with the previous one dealing with accessibility to the collection service. The rest of complaints are mostly about the general quality of the collection service and issues related with the tariff.

Table 3.3. Results for social sustainability

Municipality	Municipality A		Municipality B	
Year	2015	2016	2015	2016
% separate collection coverage	74%	42%	57%	39%
Number of complaints received	31	23	5	9
No. complaints / 10,000 inhabitants	4.03	2.98	2.87	5.15
Indicator 5: % progress in accessibility to separate collection	----	-43.2%	----	-31.6%
Indicator 6: % progress in complaints reduction	----	+26.2%	----	-79.3%

3.4.4. Global progress

The calculated values for the six indicators are jointly shown in Figure 3.4, which can be understood as a combined representation of the overall progress towards sustainability of waste management in the two municipalities A and B. From this figure it can be concluded that, while the economic situation of waste management in the municipalities is advancing in the good direction, i.e. achieving a balanced situation and reducing costs, this is not happening in the other two sustainability dimensions. In the environmental side, prevention measures are still not widespread, so that MSW generation is still dependent on economic cycles. Moreover, the flows of materials are not moving in closed loops, as would be desirable from a perspective of circular economy. The low degree of source separation in these municipalities is a major cause for this poor performance. This results in more MSW sent to landfill, with associated environmental impacts. In the social aspects, even though in Municipality A complaints from citizens have decreased, new actions and measures are still required to improve the quality of the service. Particularly, the accessibility to the collection of recyclable materials needs to be further extended. This would also be helpful to improve the environmental performance by diverting more waste from landfilling.

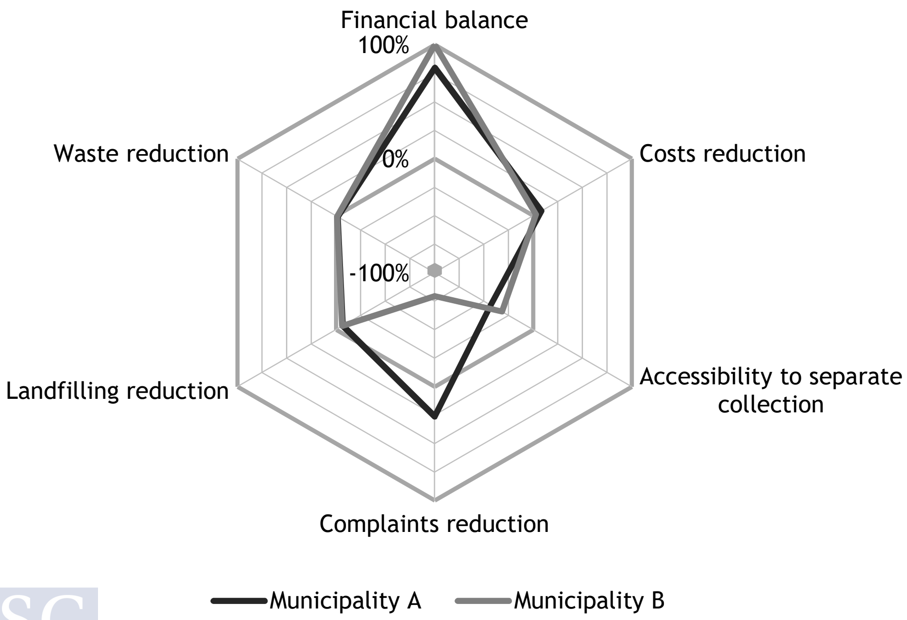

Figure 3.4. Combined results for progress towards sustainability

If the results of the six indicators are averaged, an integrated “sustainability progress index” is obtained, as seen in Table 3.4.

Table 3.4. Global sustainability progress between 2015 and 2016

Municipality	Municipality A	Municipality B
Sustainability progress	+10.4%	-2.6%

In view of the results in Table 3.4, it may be concluded that Municipality A has progressed positively during the considered period. It has improved by 10% in sustainability of MSW management, while Municipality B has worsened its situation by 3%. However, it would be necessary to discuss whether it is fair or not to attribute the same importance to every indicator. The result would probably be altered if some kind of weighing were assigned to each of the indicators in the set, in order to 'normalise' them. This normalisation step is usually regarded as a sensible matter which requires careful judgement based on experience (Rigamonti et al., 2016; Wilson et al., 2015). Moreover, the relative weight given to each variable is dependent on the context. For instance: as pointed by Greene and Tonjes (2014), preserving public health through the access to the MSW collection system might be a more urgent challenge in developing countries than in industrialised ones, where environmental issues are currently a more important concern because accessibility is already mostly assured. For the sake of simplicity, in this analysis it was decided to give the same weight to each indicator as a first assessment, leaving the question of a more accurate weighing procedure for future studies. Perhaps a transformation of the different impacts into monetary flows could be an appropriate way for normalisation.

3.5. CONCLUSIONS

The proposed framework for sustainability progress evaluation can be employed as a simple communicating tool to quickly assess the performance of the waste management system in a municipality, making use of publicly and readily available data. The results provide a clear picture of the progress towards sustainability and circular economy, thus helping to evaluate where the efforts should be focused.

The followed approach relies on calculations which, intentionally, are not excessively complicated. An open question (discussed in section 3.4.4) remains, when considering how the different results on the environmental, economic and social indicators could be numerically normalised and synthesised in a single "sustainability progress index" by some kind of weighting. This consideration about normalisation leads to another thorny question when addressing waste management assessments: how to ensure comparability between different cases? Besides the problem of comparing different sources of data, the specific features of each case play also a relevant role. The two municipalities chosen as case study in this work provide an adequate example: it seems clear that a city or urban area can count with more financial and technical resources to afford measures and policies directed to tackle the waste problem than a rural town. Therefore, the level of demanded performance achievements should be adapted to each situation – contrary to the current practice in Portugal of establishing common rankings of municipalities related to the level of accomplishment of general objectives, a subject which has been further discussed by Sousa et al. (2019). Nevertheless, the proposed framework is deemed as useful to inform if improvement efforts in MSW management are being done in the right direction, and how fast are they progressing.

Further testing of framework in new municipalities and in countries other than Portugal could be an interesting source for further improvements. Some other issues, such as the benefits of energy recovery from waste or a more thoroughly evaluation of the social implications may also be included in future updates of the study.

A final comment regarding the framework is that is not suitable for benchmarking purposes, since the progress is not an indicator of the absolute performance. In fact, worse performing utilities have a larger margin for improvement and could, therefore, present higher progress than waste management utilities already performing well (in theory, a perfect case scenario has no room for improvement and the progress will be zero). As such, the framework is particularly useful for contexts such as the Portuguese, where significant improvements have to be implemented to achieve sustainability in waste management.

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Section II.

Exploring PAYT potential through attributional Life Cycle Assessment

UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

4. Analysing the environmental role of waste collection in a Portuguese city

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Abstract

The purpose of this work is to assess the environmental impacts of the collection of mixed Municipal Solid Waste (MSW) in a selected neighbourhood of the Portuguese city of Aveiro. To this purpose, the main elements necessary for the collection process (carrier bags, dustbins, street containers and vehicles) were analysed applying the Life Cycle Assessment methodology, making use of locally gathered data. The main impacts associated with this activity are mostly related to the use of polyethylene plastic bags to carry the waste from the household to the waste container, and to the fuel consumption of collection vehicle that picks MSW from street containers and transport it to the treatment facility. The impacts associated to the plastic bags were primarily due to their disposal in a sanitary landfill after use and secondarily to the consumption of fossil raw materials required for their production. Given the relative high impact of the plastic bags, alternative scenarios were tested: using bags entirely produced with recycled polyethylene and bags produced with bio-based plastics derived from starch (TPS) and from wastewater (PHA). PHA bio-based bags were found to perform slightly better than conventional HDPE bags, but HDPE bags with 100% recycled content remained as the environmentally best option. A sensitivity analysis was performed to check the influence of bag size. Regarding the fuel consumption by collection vehicles, a comparison was performed to check how site-specific conditions can influence the impact of this activity, resulting in remarkably higher consumptions when local data were used instead of reference databases.

[4] Further details provided in the List of publications included (page xxxvii).

4.1. INTRODUCTION

Among the three main stages of Municipal Solid Waste (MSW) management – namely: collection, transport and treatment or disposal – collection is typically considered as a critical activity with regard to the costs derived from the fuel consumption of collection vehicles (Nguyen and Wilson, 2010; Faccio et al., 2011; Teixeira et al., 2014b). It has been reported that in Portugal, 79% of environmental budget in municipalities is spent on waste collection (Teixeira et al., 2014a). Performance of MSW collection can be significantly affected by a deficient understanding of the complexity of factors governing waste collection, which may result in poor planning and lack of adequate infrastructure (Abarca Guerrero et al., 2013). These problems are even more relevant in the case of urban areas located in developing countries (Hazra and Goel, 2009). Hence the need for assessing and benchmarking MSW collection systems, including their environmental implications. Specifically in Portugal, a number of studies have been published focusing on this topic: Teixeira, Avelino, et al. (2014); Teixeira, Russo, et al. (2014); Rodrigues, Martinho and Pires (2016a, 2016b); Ferreira et al. (2017); Martinho et al. (2017); Pires et al. (2017), as a necessary step in order to locate possible inefficiencies and ways for potential improvement. Likewise, economic implications of MSW collection in Portugal have been also studied (Sousa et al., 2018).

Among the existent techniques to evaluate MSW management, Life Cycle Assessment (LCA) has become a common methodology to analyse performance of waste management, thanks to its capacity to identify the environmentally critical points within a given process (Pires et al., 2011). This ability has been highlighted by an increasing number of applications (Laurent et al., 2014a, b). In the specific case of MSW collection, LCA has been applied for comparison of different collection systems (Iriarte et al., 2009; Punkkinen et al., 2012; Aranda Usón et al., 2013) or analysis of a particular element of the system, such as container types (Rives et al., 2010) or fuel consumption by vehicles (Larsen et al., 2009; Nguyen and Wilson, 2010).

Although there have been proposals of models to facilitate the assessment of MSW collection (de Oliveira Simonetto and Borenstein, 2007; Faccio et al., 2011; Bala Gala et al., 2015b), limitations of models in terms of LCA have been signalled (Winkler and Bilitewski, 2007; Teixeira et al., 2014b; Bala Gala et al., 2015b). In parallel, the trend to disregard the environmental impacts of the MSW collection stage or address them only in a superficial manner, given its minor relevance when compared to the whole management system, has been also criticised (Laurent et al., 2014b). In fact, it has already been attested the dependence of MSW management performance on local specificities and particular contexts (Passarini et al., 2011; Oliveira et al., 2017), which make of every collection system a singular case (Teixeira et al., 2014a). Therefore, supporting the analysis upon valid data gathered at local level appears as an essential procedure to ensure the reliability of the study (Ripa et al., 2017).

In contrast to previous studies focusing only in one aspect – like fuel consumption – or comparing alternatives only for some features of the system ignoring common parts, the main purpose of this work is to perform a complete assessment of the MSW collection process – from the generation of waste until its delivery to the treatment facility – in an integral manner not easily found before in the published literature. All the main elements involved in this scheme were analysed in terms of environmental impacts: carrier bags, household dustbins, street containers and collection vehicles. Special focus was put on the use of plastic carrier bags for transporting the waste, attending to the great concern currently raised on the environmental consequences of the use of plastics. In the case of European Union (EU), an integration of plastics within circular economy is currently being promoted (EC, 2018). This has led to penalising the consumption of single-use plastic items such as lightweight bags (EP, 2015), but this regulation does not consider the use of plastic bags as household waste carriers. In the case of Portugal, while the consumption of lightweight plastic bags in supermarkets has decreased since the adoption in 2015 of a tax penalising them, consumers have shifted to the use of specific household waste bags to put their rubbish in, as highlighted in a study by Martinho, Balaia and Pires (2017). Although comparative analyses of plastic bags and bags of alternative materials intended to substitute them have been already published (James and Grant, 2005; Hyder Consulting, 2007; UK Environment Agency, 2011; Saibuatrong et al., 2017; Vendries et al., 2018), no assessment of the relevance of the environmental impacts associated to plastic bags within the context of MSW collection has been made. The second relevant aspect highlighted in this work is the comparison of fuel consumption rates based on real data to the values found in alternative sources – namely ecoinvent database for life cycle inventories, analysing to which extent the source used for this parameter can influence the global LCA results of MSW collection.

4.2. Data and methodology

4.2.1. Goal and scope

4.2.1.1. Case study description and expected goals

In the present work, the LCA methodology was applied to analyse the residual MSW collection for a particular residential neighbourhood in the Portuguese city of Aveiro. The city municipal council wished to join the LIFE PAYT project for implementing a PAYT/UBP waste pricing scheme, in an attempt to improve their source separation for recycling rates.

The city council designated a particular neighbourhood as pilot area for the first testing of the PAYT system. But, prior to the future implementation of the new pricing scheme, it was planned to perform an integral assessment of current MSW management in the city. The main goal of this assessment is to serve as an initial baseline for the municipality, intended to provide a basis of comparison for the future changes experienced in MSW management as a consequence of PAYT policy adoption. The work now presented corresponds to the first part of that broader assessment, focused on the collection of mixed MSW and specifically intended to assess the importance of each of the assets and activities making part of the MSW collection process in relation to its contribution to environmental impacts. Additional studies, focused in MSW treatment options will complete the integral assessment in the future.

Given the context regarding environmental assessment of MSW collection previously explained in the Introduction, it is expected that this effort of benchmarking might also be useful for LCA studies in other MSW collection systems thanks to the ability of LCA in identifying environmentally weak points and evaluating feasible alternatives.

4.2.1.2. Definition of the studied system

The MSW collection system currently existent in the area of study of this project consists in a kerbside collection based on collective street waste 1100 L containers. Household residents and merchants bring their rubbish into containers usually within plastic carrier bags. Actual collection takes place every night (except Sundays), with 20 m^3 of loading capacity waste collection lorries. This kind of collection scheme is widespread in economically developed areas such as Europe, either in a more manual form using rear loading lorries – as in this study – or with more automated side loading vehicles; although in last years, new technologies like pneumatic collection systems have also appeared. The system for residual MSW collection is complemented by a network of drop-off containers for separate collection of recyclable materials – not analysed in this study.

For the elaboration of the study, four assets were defined as main constituents of MSW collection process: plastic bags for storing and carrying waste, dustbins used to keep rubbish in the bags at households, street containers to put together the rubbish bags and finally, the actual collection and transport process using a MSW collection lorry. Other components than these – e.g. personal protective equipment worn by collection crews –, were not included in the study since their contribution to the overall environmental impact was considered to be negligible. Each of the studied assets was analysed as an independent element making part of the studied system. This system, along with the boundary of the present analysis, is represented in Figure 4.1, where the sequential relationship between the elements represents the route of the reference flow.

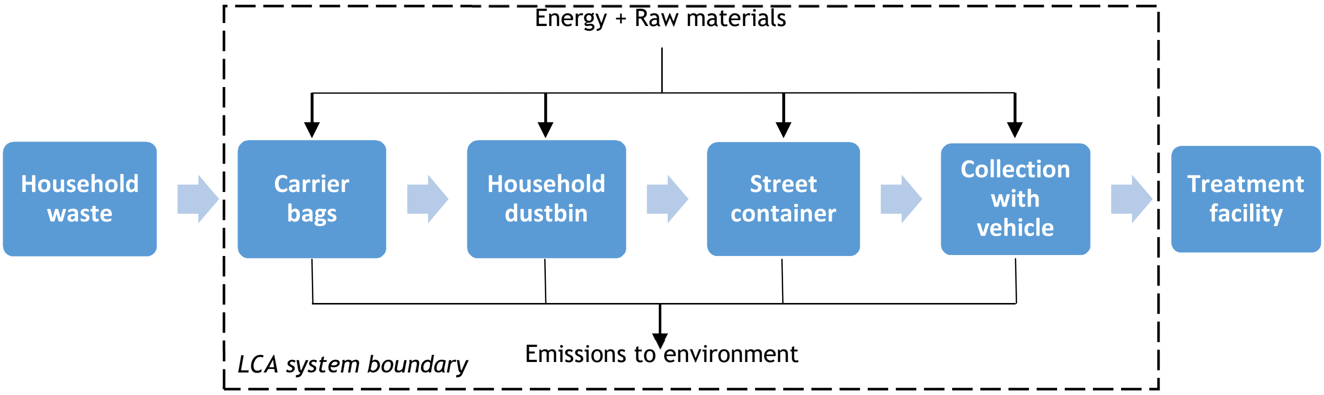

Figure 4.1. Representation of the analysed system with boundary of the study

Every unit process in this system requires consumption of energy and raw materials for its production stage and generates polluting emissions as a result of both production and final disposal, except for the case of collection and transport which is an activity rather than a product, so the consumption of energy and generation of emissions are due to the actual development of the activity – mainly fuel consumption.

4.2.1.3. Modelling framework and cut-off criteria

An attributional framework was chosen for the LCA modelling. Considering that the work included comparison of alternatives – namely different type of bags –, this would correspond to a type A goal decision-context situation, according to the ILCD handbook guidelines (JRC – IES, 2010). Concerning cut-off criteria, those amounts of materials sent to recycling as end-of-life (EoL) treatment option were counted as avoided waste, i.e. no environmental burden was allocated to the system originating the recycled material. Even when the analysed system was considered to provide a single function – i.e. collection of household waste –, a multifunctionality solving procedure through system expansion was needed to take into account the environmental credit derived from the use of bags made of recycled plastic, in accordance with the cut-off rule previously defined.

4.2.1.4. Functional unit and reference flow

The main function fulfilled by a MSW collection system like the one assessed here is supposed to be: providing citizens with an adequate waste collection service in order to ensure public health. In accordance with this function, the functional unit (FU) selected for the LCA was the total annual residual (mixed) mass of MSW collected within the considered neighbourhood. This annual residual MSW collected amount was calculated, basing on the field measurements obtained from a waste characterisation campaign which consisted in weighings of the daily amount of MSW collected in the area of study in ten selected days between 2017 and 2018. The values of weight obtained were extrapolated to a whole year by comparison with historical records kept by the Aveiro municipality. The resulting value corresponds to an annual mass of 442 tonnes of residual MSW collected in the studied neighbourhood. This number was assumed as reference flow to which all calculations are referred. Given that the population of the neighbourhood is estimated to be around 1200 inhabitants, this results in a *per capita* production of 368 kg per year. The choosing of one year as time frame for the functional unit was justified because it corresponds to a complete cycle of seasonal variation of the MSW generation pattern.

During the characterisation campaign the specific weight (apparent density) of residual MSW was experimentally found to be 75 kg/m³ on average – with a minimum value of 72 kg/m³ and a maximum of 118 kg/m³; the latter value was found only when street containers were filled at full load capacity, so probably there was a higher compaction inside containers. This value is somewhat lower compared to the typically reported range (Tchobanoglous and Kreith 2002; Karak et al. 2012; Teixeira et al. 2014b). This can affect the efficiency of collection, since vehicles would be full before reaching the maximum load capacity.

4.2.1.5. Data collection and methods of assessment

Data regarding MSW generation at local level and its characteristics were directly obtained from the characterisation campaign mentioned in subsection 4.2.1.4 and from a field survey performed in the area of study. Data concerning MSW collection activities were obtained from the municipality of Aveiro at city level and extrapolated for the area of study. All other data required for the assessment, concerning characteristics of analysed products, manufacturing activities or raw materials were obtained from the respective producers whenever possible and when not, completed with regional (Portuguese/European) or global scale data from life cycle databases – namely ecoinvent 3.3® (ecoinvent Association, 2016). All assumptions are explained in detail in the following sections.

The environmental assessment was performed using the commercial software SimaPro version 8.2.0. (PRé Consultants BV, 2016). The impact assessment method chosen was the ReCiPe Midpoint (hierarchist perspective) Version 1.12 (Huijbregts, et al., 2017), due to its completeness regarding the impact categories covered. Nevertheless, other common assessment methods were also tested, in order to ensure the robustness of the obtained results. Within the considered method, several impact categories were selected as being the most relevant regarding environmental impacts in the field of waste management: terrestrial acidification, climate change, metals depletion, fossil resources depletion, freshwater ecotoxicity, terrestrial ecotoxicity, human toxicity, freshwater eutrophication, particulate matter formation and photochemical oxidant formation. These categories were selected according to recent practice in previous studies on waste management (Andreasi Bassi et al. 2017; Pires et al. 2017; Ripa et al. 2017).

4.2.2. Life Cycle Inventory

Data concerning processes and activities involved by the MSW collection – as described in the previous section 4.2.1.5 – were gathered in order to build a Life Cycle Inventory (LCI). The amounts of main raw materials needed for the constituent unit processes are summarised in Table 4.1 for the carrier bags, household bins and street containers. Modelling for every of these units is further detailed in next subsections. The Portuguese electricity supply mix available in ecoinvent 3.3 was applied to production processes requiring a power input, such as extrusion or sorting of recyclables. The modelling process for collection activity and vehicle is explained in 4.2.2.4.

Table 4.1. LCI summary for bags, dustbins and street containers

Unit process	Weight per unit	Units per FU	Lifespan	Raw Materials	Amount per FU
Carrier bags	7.2 g	196,503	----	HDPE	1448 kg
Household bins	0.75 kg	776	7 years	Polypropylene	83.6 kg
Street containers	43 kg	26	14 years	HDPE	70.3 kg
				Steel	8.5 kg
				Rubber	2.8 kg

4.2.2.1. Waste carrier bags

It was assumed that all the household waste discarded to the collection system is carried out on plastic bags. The waste carrier bags were considered to be a single-use dispensable product, only intended to carry a single load of household waste between households and street containers. Although this assumption would be strictly true only for household waste bags specifically produced for this purpose – and thus not for other plastic bags originally produced for carrying goods and later reused for carrying waste (e.g. grocery bags of shops or supermarkets), it is thought that the function of carrying household waste would be anyway fulfilled by a plastic bag regardless of its origin – since this is the usual practice –, and therefore the plastic bag would have been produced in any case.

Plastic bags specifically intended for carrying household waste are usually made of high density polyethylene (HDPE) and produced by means of a film extrusion process – both the production of HDPE and the film extrusion are processes included in the ecoinvent 3.3 database (Hischier, 2007). Only the film extrusion process is assessed, so that other secondary processes, such as packaging, storing and selling of bags are excluded.

Given the different bag features, such as size, and also the different lifestyles of every household, it is difficult to assess how many bags are used to carry the residual MSW produced in the studied neighbourhood. For this study it was assumed a weight of 7.2 g for a plastic carrier bag with a typical carrying capacity of 30 L (60 cm for height, 50 cm for length, and 15.6 µm for thickness). It was also assumed that every bag would be delivered to the collection system only when it is full of rubbish, so that its maximum capacity of 30 L has been reached. This assumption is based on the fact that volume rather than weight is supposed to be the limiting factor governing the filling rate of bags with rubbish, in accordance with practice found in previous studies about plastic carrier bags (UK Environment Agency, 2011). Making use of the experimentally determined 75 kg/m^3 value of apparent density (for street containers), this means that every 2.25 kg of residual MSW would be carried by one single bag. Therefore, for the whole annual production of residual MSW in the neighbourhood 196,503 plastic bags totalling 1413 kg of HDPE would be needed. Nevertheless, a sensitivity test (discussed in subsection 4.3.2.2) was performed to check the influence of this assumption on the LCA results. Assuming that bags are produced in Portugal, a hypothetical transport distance of 200 km between the production site and the place of use was taken as typical situation.

For comparison between different bag types, two other carrier bags were modelled. First, a plastic bag with exactly the same features as the conventional one, but where the material was considered to be entirely recycled HDPE. Therefore, no raw materials are needed for the production of the bag, but an energy requirement of 0.6 kWh per kg of recycled HDPE – from ecoinvent database – and again, a road transport distance of 200 km were included as necessary for the recycling process. Secondly, it was also considered as an alternative to the oil-based plastics, the use of a biodegradable plastic bag. Among the different bio-based plastic types, thermoplastic starch (TPS, commercial name Mater-Bi™, produced by Novamont company in Italy), was selected as representative, given its well stablished commercial presence and the availability of data. A specific production process for this mixed bioplastic elaborated by the producer company is included in ecoinvent 3.3 (Althaus et al. 2007), and its environmental performance has also already been studied through LCA application (Shen and Patel 2008; Hottle et al. 2013; Yates and Barlow 2013; Spierling et al. 2018). In a simplified manner, this biodegradable plastic can be defined as a blend of 34% starch biopolymer – obtained from corn – and 66% polycaprolactone (PCL) with petrochemical origin, which acts as plasticiser (Gironi and Piemonte 2010, 2011). Carrier bags made of this material are also available in Portugal, even though to a limited extent. The raw polymer is produced in a factory located in Italy, and the bags are later manufactured by a Portuguese company. Thus, a road transport distance of 2550 km for the polymer followed by another road transport distance of 230 km for the bags are assumed. Also, electricity consumption for extrusion process was modified accordingly to the values found for another TPS material (Novamont S.p.A., 2001). Impacts due to construction and maintenance of the factory were excluded, since this is also the case for the HDPE inventory.

A summary of the features for the three alternatives of carrier bags is given in Table 4.2.

Table 4.2. Alternatives for carrier bags

Type of bag	Material	Capacity (L)	Weight (g)	End of life	Biodegradability in sanitary landfill (in 100 years)
Conventional plastic bag	HDPE	30	7.2	19% recycling 81% landfill	0%
100% recycled plastic bag	HDPE (recycled)	30	7.2	19% recycling 81% landfill	0%
TPS Bio-based bag	TPS/PCL	30	10.1	100% landfill	27%

Following usual practice when comparing bio-based plastics with fossil-based plastics (Yates and Barlow, 2013), it has also been considered that bioplastic bags require a higher amount of material in order to achieve the same performance as conventional HDPE bags regarding mechanical properties – as demonstrated by Davis (2003). For a bioplastic bag with the similar size of the HDPE bag, a weight of 10.1 g was assumed – following producer specifications for a 60 cm high, 50 cm long bag and 15 µm thick bag, hence a volume of 30 L (Novamont 2019). This is roughly a 40% heavier than the HDPE bag, which is justified on the one hand by the higher density of the TPS/PCL compound, and on the other hand by the higher tensile strength of HDPE.

Regarding EoL of plastic bags, in the particular case of the city of Aveiro residual MSW is treated in a Mechanical-Biological-Treatment (MBT) facility, so that during the mechanical treatment stage plastic materials are separated for recycling along with materials other than biowaste, prior to anaerobic digestion of the mostly remaining biowaste. The recovery level of plastic films from residual MSW was estimated to be around 19% – this was the number selected for calculations in this study, based on the recovered quantities publicly declared by the correspondent waste management company in 2015 (ERSUC, 2016). Accordingly, an equivalent production of primary HDPE was considered to have been avoided, after applying a correction factor of 0.9 to account for material losses during reprocessing (Gironi and Piemonte 2011) and further correction factors of 0.75 (Bala Gala et al. 2015a) to account for the loss of quality of the recycled material and 0.88 – taken from (OECD, 2018) – to account for the market share of recycled plastics, following the methodology proposed by Bala Gala et al. (2015a) to calculate the "environmental credit" derived from recycling. However, it is noteworthy to remark that this separation degree could be overestimated: technical difficulties for recycling these highly contaminated plastic materials imply troubles in finding possible buyers of this low value product. Thus the effective recovery of plastic film is losing interest for waste management entities. Furthermore, as a consequence of the recent ban (effective from January 2018) imposed by Chinese authorities on the import of certain plastics, MSW management systems in Europe are being forced to treat a larger amount of plastic wastes which otherwise would have been shipped to China for recycling (EC, 2018). In addition, there is also in Portugal a current lack of buyers of locally produced Refuse Derived Fuel (RDF) – which is a usual mean of valorisation of non-recyclable plastics –, due to the lower prices offered by foreign competitors selling their own RDF. As a result of this situation, a large part of plastic bags has currently as final destination the disposal in a sanitary landfill.

In the alternative case of the bio-based bags, it is assumed that these kinds of bags are not distinguishable from the more common conventional plastic bags. Even though its constituent material is theoretically suitable for a composting treatment, in practice the current system is not adapted for the separation of biodegradable from conventional bags, and these are assumed to be treated in the same way as the majority of rejected conventional plastic film, that is: disposed in a sanitary landfill. This implies an additional challenge: how to assess the behaviour of biodegradable plastics in a landfill environment. Several researchers have discussed the topic of degradation of bioplastics in different environments (Mohee et al. 2008; Gómez and Michel 2013; Emadian et al. 2017), including landfills (Cho et al. 2011; Adamcová and Vaverková 2016; Adamcová et al. 2017). In the case of landfilling, it seems that even though biodegradation is possible, it might be hindered by the existent conditions inside of the landfill: anaerobic environment, temperature, etc. There is currently no agreement about which degradation degree should be assigned to bioplastics, whit different choices found in published literature (Hottle et al. 2013; Yates and Barlow 2013). In the best case, bioplastics were found to be decomposed at the same rate than cellulose paper, which is often used in this kind of studies as a reference to verify the existence of favourable conditions to biodegradation. This is similar to the approach followed by Hottle et al. (2017), who employed as a proxy the landfilling process of packaging cardboard. According to these results and also for simplicity, it was chosen for the current assessment to assign to the bio-based waste carrier bag the same level of biodegradation after one hundred years given to average paper in ecoinvent database: 27% (Doka, 2007); (Table 4.2). For the modelling of TPS landfilling, the previously mentioned composition was employed to generate an own LCI using the calculation tool for waste disposal in landfill provided for ecoinvent 2.1 (Doka, 2008).

4.2.2.2. Dustbins

It was considered that every household has at least one dustbin intended for temporary keeping waste until discarding it in the street container. Given that there are 776 households on the considered neighbourhood (including in this number not only domestic households but also offices, restaurants, shops and other working places), the same number of dustbins is defined. An "average" household dustbin was defined as weighing 0.75 kg, being entirely made of polypropylene (PP) and produced through an injection moulding process. A hypothetical road transport distance of 100 km between the producing and the use places was defined. The lifetime was estimated to be 7 years, after which the bin is assumed to be sent to recycling. This modelling was adapted with own assumptions from a previous example (Martínez-Blanco et al. 2010).

4.2.2.3. Street waste containers

The municipality informed that in the area of study 26 street waste wheeled containers with 1100 L of nominal capacity are currently placed and available for residual MSW disposal by citizens. The containers are made mainly of high density polyethylene (HDPE) with some metallic parts and equipped with rubber wheels. Specifications of these containers regarding weight and construction materials were obtained from the supplier and completed with data from other research works (Bovea et al. 2010; Teixeira et al. 2014b). The following inventory data for one street container are shown in Table 4.3.

Table 4.3. LCI for one 1100 L street waste container

Raw Material	Production process	Weight (kg)
HDPE	Injection moulded	37.63
Steel	Mechanised (80%)	3.10
	Forged (20%)	0.78
Rubber	Injection moulded	1.50
TOTAL		**43.00**

Further inventories and impacts assessment for raw materials and production processes were taken from ecoinvent 3.3 database. In this case the road transport distance from the production factory to the city of Aveiro is known to be 276 km. The average lifetime of street containers is 14 years (information from municipality), after which the container was considered to be sent to recycling. Furthermore, it is estimated that 9.33 L of water per 1 tonne of MSW are spent in periodical street container cleanings (Iriarte et al. 2009).

4.2.2.4. Collection vehicle and collection activity

The collection of residual mixed MSW in the neighbourhood is done with diesel engine and three axles MSW collection lorries with a maximum load volume of 20 m^3 and payload of 15 tonnes. Every lorry is operated by one driver and two additional workers, responsible for hauling containers in and out the loading device of the lorry. For the assessment of production process of the lorry and end-of-life management the infrastructure process 'Waste collection lorry, 21 metric ton' existing in the ecoinvent 3.3 database (Doka, 2007) was used as proxy.

Regarding the MSW collection activity, the municipality provided performance data referring to the collection circuit where the studied neighbourhood is assigned. Collection takes place on a daily basis (six times per week), so that data consist on daily residual MSW quantities collected, daily distance travelled by the vehicle and monthly registers of fuel consumption, all referred to the period 2016–mid 2018. Based on these data the following performance rates, shown in Table 4.4, were calculated. The fuel consumption rate was calculated as an average of the monthly registered values (30 measurements) while the mean distance travelled was calculated as an average of the daily values, after removing outliers by making use of the Tukey's criterion based on the interquartile range, thus giving 757 valid measurements. Both values are presented along with correspondent standard deviations.

Table 4.4. Vehicle use, fuel consumption and travelled distance rates for MSW collection service

Performance variables of collection activity	Performance rate (per MSW collected)	Amount per FU
Collection lorry used	1.955·10-5 parts/tonne	8.643·10-3 parts
Diesel fuel consumed	4.23±0.27 L/tonne	1870 L
Distance travelled	5.84±1.11 km/tonne	2580 km

After applying these factors to the functional unit, it was possible to obtain the values of consumed fuel and transport distance necessary for the LCA. As discussed in section 4.3.3, the values of Table 4.4 are found to be highly dependent on the specific case analysed. Based on these calculated values, the air emissions included in the ecoinvent 3.3 process "municipal waste collection service by 21 metric ton lorry, RoW" (Doka, 2007) were adapted to the specific case study. This means that values for direct emissions related to the fuel consumption were modified accordingly to the fuel consumption and distance shown in Table 4.4. The same reasoning was followed with the material assets included in the same inventory: the road use and the waste collection lorry, except for the lifespan of lorry which was fixed in 10 years, corresponding to 286,722 km travelled, according to data reported by the municipality (ERSAR, 2017). On the other hand, indirect emissions of particulate matter (PM) related to the route features – mainly due to tyre wear, brake wear and road surface wear – were recalculated following the procedures recommended in the Tier 2 methodology of the EMEP/EEA guidebook (EMEP/EEA, 2016). Required parameters for this calculation were set as 50% for load factor (i.e. 7.5 tonnes, a valued applied to all calculations in accordance with the use of tkm, tonnes kilometre, as transport unit) – actually the lorry is empty at the beginning of the collection route and is progressively loaded until either reaching its maximum capacity or completing the route, thus a 50% loading level is considered a reasonable simplification – and 40 km/h for average travelling speed; speed would actually be lower during the stop & go driving phase corresponding to the collection of MSW containers, but no specific methodology is provided for calculation of PM indirect emissions in this situation, 40 km/h being the minimum average speed applicable. Other specific indirect emissions, namely heavy metals cadmium, lead and zinc with origin in tyre wear, were accounted through the emission factors proposed in ecoinvent (Doka, 2007): 0.0011 mg Cd, 0.0049 mg Pb and 3 mg Zn per tkm.

4.3. RESULTS AND DISCUSSION

4.3.1. Impact assessment

The results for the environmental impact assessment in the selected impact categories are shown in Table 4.5 in form of characterisation values, while Figure 4.2 shows the contribution of each unit process to the global impact, for every selected category.

Table 4.5. Characterisation of environmental impact

Impact category	Results per FU	Unit
Terrestrial acidification	46.8	kg SO2 eq.
Climate change	10740	kg CO2 eq.
Fossil depletion	4906	kg oil eq.
Metal depletion	331	kg Fe eq.
Freshwater ecotoxicity	122	kg p-DCB eq.
Terrestrial ecotoxicity	0.39	kg p-DCB eq.
Human toxicity	1425	kg p-DCB eq.
Freshwater eutrophication	0.78	kg P eq.
Particulate matter formation	19.1	kg PM10 eq.
Photochemical oxidant formation	77.3	kg NMVOC eq.

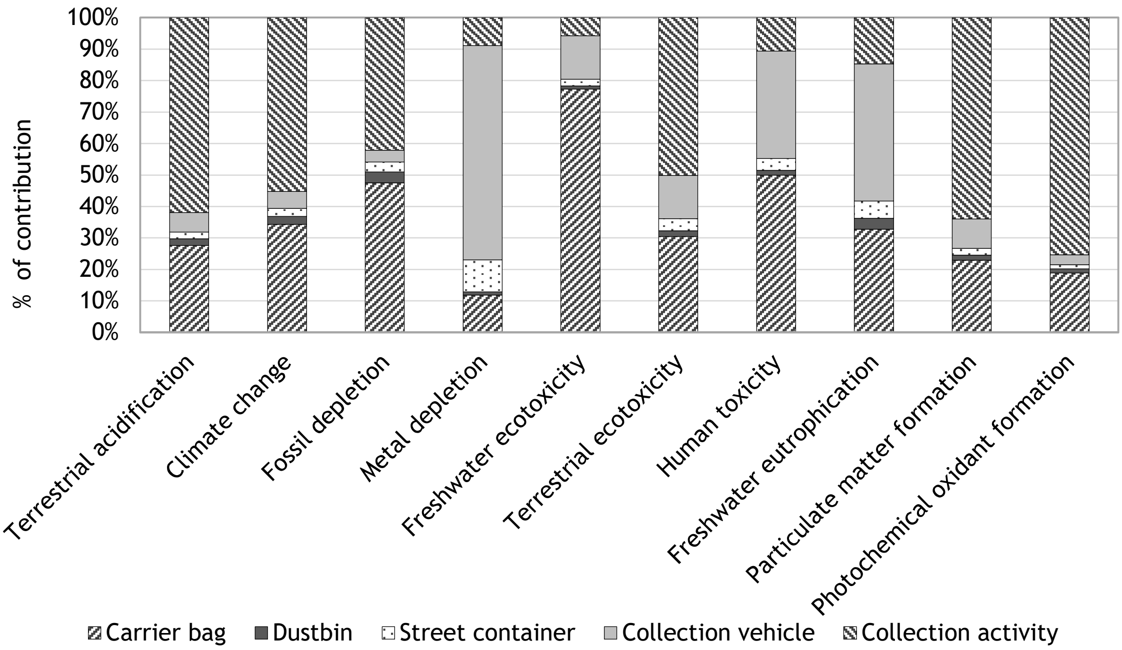

Figure 4.2. Contribution of unit processes to the environmental impacts within each category

The collection activity has a major contribution to environmental impacts in those categories more related to air pollution – terrestrial acidification, climate change, PM formation and photochemical oxidant formation – due to emissions originated from fuel consumption. Meanwhile, plastic bags are the top contributor regarding fossil depletion – due to raw materials consumption, freshwater ecotoxicity and human toxicity – due to landfilling at the end of useful life. Furthermore, the material assets dominate the category of metal depletion, due to raw metals consumption for their production – especially for the collection lorry, which is the same conclusion already indicated by Brogaard and Christensen (2012). Next, the freshwater eutrophication category relies mostly on the energy sources used for the industrial processes involved in the system, with the highest impacts being caused by coal mining. Finally, the terrestrial ecotoxicity category is largely explained by the indirect emissions derived from driving: brakes and tyres wear and road abrasion, which are relevant for the collection activity, but also for the transports of materials needed for production processes. From these results, it can be concluded that for almost all of the selected impact categories, the two main contributors to environmental impacts are the utilisation of plastic bags and the waste collection and transport activity with lorries.

The toxicity categories – with exception of terrestrial ecotoxicity – are clearly dominated by the impact derived of carrier bags (50% of total impact in human toxicity and 77% in freshwater ecotoxicity). Moreover, if a subsequent normalisation step was applied, these categories would be the highest contributors to global impact, particularly freshwater ecotoxicity. This high contribution is explained by the assessment of the environmental consequences attributed to the disposal in landfills of the bags, due to long-term emissions. These emissions, particularly those of heavy metals, are thought to be released mainly as leachates to freshwater systems. In a more detailed view, the assessment found the vanadium emissions to be the largest contribution from plastic bags to freshwater toxicity. These emissions were thought to be caused by the higher content of vanadium in polyethylene compared to other metals and also by its tendency to form oxyanions (vanadates), more likely to be released due to their increased solubility in the high pH environment found during the carbonate phase in landfills (Doka, 2007). Nonetheless, modelling of long-term emissions from landfills is usually a delicate issue and results should be regarded with caution.

In contrast to the other toxicity categories, the main responsible for impacts in terrestrial ecotoxicity seems to have been not the emissions from landfills, but, as already explained, the indirect emissions produced by collection vehicles. Anyway, this toxicity impact is much lower, compared to the ones caused by landfilling in the other two categories (freshwater ecotoxicity and human toxicity).

The other remarkable impact due to plastic carrier bags is found in the fossils depletion category. In this case, the relative high contribution associated to the plastic bags (almost 48% of total impact) can be explained by their relative short useful lifespan when compared to the other items: dustbin, waste containers and collection lorry. That causes the consumption of fossil raw materials and energy required for production of polyethylene to play a significant role, even higher than that of producing the fuels required for running collection lorries – which are optimised by using the vehicles to their full capacity.

4.3.2. Alternative scenarios for carrier bags

As indicated in 4.2.2.1, given the relative high impact of the plastic bags in some categories, alternative scenarios for waste carrier bags were tested. The influence of different bag features – i.e. constituent material and size – was checked.

4.3.2.1. Influence of material

Concerning the material of the bags, results for different alternatives are shown in Figure 4.3, namely for the comparison of conventional HDPE carrier bags with bags entirely produced with recycled polyethylene – which allowed to prevent the impacts related to production processes, but did not avoid the effects of disposal in landfill – and the use of bags produced with bioplastics derived from starch (described in 4.2.2.1) – which did not present such a harmful behaviour in landfill, but on the other hand are a source of major emissions as a consequence of agricultural practices to obtain the required starch and energy consumption for polymer fabrication.

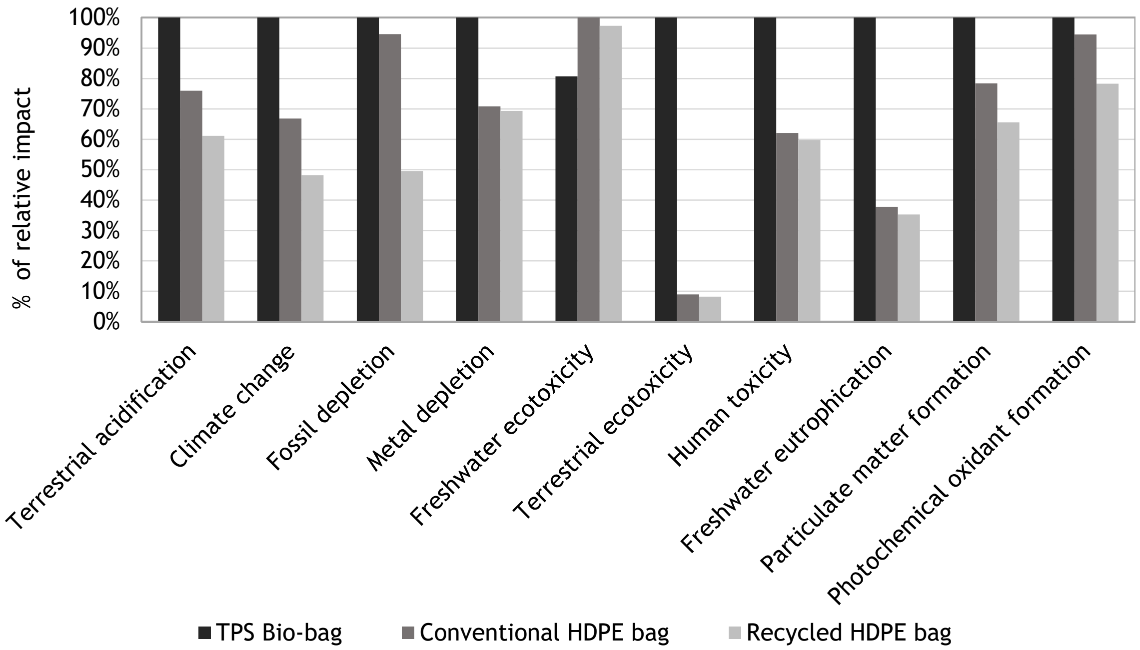

Figure 4.3. Comparison of relative collection impacts for different waste carrier bag materials (TPS Bio-bag = 100%)

In this case, it is clear that a modification of the constituent material of waste carrier bags may have relevant consequences on the impacts assessment. The analysis shows that the best alternative for carrier bags corresponds to the use of plastics entirely produced from recycling. In this way, the activities related to extraction and processing of primary raw materials might be to a great extent avoided – as seen in the fossils depletion category, where the impact of the recycled bags represents a 52% of the impact attributed to the conventional bags. With regard to the TPS bioplastic bags, in this assessment they are not able to compensate the impacts of the polyethylene bags due to their own production impacts. These are to a great extent originated by the cultivation methods employed to obtain the starch required for polymer fabrication. For instance, this is especially relevant in the terrestrial ecotoxicity impact category, where the impact of bioplastic bags is 11 times higher than that of the conventional bags. This has also an impact on human toxicity (61% higher than HDPE bags). In the case of TPS bags, the freshwater eutrophication category is also influenced by agricultural activities. The link to energy consumption observed in the HDPE bag is now not straightforward. In fact, although the energy required for obtaining the raw material is actually lower in the case of the biopolymer, energy consumption is also determined by the specific features of this particular case. For instance: different energy sources depending on the place of production and different transport distances. Also, the use of raw materials with origin in fossil resources as plasticisers for the biopolymer brings a negative impact related to the consumption of fossil resources and release of emissions during extraction and processing.

Concerning the contribution to climate change, it has to be noticed that the biodegradation of biopolymers in the anaerobic environment of landfills may contribute to the release into the atmosphere of greenhouse effect gases (GHG), comprising the fraction of landfill gas which cannot be captured by collection systems. This impact could be to some extent avoided if bioplastic bags were recovered for suitable organic valorisation treatments such as composting, however this procedure is currently not feasible since carrier bags are not separated at the waste sorting facility according to its material. Actually, at the study site the household biowaste is not segregated at source, but collected as part of residual MSW. Therefore, all separation of materials suitable for organic valorisation processes is done at the regional waste treatment facility by mechanical sorting, where the effectiveness of the separation is lower.

In the rest of categories, bioplastic bags appear always as the option with highest impact except in freshwater ecotoxicity, where it is concluded that the use of such bags can avoid the long-term environmental consequences of landfilling conventional plastic bags, but nonetheless this advantage for bioplastics is again mitigated by the emissions originated during agricultural activities, thus the difference is not so great (17% less impact than recycled bags). In fact, from a global perspective of the results, the use of bio-based bags does not constitute an environmentally better alternative, since their environmental performance is mostly offset by the benefits of recycling plastic. Recycled bags clearly contribute to saving of primary raw materials and energy, avoiding pollutant emissions.

In view of these results, it was decided to further test MSW collection with two alternative "improved" models of bio-based carrier bags, to check their ability to overcome the bottlenecks detected in the previous analysis. In the first improved bag – marked in Figure 4.4 as 'TPS Bio-bag (local, composted)' –, EoL management of carrier bags was changed from the current landfilling to a hypothetical composting treatment and, on the other hand, the biopolymer was assumed to be produced within Portugal, thus reducing the road transport to the same 200 km applied in the HDPE bag. The composting process was modelled following the same approach as Gironi and Piemonte (2011): 60% of carbon degradation, from which 95% is released as carbon dioxide – part with biogenic and part with fossil origin, accordingly to the TPS composition previously defined –, and the remaining 5% is released as methane. For the resulting mass of compost, an environmental credit was assigned, considering that compost (44% water content) can substitute peat in a 1:0.79 mass basis (Hermann et al. 2011).

Secondly, in order to avoid the impacts derived from agricultural practices and use of non-renewable resources, it was decided to test an alternative biopolymer which does not rely upon the use of neither agricultural outputs nor substances based on fossil resources as raw materials. Instead of that, a bioplastic based on the reutilisation of waste materials – hence, having no environmental burdens associated, was preferred. Therefore, TPS was substituted by another kind of bioplastics: polyhydroxyalkanoates (PHA). These compounds have also been the subject of thorough investigations, given that they can be obtained from the activity of mixed cultures of microorganisms which are able to develop in substrates such as fermented wastewater (Salehizadeh and Van Loosdrecht 2004; Serafim et al. 2008; Koller et al. 2017). In contrast to TPS, and even though it has been considered technically and economically viable (Gurieff and Lant 2007; Fernández-Dacosta et al. 2015; Morgan-Sagastume et al. 2015; Bengtsson et al. 2017), industrial production of PHA from wastewater is still in development (Rodriguez-Perez et al. 2018). The consequent scarcity of data makes difficult the establishment of a suitable inventory to be used in LCA, since most studies consist on upscaling of pilot experiences or adaptations of pure culture production processes based on clean substrates (Yates and Barlow 2013). PHA has higher tensile strength than TPS (Gross and Kalra 2002). By taking into account this difference, the hypothetical PHA waste carrier bag ('PHA Bio-bag' in Figure 4.4) would require less thickness – i.e. less amount of material – than the TPS bag to have the same carrying capacity. Hence, following the same procedure as Khoo et al. (2010), the weight of the PHA was accordingly calculated as 8.7 g. For building a Life Cycle Inventory, it was chosen to adopt the combination of processes which showed the best environmental performance in the study of Kendall (2012), which is in turn based on the work of Kalogo et al., (2007) for the hydrolysis stage and Harding et al. (2007) for the fermentation and subsequent extraction of the biopolymer. The inventory was adapted to the case of a mixed culture fed with wastewater – so there is no need of sterilising processes and no addition of process water. Addition of nutrients and trace salts was kept as in Harding et al. (2007). Although for a mixed culture based on wastewater they are likely unnecessary, their environmental impacts are anyway small. As stated before, the raw material is wastewater, free of environmental burdens. Accordingly, the wastes and further wastewater generated by the production of bioplastic are excluded from the assessment (except water for equipment washing), since the impact of their subsequent treatment is assigned to the original raw wastewater, following the first approach suggested by Heimersson et al. (2014). Again, the end of life of the bag was assumed to be composting – modelled according to the composition of the biopolymer, and the road transport was set as 200 km for the raw bioplastic and again 200 km for the finished bags. The extrusion process was kept the same as for HDPE.

Both "improved" bioplastic bag models were compared with the initial TPS bag and also with the conventional and recycled HDPE bags as shown in Figure 4.4. Additionally, the different contributions were aggregated to a single value for each bag case through the normalisation procedure provided by the ReCiPe methodology and presented in Figure 4.5 to enable a global comparison. Even though normalisation of results is usually considered a delicate step in LCA due to the difficult comparing of different impact categories, in this case it is considered worthy to take advantage of this tool to better present the results, and its use is justified because of the analogous function environmental profile of the items compared: plastic bags.

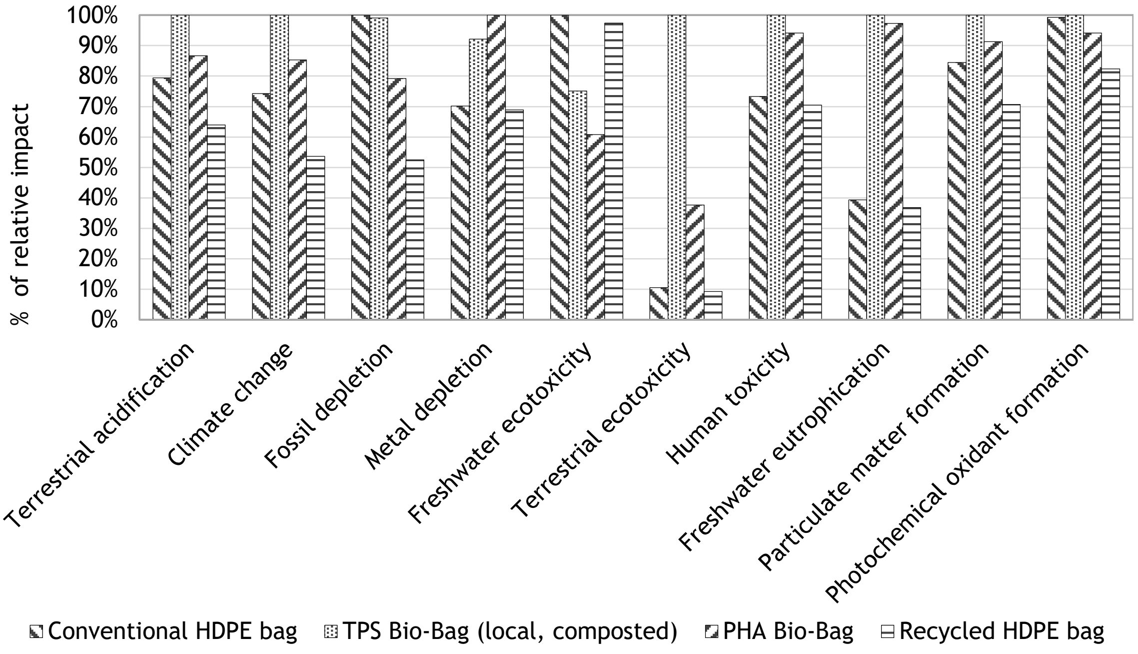

Figure 4.4. Comparison of relative collection impacts for improved alternative waste carrier bag materials (TPS Bio-bag = 100%)

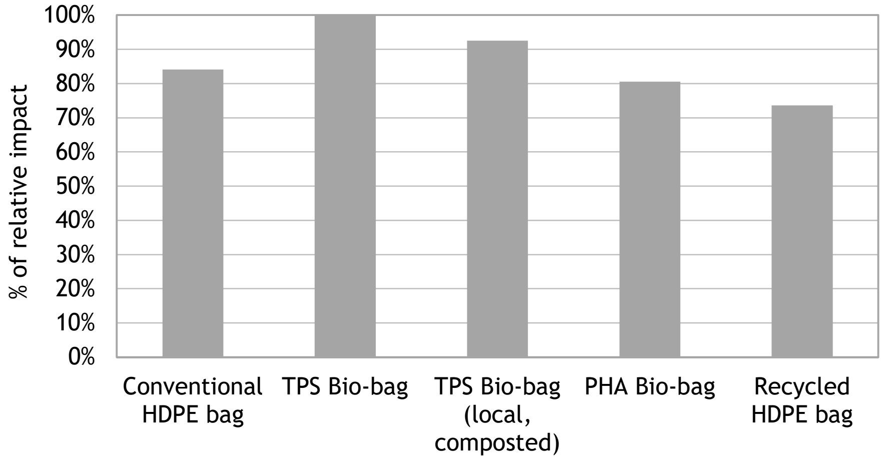

Figure 4.5. Comparison of relative normalised environmental impact assessment for collection with improved alternative waste carrier bag materials (TPS Bio-bag = 100%)

The compared assessment represented in Figure 4.5 shows that the composting treatment chosen for EoL option and the local production allows to reduce the environmental impacts of bioplastic bags by 7.5%, which is still not enough to compensate that of conventional HDPE bag (9% lower than composted TPS bags). In fact, most of the improvement came from transport reduction – which proved itself to be a sensitive variant – rather than from composting process. This finding supports the conclusion that, except for GHG emissions, composting of bioplastics does not bring great environmental benefits compared with other EoL treatments including landfilling (Hermann et al., 2011; Rossi et al., 2015; Franklin Associates, 2018). However, alternative bags made of PHA show slightly better impacts than conventional HDPE bags – actually, 4% less impact, but still worse than recycled HDPE bags (9% more impact). PHA bags have less impact than TPS bags in all the selected categories except metals depletion. Particularly in terrestrial ecotoxicity, the impact of PHA bags is 2.6 times lower, due to the absence of agricultural activities in the inventory. HDPE recycled bags remain as the environmentally best option, which is coincident with the results found in other studies (Gironi and Piemonte 2010, 2011; Hottle et al. 2017). This is explained by the lack of environmental burdens associated to obtaining raw materials, in contrast with bio-based bags, whose production involves always some kind of necessary processing for the elaboration of the polymer: through either agriculture (for starch) or industrial procedures, which in the case of PHA consist indeed in several stages (fermentation of substrate, polymerisation and extraction of the biopolymer).

In particular, the higher impacts showed by bio-based bags in terrestrial acidification and especially in freshwater eutrophication categories are in agreement with other authors (Harding et al. 2007; Gironi and Piemonte 2011; Hottle et al. 2013, 2017). Regarding GHG emissions, in this case neither TPS nor PHA are better than HDPE, but results from published literature show a broad range of values which allow for supporting different conclusions (Narodoslawsky et al. 2015). Something similar happens in human toxicity and PM formation, where both TPS and PHA bags present higher impacts than fossil-based bags, accordingly to Hottle et al. (2013) for TPS but contrarily to other sources (Harding et al., 2007; Franklin Associates, 2018). On the other hand, both TPS and PHA bags were found to give the best result regarding freshwater ecotoxicity – due to the absence of landfilling –, coincident with (Harding et al., 2007; Hottle, Bilec and Landis, 2017; Franklin Associates, 2018). Also, both bio-based bags require less fossil resources than fossil-based bags, which is a usual conclusion also in other works.

Moreover, PHA bags contribute less than conventional HDPE bags to photochemical oxidant formation, as indicated by Yates and Barlow (2013). An explanation for the discrepancies is probably to be found on the different assumptions and methodological choices made by authors. Given the different years of publication, it is also likely that LCIs – like those for fossil-based plastics – have been upgraded with time. Still, great uncertainty exists on the LCI of PHA production, mostly based on extrapolations rather than real data from production facilities.

4.3.2.2. Influence of size

The results obtained in LCA are dependent on the amount of polymer which is required to produce the carrier bag. This amount of material is reflected on the thickness of the walls of the bag, which should be calculated in order to assure that the bag is strong enough to carry without failure the intended weight of load during its useful lifetime. Actually, there exists a broad variety of bag models with different sizes and thicknesses, therefore the environmental assessment is always referred to the specific bag modelled within LCI. To check the influence of size, a comparison was made between four bag models corresponding to usual sizes found for waste carrier bags: 20 L, 30 L (the initially applied value), 50 L and 100 L. In each case the different weight of the bag and also its thickness were measured on real bags: 6.9 g and 14.8 µm for the 20 L bag, 7.2 g and 15.6 µm for the 30 L bag (as previously used), 13.3 g and 16.7 µm for the 50 L bag and 49.0 g and 31.8 µm for the 100 L bag. For the sake of simplicity, since the trend of the results found in the impact assessment was found to be the same for every impact category, the different contributions were again aggregated through normalisation for each bag size, as shown in Figure 4.6.

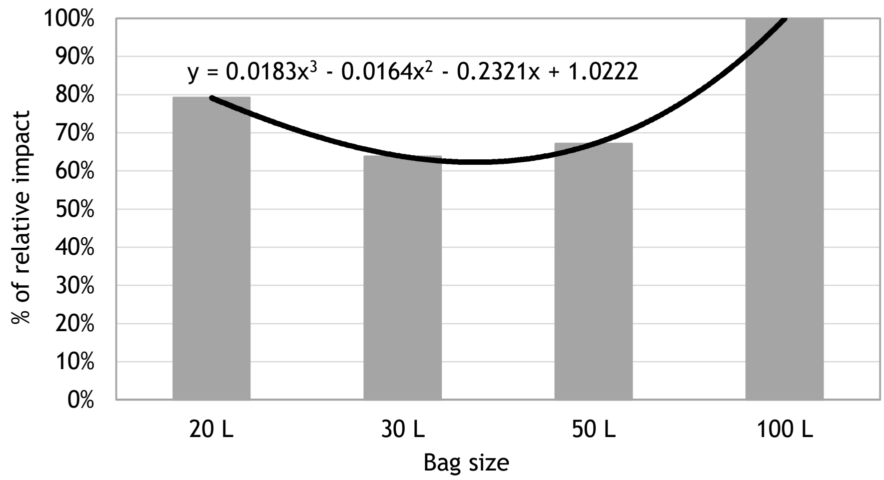

Figure 4.6. Comparison of relative normalised environmental impact assessment for collection with different waste carrier bag sizes and non-linear relationship obtained (100 L = 100%)

Smaller bags require a large number of units to carry the same amount of waste, while bags with bigger size need a larger thickness of plastic material in order to keep the same performance regarding mechanical properties, such as tensile strength, provided that they are intended to carry a heavier load. On average, the 30 L bags show 26% less impact than the 100 L bag, 11% less than the 20 L bag and 2% less than the 30 L bag. The impact category with the larger influence of bag size was found to be freshwater ecotoxicity, where the 30 L bags have up to 44% less impact than the 100 L bag, and on the other hand, metal depletion was the category with the smallest differences: 30 L bags show only 11% less impact than 100 L bags. The 100 L bag is clearly oversized for average households taking into account the daily collection frequency, but for the other bag sizes the differences are not so pronounced. The trend line in Figure 4.6 intends to represent a non-linear relationship linking the size of the carrier bags with their correspondent environmental impact. The line is fitted as a cubic function, whose equation is shown in the graph, and it shows that the bag with the intermediate size of 30 L seems to provide the optimal solution. However, the difference with the 50 L bag is so small that different models of 30 L bags with larger thicknesses will likely have more amount of material – hence, larger impacts – than the thinnest 50 L bags. Actually, the reasoning behind this behaviour lies in the amount of polymer required in order to produce a bag of a given volume, as shown in Figure 4.7 with analogous equation.

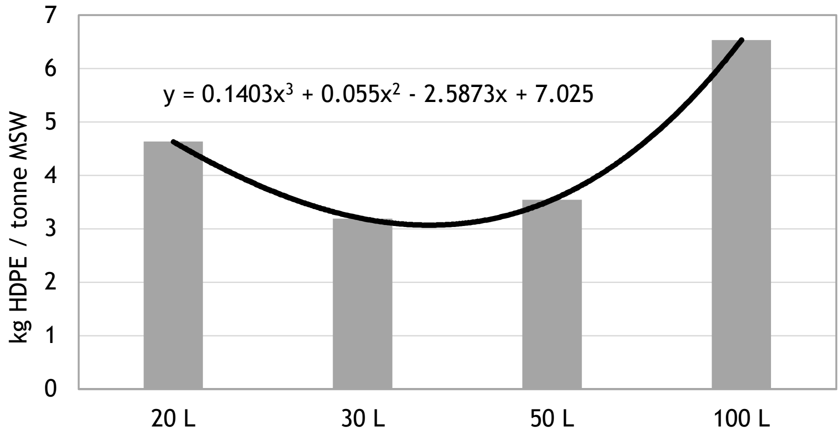

Figure 4.7. Relationship of bag volume with required mass of polymer and non-linear relationship obtained

Since the function of carrying household can be fulfilled by bags of any size and thickness, it is, therefore, relevant to take into account these features when assessing environmental performance of carrier bags, in order to obtain fair results.

4.3.3. Influence of the waste collection performance on the environmental impacts

If the results found in Table 4.5 are divided by the amount of waste produced at the study site during one year (functional unit – see subsection 4.2.1.4), for the case of climate change a value of 26 kg CO_2 eq. per tonne of residual MSW is obtained. Taking into account that this value is attributable in roughly a 50% to the actual collection activity, excluding the material assets contribution (see Figure 4.4), the subsequent value fits into the range of 9–17 kg CO_2 eq. per tonne proposed by Eisted et al. (2009) for a similar case to the one addressed here: residual MSW in an apartment area with a 20 km transport to the treatment facility.

Notwithstanding, the influence of the performance rates of the MSW collection activity was evaluated. A comparison was performed between the environmental impact calculated when real fuel consumption data supplied by the municipality was used against the average fuel consumption in the ecoinvent 3.3 database for the MSW collection (process 'municipal waste collection service by 21 metric ton lorry, RoW'). In practice, this meant that the impacts of the whole annual waste collection were analysed making use of the two performance rates adopted in ecoinvent 3.3 for fuel consumption (4 L/t of collected waste) and distance travelled (10 km/t of collected waste). These numbers were compared with the previous analyses based on the values given in Table 4.4 (4.2 L/t for fuel consumption and 5.8 km/t for distance travelled). The results are shown in Figure 4.8.

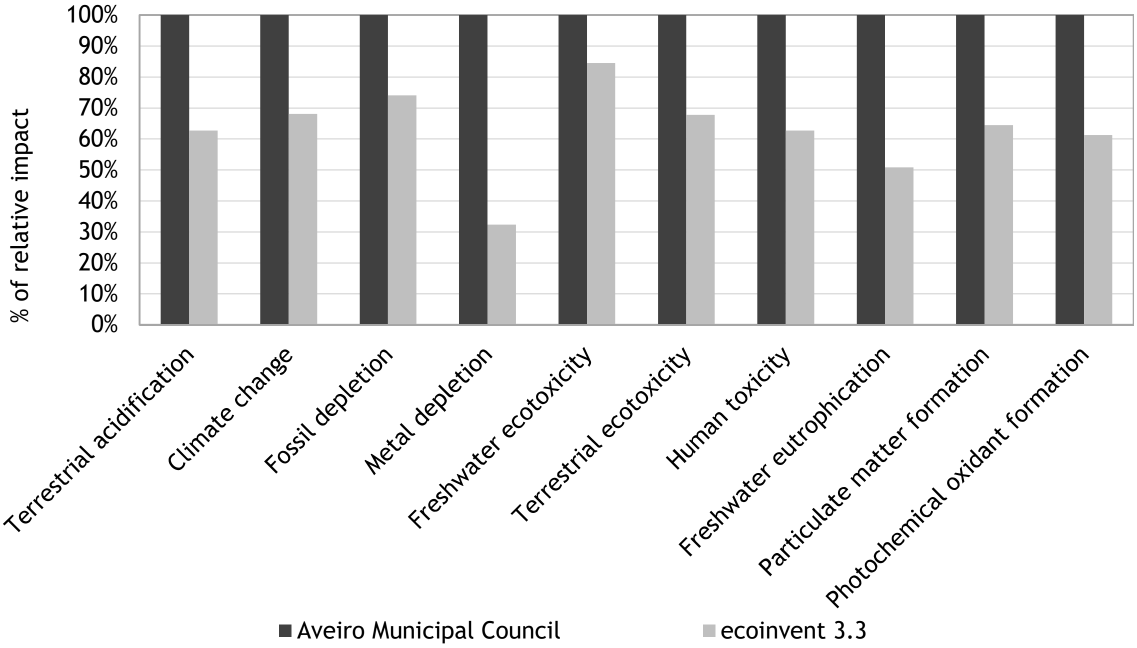

Figure 4.8. Comparison of different waste collection performance rates, using real data and average values from ecoinvent 3.3 (Aveiro municipality data = 100%)

A considerable divergence between the two sources is found in Figure 4.8. The collection in the studied area based on real data has more impacts than the process modelled with ecoinvent 3.3 parameters for every analysed category. The differences go from 18% more impact in freshwater ecotoxicity to more than 300% more impact in metal depletion. The disparity in results may be explained by the local conditions and specific route performances.

Several studies in other different locations were further checked in order to contrast data. Larsen et al. (2009) studied fuel consumption of MSW collection in two municipalities in Denmark, obtaining a value of 1.4±0.4 L/tonne for residual MSW collection for an urban area with both apartment buildings and single-family houses, thus comparable with the neighbourhood analysed in this study. Although this value does not include neither the transport of MSW to the treatment facility nor the journey from and to the garage, following the model proposed by the authors for a transport distance of 10 km (this is the approximate road distance from the city of Aveiro to the treatment facility), the final consumption rate would be around 2.5 L/tonne, which is nevertheless lower than the real value for the case study. Another study by Sonesson (2000) gives a similar value of 2.2 L/tonne for the Swedish city of Uppsala, but 3.0 and 3.4 L/tonne for collection circuits in Stockholm, in all cases including transport to the treatment facility. In this same work the average distance travelled is 3.8 km per tonne of collected MSW in Uppsala and 5–6 km in Stockholm, in any case this is less than the value in Aveiro, thus indicating also better performance. A significant difference between the work of Larsen and the Aveiro case is found in the frequency of collection: whereas in Aveiro this collection is done every day (except Sundays), in the studied Danish municipalities residual MSW is collected only 2–4 times per month. This may be a factor which contributes for a better optimisation of the waste collection in terms of fuel consumption.

In another study, Nguyen and Wilson (2010) recorded the fuel consumption of municipal waste collection lorries in a Canadian city. For residual MSW collection in a high density populated area, they obtained a global fuel consumption rate of 3.2±0.4 L/t. The authors also present a value of 2.0 L/km of fuel consumption relative to travelled distance, but this number does not include the journeys outside the actual collection, in contrast to the 0.7 L/km obtained in this study (from the numbers in Table 4.4). Other authors showed results closer to this case study: in the work of Teixeira, Russo, et al. (2014), focused in the Portuguese city of Porto, an average value of 3.96 L/tonne was obtained for collection and transport of mixed MSW, with an average travelled distance of 6.31 km/tonne, hence suggesting a similar performance. This is maybe justified because the geographical scope of this studies is the same: both are, namely, Portuguese cities.

In principle, it seems evident that the collection frequency should be set accordingly to the MSW generation rate. Hence, in an optimised collection system its efficiency is independent of frequency, as demonstrated by De Jaeger et al. (2011). Notwithstanding, it is also intuitive that a reduction of collection frequency would imply less fuel consumption and thus less environmental impacts. Actually, as indicated in subsection 4.2.2.4, residual MSW is collected every evening in Aveiro and in other Portuguese cities, while in other European cities this collection, including also biowaste, takes place weekly or even in longer periods. A comparison between European capital cities is available: (BiPRO GmbH & Copenhagen Resource Institute, 2015). This practice has been traditionally justified, in order to prevent possible public health problems which may appear if waste is stored for several days, especially during periods with warm temperatures: odours, dust or proliferation of flies and other biological vectors. However, it could be argued that this concern may be to some extent consequence of an overperception of health hazards caused by household waste – perhaps due to cultural beliefs. In fact, studies on the issue have shown no actual threats to public health derived from longer storing periods of garbage, other than nuisances like odour or insects, and even these can be minimised by following proper handling procedures (Enviros Consulting Ltd. & Cranfield University, 2007; Open University, 2009). Indeed, weekly or even fortnightly collection of residual MSW is a well-established practiced also in hot climate locations (EPA, 1997, 1999).

4.4. CONCLUSIONS

A structured framework for the environmental assessment of mixed MSW collection schemes was developed, covering the main elements and activities involved. This structure might be useful as a common reference for further environmental assessments of other waste collection systems. The present study allowed assessing the environmental profile of residual MSW collection in a selected neighbourhood of the Portuguese city of Aveiro through the use of the LCA methodology. The results of LCA highlighted the role of plastic carrier bags to handle household waste as the element with the highest contributions to environmental impacts, even exceeding the contribution of the other relevant influencing factor: fuel consumption by collection vehicles.

The importance of impacts caused by carrier bags was of special relevance in the categories most related to the consequences of landfilling plastic materials. The widespread use of plastic carrier bags made of oil-based polyethylene – i.e. a non-renewable and non-biodegradable raw material – poses a considerable long term harmful effect to the environment when disposed of in landfills. The potential environmental impacts of plastic bags are more pronounced in comparison to the other material assets analysed in this work, due to their use as a throwaway product with short useful lifetime. However, the use of plastic carrier bags is widely established as an easy and clean manner of handling waste – particularly biowaste, prone to problems such as bad smell or biologic vectors proliferation. Bagless collection schemes, where rubbish is directly thrown from dustbins into waste containers already exist in some places, for example in the separate collection of biowaste, but are not frequent for mixed household waste.

Moreover, this study confirms that the use of alternative materials for carrier bags, such as biodegradable polymers is not an environmentally better option than recycling conventional polyethylene bags. In view of the LCA results, the most immediate measure to lessen the environmental impacts would be to encourage the use of recycled plastics for the production of bags – which in turn, implies the reinforcing of the recycling schemes to enhance the recovery and further recycling of plastic films. In the future, biodegradable bags might however have an opportunity to establish themselves as a viable alternative in case biowaste separate collection becomes generalised in Portugal – as encouraged by EU initiatives. Further development of the bioplastics industry may well imply improvements in optimisation of production processes, resulting in better environmental performance, and the production of bags made with biopolymers obtained from residual materials with virtually null environmental burdens reveals itself as a future efficient alternative to conventional oil-based plastics. A last question remains open, concerning the possibility of recycling bioplastics, which would bring further environmental benefits, in the same manner as recycled oil-based plastics. This possibility was not considered in this work, since it is not currently considered to be a realistic alternative, but some authors have already explored its feasibility (Yates and Barlow 2013). Another interesting finding of the study corresponds to the influence of the bag volume on the environmental impact, through the variation of the mass of polymer present in a bag of a given size. This relationship between volume and mass corresponds to a non-linear dependence with form of a cubic function, which points to bags with 30 L of capacity as the optimal ones from an environmental perspective.

Regarding the other major issue found in the assessment – fuel consumption by collection vehicles, results were found to be similar to other cases in Portugal, but different – actually higher – from other examples abroad. It is suggested that one of the factors contributing to this higher impact might be an excessive collection MSW frequency, in the opinion of the authors an issue not sufficiently discussed in Portugal. The comparison of results confirms the dependence of waste collection performance rate on site-specific conditions, therefore highlighting the importance of obtaining real data at a local level in order to get accurate results.

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5. Turning waste management into a carbon neutral activity

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ABSTRACT

A Life Cycle Assessment (LCA) with focus on carbon footprint, followed by Life Cycle Costing (LCC) of municipal solid waste (MSW) management were conducted in a residential area of a medium-sized European city of 80,000 inhabitants. The initial results showed high environmental impacts and lack of economic sustainability, due to the high amounts of waste landfilled, the low extent of separate collection, low performance of mechanical-biological treatment as well as absence from alternatives to landfilling of non-recyclable materials. Taking this result as a baseline scenario, three improvements were tested with the aim of turning the carbon footprint of the local MSW management system into a neutral value: (i) increased separate collection of recyclables, (ii) enhanced biogas production and (iii) refuse-derived fuel (RDF) production. Successively adding the improvements, three alternative improved scenarios were defined, until reaching a negative carbon footprint, meaning that an optimised system would avoid GHG emissions. The proposed changes were sufficient to achieve carbon neutrality, as well as reduce overall environmental impacts, but were not enough for achieving economic sustainability due to the great influence of collection costs, especially for separate collection. It was concluded that by using an adequate combination of several treatment options and increasing the separate collection of recyclable materials it is possible to turn MSW management into a carbon neutral activity as well as improve its economic balance.

[5] Further details provided in the List of publications included (page xxxviii).

5.1. INTRODUCTION

As an expanding methodology for the analysis of MSW management systems – as shown in reviews by Khandelwal et al. (2019) and Laurent et al. (2014a, b), LCA has been useful to demonstrate that increasing recovery of waste resources brings decisive environmental benefits, if compared to other treatment options (Cimpan et al., 2015; Montejo et al., 2013; Song et al., 2013; Rigamonti et al., 2010), in line with the waste management hierarchy adopted by the European Union (EU) through the Waste Frame Directive 2008/98/EC (European Parliament, 2008). In the present article, besides the assessment of the common impact categories covered by the LCA methodology, the authors have decided to specially focus their assessment of MSW management on the issue of carbon footprint, an impact category of current general concern due to its contribution to climate change. The role of carbon footprint in the environmental profile of MSW management has been already analysed for the case of big-sized cities like Madrid (Pérez et al., 2018, 2017a, b), and metropolitan areas, exploring management options such as energy recovery or composting, along with the economic implications derived from it (Halil Yılmaz and Abdulvahitoğlu, 2019; Maalouf and El-Fadel, 2019).

Some authors have proposed through LCA application that MSW management may itself become, under certain conditions, a net carbon sink instead of a carbon generator. Erses Yay (2015) assessed a case study in Turkey, considering recycling, composting and incineration as alternatives to landfilling, while Abu Qdais (2019) proposed alternative scenarios for MSW management in Jordan. However, these works did not consider the contribution of the waste collection to the carbon footprint, thus lacking an integral perspective of the whole MSW management system. The feasibility of a beneficial carbon footprint scenario which such an integral approach has constituted the object of research of this work, in which the authors have analysed a particular case study located in a medium-sized European city in Portugal, where mixed MSW is treated in a Mechanical-Biological Treatment facility (MBT), comprising recovery of recyclable materials, production of biogas and compost from biowaste and controlled landfilling of the residual waste.

These treatment facilities have already been object of specific LCA studies (Bang Jensen et al., 2016; Tagliaferri et al., 2016; Beylot et al., 2015; Montejo et al., 2013; Abeliotis et al., 2012), with different outcomes regarding carbon footprint, depending on each particular case: features like previous source separation of biowaste and existence of heat recovery mostly contributed to negative (avoided) GHG emissions, while the opposite cases (biological treatment of mixed MSW and no heat recovery) lead typically to net GHG emissions in the final balance.

Regarding the particular situation of MSW management in Europe, 47% of MSW was recovered for recycling and valorisation in 2018 (Eurostat, 2020), still below the 50% goal required in 2020 by the 2008 European Waste Frame Directive. Furthermore, the situation is heterogeneous between different member states: whereas the “best performers” have already exceeded the 50% mark (e.g. Germany: 67.3%), while other countries such as Portugal lay still below 30% of recycling rate. Although the separate collection of MSW in Portugal has slightly increased in last years, it reaches only 18.4% of total generation (APA 2019b). In the particular city of this study, the situation follows a similar pattern, but with lower numbers in relation to the national situation: only 6.1% of the total MSW generated was separately collected in 2018 (ERSUC, 2019). The trend in recent years corresponds actually to a stagnation of separate collection percentage, with some decreases between years: 6.4% in 2015, 6.1% in 2016, and 6.0% in 2017 (ERSUC, 2018, 2017, 2016). In this context, the reversal of the current situation in order to meet the proposed goals will require a significant effort of MSW management systems, again combining the different management alternatives existent, as pointed by Lorena and Carvalho (2019).

Hence, the main intention of this study is to demonstrate that, even in these situations of low environmental performance of a given MSW management system, the goal of "carbon neutrality" can constitute a reasonable objective to be achieved, if adequate incentives are strategically introduced to enhance performance. In the case here analysed, different scenarios were explored, where several successive improvements in the MSW management performance were gradually introduced in respect to the current baseline situation, in order to illustrate the ability of the system to become itself a carbon sink.

To this respect, several authors have tested the performance of given MSW management systems comparing scenarios which assumed different levels of MSW source separation, including separate collection of biowaste (Ripa et al., 2017; Giugliano et al., 2011). The results showed that an increased recovery of recyclable materials brings environmental benefits as a consequence. However, as pointed by Rigamonti et al. (2009), a limitation would be the likely loss of quality in recyclable materials due to higher contamination: in their study, this effect would render separate collection ineffective beyond a 50% level. Therefore, the target of waste management sustainability – in this case, carbon neutrality – should rely on a "smart" combination of treatment options, aiming separately at each of the specific MSW fractions.

Moreover, the economic and social implications of the distribution of waste in its different fluxes have not been yet widely studied (Khandelwal et al., 2019) and, as shown by Struk and Pojezdná (2019), it is not always possible to assess them in a straightforward approach. While Tulokhonova and Ulanova (2013) concluded in their study that landfilling was the option with lower costs, Tan et al. (2014) concluded that valorisation processes of MSW can be economically profitable, contrary to landfilling. Similarly, Mirdar Harijani et al. (2017) associated the economic sustainability with the environmental one. Fernández González et al. (2017) pointed to energy recovery from residual MSW as an economically attractive option, a result coincident with Massaruto et al. (2011); however, this study differs in methodology, since the authors also considered assigning monetary values to reflect the adverse health consequences of air pollution and disamenities, thus overlapping environmental and economic assessments. Nevertheless, the integration of environmental assessments such as carbon footprint with economic evaluation of MSW management alternatives from a circular economy perspective should be useful to verify that environmentally efficient MSW management systems can lead in ultimate terms to reduced costs, as pointed by Van Fan et al. (2020).

The work now presented corresponds to the results of an integral assessment of MSW management in a particular neighbourhood of a selected Portuguese city, comprising both the collection and treatment of mixed MSW and of the three separate collection fluxes existent in the city, corresponding to glass, paper and cardboard, and packaging materials made of plastic and metals. This assessment was planned to test the implementation of a *pay-as-you-throw* (PAYT) pricing scheme (Dias-Ferreira et al., 2019). The main purpose of this assessment is to serve as an initial baseline of the environmental impacts derived from MSW management to be taken as a comparison reference for the project implementation. To perform this environmental assessment, the LCA methodology was applied. Besides the evaluation of environmental impacts, an additional economic assessment based on Life Cycle Costing (LCC) methodology – the economic counterpart to LCA –, was also applied to the studied system.

5.2. METHODOLOGY AND DATA COLLECTION

5.2.1. Goal and scope

5.2.1.1. Case study description

The city which is object of this study is located on the Portuguese Atlantic coast. With nearly 80,000 inhabitants, it may well be considered as a representative example of a typical medium-sized European city (European Commission & UN-Habitat 2016), which makes it an object of interest for gaining a better understanding of environmental issues in similar cities. The neighbourhood selected for the assessment currently hosts roughly 1500 inhabitants and consists of 500 households – both apartment buildings and single-family houses –, as well as some commercial establishments.

The MSW collection system currently existent in the area of study consists in kerbside collection based on twenty-six collective street waste 1100 L containers with open access, where household residents and merchants put their rubbish usually within plastic carrier bags. Collection takes place every night (except Sundays), with 20 m^3 of loading capacity waste collection lorries. This system for residual MSW collection is complemented by five groups of drop-off containers, also with open access, intended for separate collection of the three main classes of recyclable materials: paper and cardboard, plastic and metal packaging and glass.

After being collected, MSW from the city is transferred to the MBT facility, roughly 12 km away. There, materials suitable for recycling are recovered from the mixed MSW, namely: metals and the most valuable plastics – mostly polyethylene (PE) and polyethylene terephthalate (PET). Thereafter, biowaste is degraded through anaerobic digestion with the goal of producing primarily biogas – used as biofuel in gas engines for electricity production –, and secondarily, compost, suitable for agricultural application. In parallel, MSW from separate collection are sorted in another section of the same facility and packed for being sent to recycling companies. Materials not suitable for recycling are processed as refuse-derived fuel (RDF) or (mostly) landfilled.

A relevant consequence of the low extent of source-separation (as explained in the Introduction) is the excessive presence of heterogeneous materials within residual MSW, difficult to separate in the MBT facility. In 2018, 62% of the total waste mass which entered the MBT was disposed of in a sanitary landfill (APA, 2019a). Most of this landfilled waste corresponded to rejected materials, unsuitable neither for recycling nor for organic valorisation in form of biogas or compost, but also biowaste which is not recovered. It is expected that this study will contribute to assess how this lack of quality affects the environmental performance of the overall MSW management.

5.2.1.2. Main outcomes expected and definition of the studied system

The goal of the study is to assess the environmental and economic balance credited to the management of MSW collected during one particular year within the area designated for the previously mentioned pilot experience, considering both mixed MSW and separately collected MSW streams. Consequently, one year was the time reference set for the analysis. In particular, the year 2017 was established as temporal reference to which all data – prices included – were referred.

The whole system encompassing all the analysed activities – known as unit processes –, along with the boundary of the present analysis, is shown in Figure 5.1 in form of a flow diagram, where the sequential relationship between the elements represents the route of the reference flow. Solid arrows represent MSW fluxes, dashed arrows represent primary resources replaced by resources recovered from waste and a dotted arrow represents the main residual outputs of the system emitted to the environment, namely carbon dioxide, methane and water resulting from biologic degradation.

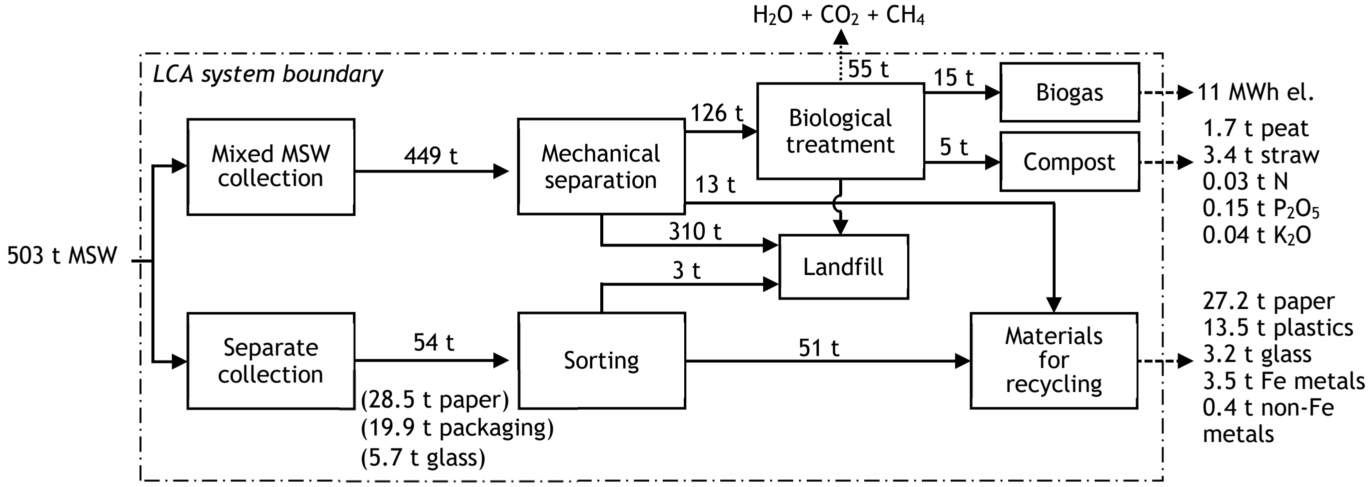

Figure 5.1. Representation of the analysed system (mass fluxes) in reference year 2017

Every unit process in this system requires consumption of energy and raw materials and generates polluting emissions. Each of the unit processes was analysed as an independent element making part of the studied system. Modelling of the MSW management processes involved – i.e. collection, treatments and disposal – is described in Section 5.3.

5.2.1.3. Modelling framework and cut-off criteria

An attributional framework was chosen for the LCA modelling. Considering that the work included comparison of alternatives – different scenarios for MSW management –, this would correspond to a type A goal decision-context situation (JRC – IES, 2010). The analysed system was considered to provide the single function of managing municipal solid waste; notwithstanding, a multifunctionality solving procedure through system expansion was needed to account for the environmental credit derived from the recovery of valuable materials and energy extracted from the reference flow (Figure 5.1), further explained with more detail in subsection 5.3.2.1. Regarding cut-off criteria for the analysis of the unit processes, all materials sent to recycling as end-of-life (EoL) treatment option were counted as avoided waste, i.e. no environmental burden was allocated to the system originating the recycled material.

5.2.1.4. Functional unit and reference flow

The main function fulfilled by a MSW management system like the one assessed here consists on providing citizens with an adequate waste management service – comprising collection and treatment of mixed and source-segregated MSW fluxes – in order to: firstly, ensure public health and secondly, preserve the environment. In accordance with this function, the functional unit (FU) selected for the LCA was the total annual mass of MSW collected within the study area. In this definition of the FU are comprehended the residual or mixed MSW amount and three separate collection fluxes: glass, paper and cardboard and plastic and metal packaging. Other separate collection fluxes – green (garden) waste, used cooking oil, wood and furniture, waste electric and electronic equipment (WEEE), etc. – were not addressed in this study. The choosing of one year as time frame for the functional unit was justified because it corresponds to a complete cycle of seasonal variation of the MSW generation pattern.

The corresponding amount and physical composition of the mixed MSW were determined basing on field measurements obtained from a specific waste characterisation campaign (Dias-Ferreira et al., 2019). For the recyclable MSW streams the annual numbers were obtained from the managing company referring to the filling degree of drop-off containers in the neighbourhood and general composition data for each of the three main recyclable fluxes. The resulting figures are seen in Figure 5.1.

5.2.1.5. Data collection and methods of assessment

Data regarding MSW generation at local level and its composition and properties were obtained from the characterisation campaign mentioned in subsection 5.2.1.4. The municipality supplied their own data concerning MSW collection activities at city level, thereafter adapted for the area of study. General characteristics, manufacturing activities and raw materials for analysed material assets (waste bags, bins and containers, collection vehicles) were obtained from the respective producers whenever possible. In parallel, the company responsible for MSW treatment provided information about performance and main features of the MBT facility. In case direct sources of information were not available, complementary data were compiled from life cycle databases – namely ecoinvent 3.5® (ecoinvent Association, 2018). Own assumptions are explained in the following sections.

The environmental assessment was performed using the commercial software SimaPro version 9.0.0. (PRé Sustainability, 2019). The impact assessment method applied for carbon footprint was the 2013 IPCC characterisation methodology for a 100-year period (version 1.02), while for the rest of selected categories the ReCiPe Midpoint (hierarchist perspective) Version 1.12 (Huijbregts, et al., 2017) was applied. The ReCiPe method was chosen due to the completeness of the impact categories covered. In particular, several impact categories were selected as being the most relevant regarding environmental impacts of waste management: along with global warming, also ozone formation (impact contribution to human health), fine particulate matter formation, terrestrial acidification, freshwater eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, human non-carcinogenic toxicity, mineral resource scarcity and fossil resource scarcity. These categories were chosen according to practices observed in similar recent LCA studies on waste management (Andreasi Bassi et al. 2017; Pires et al. 2017; Ripa et al. 2017).

5.3. LIFE CYCLE INVENTORY

The data gathered allowed the elaboration of a Life Cycle Inventory (LCI), intended to comprehend the environmental profile of the activities related to MSW management, namely: collection, transport to treatment facilities, the subsequent treatment and valorisation processes and final disposal of the materials not suitable for recovery. The inventory for each activity is described next.

5.3.1. MSW collection

The modelling of the mixed MSW collection process in the studied area was already addressed with detail by the authors in a previous study (Fernández-Braña et al., 2019). In this present work, the aspects corresponding to the modelling of the separate collection have been subsequently added. For modelling purposes, four main elements have been considered as necessary constituents of MSW collection process: on the one hand, three static material assets intended to store household waste at different stages of the collection process: carrier bags, household waste bins and street containers; on the other hand, the specific features for collection vehicles in the different collection routes (mixed and separate collection) have been modelled separately.

For the three assets mentioned before (bags, household bins and containers), the amounts of main raw materials required for their production are summarised in Table 5.1. In the modelling of production activities requiring a power input, such as extrusion and metal working, the Portuguese electricity supply mix available in ecoinvent 3.5 was applied. Furthermore, for modelling of EoL management of these assets, bins and containers were assumed to be recycled after their use period, therefore no environmental burden was assigned to this respect. In analogous manner, no environmental burden was neither assigned to the EoL of carrier bags, since they are already assumed to have been counted as part of the FU (i.e. part of the MSW mass collected).

Table 5.1. Raw materials summary for bags, dustbins and street containers (HDPE: high density polyethylene)

Unit process	Weight per unit	Units per FU	Lifespan	Raw Materials	Amount per FU
Carrier bags	7.2 g	199,597	----	HDPE	1470.4 kg
Household bins	0.75 kg	3104	7 years	Polypropylene	334.6 kg
Street containers (for mixed MSW)	43 kg	26	14 years	HDPE	70.3 kg
				Steel	8.5 kg
				Rubber	2.8 kg
Street containers (for separate collection)	156 kg	15	14 years	HDPE	163.0 kg
				Steel	5.6 kg
				Rubber	0.3 kg

The assessment of MSW collection vehicles was based on the data provided by the municipality regarding equipment and fuel consumption of mixed MSW collection. For fuel consumption of separate collection, the approach proposed by Moreira Monteiro (2013) was considered as a reasonable estimation, since it is based on a specific study for the same city. The fuel consumption rates and mean distance travelled for all cases are presented in Table 5.2.

Table 5.2. Fuel consumption and travelled distance of MSW collection vehicles

Type of collection	Fuel consumption		Distance travelled	
	Per MSW collected (L/t)	Per FU (L)	Per MSW collected (km/t)	Per FU (km)
Mixed MSW	4.23	1901	5.84	2622
Glass	19.79	114	32.32	186
Paper	23.77	679	44.17	1261
Plastic / metals	50.55	1006	101.47	2019

Data for the production and EoL of the collection vehicle was adapted from ecoinvent 3.5 database (Doka, 2007), along with the inventory of pollutant emissions. This inventory was combined with the Tier 2 methodology of the EMEP/EEA guidebook (EMEP/EEA, 2016) for indirect emissions of particulate matter (PM) related to tyre wear, brake wear and road surface wear. Lifespan of the mixed collection vehicles – which are employed in typical heavy-duty working mode – was set at 10 years, corresponding to a total travelled distance of 286,722 km; whereas for the other collections it was kept as in ecoinvent database. The load factor was set as 50% as a mean value and the average speed was set at 40 km/h.

5.3.2. MBT facility

The treatment processes at the MBT facility were modelled according to the process diagram scheme shown in Figure 5.1, basing on the annual mass balances (ERSUC, 2018). These mass balances were combined with the physical composition of mixed MSW generated in the studied neighbourhood (subsection 5.2.1.4). In Table 5.3 this composition of mixed MSW is presented, along with the amounts from separate collection and the resulting mass flows entering the MBT and sorting facility, referred to the scale of the FU.

Table 5.3. Composition of reference waste flow

Type of MSW	Total mass per FU (t)	MSW fractions	Physical composition (mass)	Mass per FU (t)
Mixed MSW	449.1	Biowaste	59.8 %	268.6
		Textiles	10.5 %	47.2
		Paper and cardboard	9.0 %	40.4
		Plastics	5.6 %	25.1
		Glass	5.1 %	22.9
		Hazardous and WEEE	1.3 %	5.7
		Composites	1.1 %	5.0
		Wood	1.0 %	4.4
		Metals	0.8 %	3.5
		Fines	2.6 %	11.8
		Others	3.2 %	14.4
Glass packaging	5.7	Glass	100.0 %	5.7
Paper and cardboard packaging	28.5	Paper and cardboard	99.0 %	28.3
		Others	1.0 %	0.2
Plastic and metal packaging	19.9	Plastics	58.8 %	11.7
		Composites	11.5 %	2.3
		Fe-metals	13.6 %	2.7
		Non Fe-metals	1.2 %	0.2
		Others	14.9 %	3.0

The fate along the treatment processes present on the MBT of each of the material fractions shown in Table 5.3 was tracked until finally reaching one of the three main outcomes reflected in Figure 5.1: either the waste material is recovered for recycling (if suitable), valorised as biogas or compost (in the case of biowaste) or disposed in the landfill (in the case of materials rejected from the other valorisation processes). The particular modelling for each of these three main outcomes is described with detail in the next sub-sections.

Consumption of utilities and consumable goods was obtained through personal communication or estimated with data from other MSW treatment facilities. In brief numbers, the MBT was estimated to consume per FU: 78.4 L of water (later treated as wastewater), 68.5 L of diesel fuel, 46.6 L of lubricating oil and 4.3 kg of printed paper. Impacts related to the construction of buildings and equipment were taken from ecoinvent 3.5® database and adapted to the actual surface occupied by the analysed facility (95.7 m^2 per FU).

5.3.2.1. Sorting and recovery of materials for recycling

In 2017, 94% of materials received from separate collection at the sorting facility were sorted and prepared for recycling (ERSUC, 2018). It was assumed that 100% of glass and 99% of paper and cardboard were sent to further recycling, while only 85% of the packaging plastics and metals could be recovered. Besides the materials from separate collection, other valuable materials – namely metals and some common plastics (polyethylene, polyethylene terephtalate) – are recovered from mixed MSW in the mechanical separation stage. All rejected materials were landfilled without further treatment.

The energy consumed by the sorting process was not accounted for specifically as part of the LCI, since it was included as part of the general energy balance of the MBT. On the other hand, the system expansion procedure allowed to account for "environmental credit" attributed to the materials recovered for recycling in exchange for replacing primary raw materials, while the recycling processes themselves are accounted for as generators of environmental impacts. To perform the accounting of credits, it was decided to follow the approach suggested by Bala Gala et al. (2015). The approach considers not only the loss of quality of some recycled materials when compared to the primary ones – through a quality loss factor –, but also the fact that for materials where recycling is already well established – like glass or metals – it is more realistic to model recycling as a replacement of a mixture of primary and secondary (recycled) materials (substitution factor), rather than a 100% amount of primary material (a 1:1 basis substitution). These two correction factors are combined to obtain global substitution factors, presented in Table 5.4 for all recovered materials. Both production processes for primary and recycled products were taken from ecoinvent 3.5 database.

Table 5.4. Environmental credits assigned to recovered materials (PE: polyethylene, PET: polyethylene terephthalate, PP: polypropylene, PVC: polyvinyl chloride, PS: polystyrene)

Material	Recovered amount (per FU)	Quality loss factor	% recycled market share	Substitution factor	Substituted product	Substituted amount (per FU)
Glass	5.7 t	1	0.45	0.84	Primary glass	4.8 t
Fe-metals	7.0 t	1	0.50	0.59	Pig iron	4.2 t
Non Fe-metals	0.68 t	1	0.37	0.65	Primary aluminium	0.45 t
Paper / cardboard	30.0 t	0.80	0.47	0.47	Primary paper	13.4 t
Plastics	19.8 t	0.75	0.12	0.67	Mix of primary plastics: 39% PE 33% PET 19% PP 5% PVC 3.5% PS	13.2 t

Metals other than iron were modelled as aluminium – being the most common non-Fe metal. Plastics were modelled as a generic mix of common plastic materials – as seen on Table 5.4. Lastly, composite materials were modelled basing on a typical beverage packaging: a mixture of 75% cardboard, 20% PE and 5% aluminium. For all recovered materials, a generic transport distance of 200 km by road to the recycling facilities was assumed as a typical transport distance at Portuguese scale.

5.3.2.2. Biological treatments

As specified in Figure 5.1, it is estimated that 126 tonnes of the FU are admitted to the biological treatment, after mechanical separation of elements other than biowaste. This treatment consists of two main stages: first, an anaerobic digestion process in order to obtain biogas to be used as fuel and thereafter, a composting stage in aerated closed tunnels, followed by compost maturation in windrows, for biological stabilisation of the digested matter. It was estimated that around 15 tonnes of this substrate – consisting mostly in biodegradable organic matter – was transformed into biogas through the anaerobic digestion stage, while the rest was sent to the composting treatment. The company sold 5 tonnes of compost for application in agriculture in 2017, thus implying that the rest of compost produced (around 51 t) was disposed of in the landfill, where it is used as covering material.

The produced biogas allowed generating almost 47 MWh of energy per FU, from which 18 MWh were effectively transformed in electricity – considering a 40% generation efficiency and transmission losses. After subtracting the internal consumption of electricity required by the facility, the surplus of electricity supplied to the public distribution grid corresponded to 11 MWh per FU. This means that an environmental "credit" is generated, since it was assumed that power generation from biogas allowed to replace in a 1:1 basis an equivalent amount of the average Portuguese high voltage electrical power generation (as represented in ecoinvent 3.5). Methane emissions were considered as for the worst cases in Dinkel et al. (2012): per each tonne of digested biowaste, 0.9 kg CH_4 are lost due to biogas leaks and 2.5 kg CH_4 are released due to methane diffuse emissions during composting.

Regarding the compost use on land, soil emissions were kept as in ecoinvent 3.5® database, but complemented in some cases with own composition measurements of the digested matter – published by Oliveira et al. (2017). For the environmental credits gained by the use of compost, the approach of Hermann et al. (2011) was followed: industrial compost substitutes (as a soil conditioner) in a 1:1 basis a mixture of 25% peat and 75% straw. Substitution capacity regarding fertiliser potential was set at 20% for N, 100% for P and 100% for K, as in Boldrin et al. (2010).

5.3.2.3. Disposal in landfill

After the recovery and valorisation treatments, there still remains a considerable amount of materials rejected from these treatments: 314 tonnes per FU, whose final destination is a safe disposal in the sanitary landfill. The estimated composition of this refuse material is presented in Table 5.5.

Table 5.5 Composition of landfilled refuse

Fractions	%	Mass per FU (t)
Biowaste	47.7	149.5
Textiles	15.0	47.2
Paper and cardboard	11.4	35.7
Plastics	5.5	17.2
Glass	7.3	22.9
Hazardous and WEEE	1.8	5.6
Composites	1.6	5.0
Wood	1.4	4.3
Metals	-	0
Fines	3.8	11.8
Others	4.6	14.4
TOTAL	**100.0**	**313.7**

Given that the landfilled mass of refuse MSW corresponds to 62% of the total weight, it is expected that the particular behaviour in the landfill of this refuse – in terms of pollutant emissions – will influence to a great extent the overall environmental results of the LCA.

Consequently, it was necessary to model this behaviour as close as possible to the particular case here analysed. The selected approach was to adapt the procedure in the ecoinvent database for the modelling of municipal waste disposal in a sanitary landfill (Doka, 2007) to the specific refuse composition shown in Table 5.5. Furthermore, the 51 tonnes of unsold compost which are also disposed of in the sanitary landfill, can be modelled as a hardly degradable organic matter (for instance wood), since this biowaste has, to a great extent, been already stabilised from a biological point of view.

5.4. Costs and revenues inventory

In parallel to the development of the life-cycle inventory, the costs and revenues related to the main activities comprised by MSW management were grouped together in an analogous inventory, presented in Table 5.6, where the monetary values are already extrapolated to the scale of the FU, i.e. the analysed neighbourhood.

The costs shown in Table 5.6 comprise: the price for collection of mixed MSW along with a gate fee for the MBT facility. Additionally, the municipality also dedicates human resources to perform all administrative tasks related with MSW management. The gate fee for mixed MSW is intended to cover the treatment costs, and is regulated by this governmental body. On the other hand, the costs of recyclables collection and sorting are supported by the same entity responsible for mixed MSW treatment. The corresponding values are estimates communicated by the responsible entities to the same governmental regulator. Lastly, the value of the waste management tax is fixed by the Portuguese environmental authority as a financial instrument intended to penalise MSW disposal, particularly in the cases of landfilling and incineration. Given that MSW incineration is not directly present for the case study, to the extent of this study the application of the waste management tax refers only to landfilling and its value was last actualised in 2014 (AR, 2014).

Regarding revenues, these are originated by the selling of resources (materials and energy) recovered from MSW. In the case of Portugal, the national government establishes a minimum selling price for recyclable materials – intended to cover the costs derived from its separate collection and sorting. These minimum prices were last officially fixed in 2016 (ME&MA, 2016). It is noteworthy to recall that the prices are different depending on the quality of the material: a distinction is made between recyclable materials separately collected (cleaner) and those materials recovered in the MBT from mixed MSW (dirtier, i.e. with less quality). Finally, the revenues from the selling of compost and electricity were estimated from the financial report presented by the company (ERSUC, 2018).

Table 5.6. Costs and revenues inventory for MSW management in the study area in reference year 2017

Costs			Revenues		
Expenses	**Unitary cost**	**Total per FU**	**Incomes (sales)**	**Unitary price**	**Total per FU**
Municipal administrative staff	2.89 €/tonne	1456.86€	Beverage cardboard	564.00 €/tonne	1289.14€
Mixed MSW collection	46.45 €/tonne	20862.04€	Glass	36.00 €/tonne	206.79€
Gate fee for mixed MSW	27.29 €/tonne	12255.78€	Paper/cardboard	173.00 €/tonne	4889.62€
Separate MSW collection and sorting	225.00 €/tonne	12192.09€	Fe metals (from mixed MSW)	131.00 €/tonne	564.10€
Waste management tax	5.39 €/tonne	2420.75€	Fe metals (from separate MSW)	649.00 €/tonne	1752.50€
			Non-Fe metals (from mixed MSW)	180.00 €/tonne	59.15€
			Non-Fe metals (from separate MSW)	761.00 €/tonne	183.82€
			Plastics (from mixed MSW)	136.00 €/tonne	1079.76€
			Plastics (from separate MSW)	545.00 €/tonne	6376.66€
			Compost	9.82€/tonne	50.61€
			Electricity production	114.75 €/MWh	1291.52€
	TOTAL COSTS	**49187.50€**		**TOTAL REVENUES**	**17743.67€**

It should be stressed that the costs and revenues reunited in Table 5.6 are not borne by a single stakeholder. That is, Table 5.6 is not a financial balance, neither of the municipality nor for the responsible entities. It should instead be comprehended as an overall picture of all monetary operations between the stakeholders directly involved in MSW management.

5.5. RESULTS AND DISCUSSION

5.5.1. Environmental assessment

As part of the initial environmental life-cycle impacts assessment (LCIA), the ReCiPe methodology was applied to the LCI built in Section 5.3. The assessment led to the following results, presented in Table 5.7 in form of characterisation values.

Table 5.7. Characterisation of environmental impacts

Impact category	Results per FU	Unit
Global warming	165,064	kg CO2 eq.
Ozone formation (human health)	92.96	kg NOx eq.
Fine particulate matter formation	25.98	kg PM2.5 eq.
Terrestrial acidification	76.4	kg SO2 eq.
Freshwater eutrophication	12.76	kg P eq.
Terrestrial ecotoxicity	170,372	kg p-DCB eq.
Freshwater ecotoxicity	170,876	kg p-DCB eq.
Human non-carcinogenic toxicity	5370,443	kg p-DCB eq.
Mineral resources scarcity	-133	kg Cu eq.
Fossil resources scarcity	-5874	kg oil eq.

If a subsequent normalisation step were to be applied to the results of Table 5.7, the toxicity categories would clearly dominate the whole impacts assessment, especially the freshwater ecotoxicity followed by the human toxicity. The prevalent contribution for these two categories corresponds to the long-term pollutant emissions originated in landfills. This effect can be verified in Figure 5.2, which offers a picture of the relative contribution to the environmental impact categories resulting from each element of the analysed system.

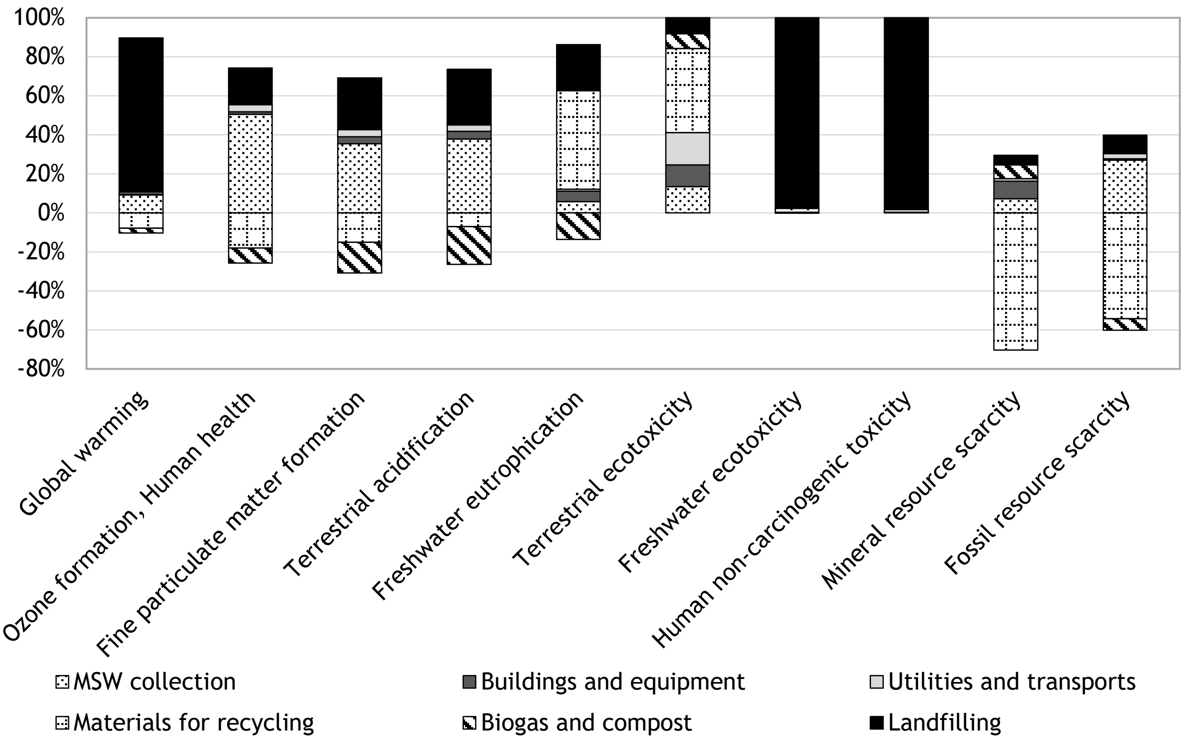

Figure 5.2. Relative contributions of MSW management to environmental impacts

Besides the two previously mentioned toxicity categories, landfilling is also the main contributor to global warming. This is mainly due to methane emissions produced by degrading landfilled biowaste. Even though sanitary landfills have their own collection systems for capture and utilisation of the biogas generated within the landfill, some degree of uncontrolled methane release to the atmosphere always exists (Themelis and Ulloa, 2007). Other impact categories, such as those more related with air quality (for instance, fine particulate and ozone formation and terrestrial acidification) are more dominated by the pollutant emissions generated by collection vehicles. Finally, the recycling of valuable materials contributes greatly to a beneficial outcome (impacts with negative value) in the resources categories (minerals and fossil resources), while the energy and resources consumption implied by recycling processes are penalised in categories such as freshwater eutrophication and terrestrial ecotoxicity; nonetheless, the impacts generated by recycling are still less pronounced than those generated by other waste treatment options, namely landfilling.

5.5.2. Alternative carbon footprint scenarios

The carbon footprint for the studied system is shown in Figure 5.3. The global greenhouse gases (GHG) emissions amounted to +165,064 kg CO_2 eq., assigned to the functional unit (see Table 5.7), or alternatively: 328 kg CO_2 eq. per tonne of collected MSW. This result is higher than the values reported by Pérez et al. (2018, 2017 a, b): 3.59 kg CO_2 eq. / t for MSW street containers, 25.1 kg CO_2 eq. / t for collection and 224 kg CO_2 eq. / t for treatment, although these numbers are referred to the much larger city of Madrid, so that the scale effect may have influence. Nevertheless, the considerable high amount of landfilled waste in the study area (62% of total waste generated) results in a high environmental impact in form of methane emissions, confirming the low environmental performance of the current MSW management system in terms of carbon footprint.

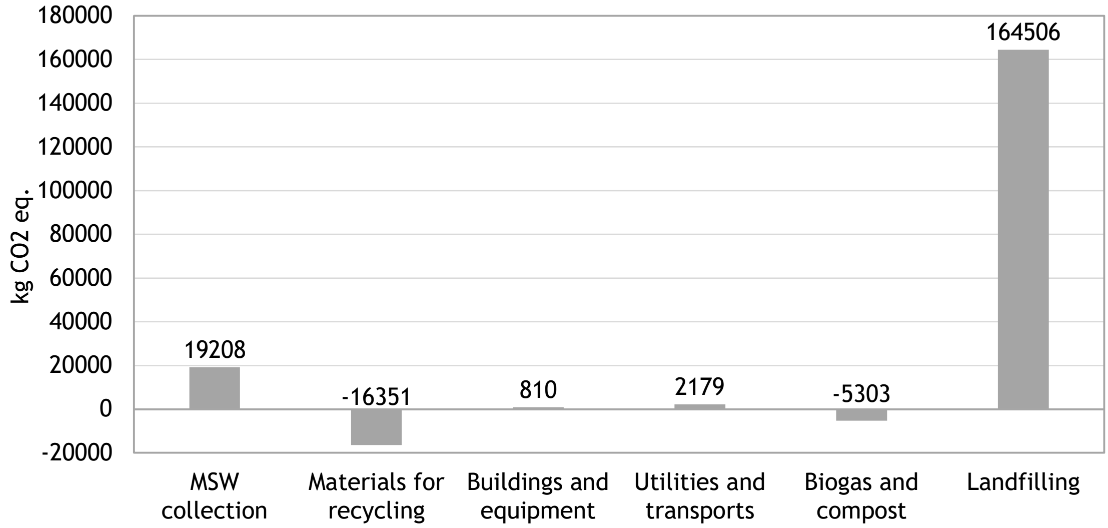

Figure 5.3. Relative contribution to global warming in base scenario S0 (global carbon footprint: 165,064 kg CO_2)

Among the activities contributing to the carbon footprint in the MSW management system shown in Figure 5.3, four have positive footprints – collection, material assets and landfilling – while the remaining two – biogas/composting and recycling – have negative footprints. It is noteworthy to observe that the contribution of the biologic processes – anaerobic digestion (biogas) and composting – offsets the emissions through the benefits derived from replacing conventional electricity production with generation based on biogas combustion. However, the benefits are not as large as could be expected given the already large influence of renewable energies in the Portuguese electric generation mix, the biowaste being partially landfilled (instead of converted to biogas), and the expected lower performance of the biogas production in the MBT due to the heterogeneity (i.e. low quality) of the waste substrate introduced in the anaerobic digesters. Regarding the recycling of recovered materials, the low separate collection rate (12%) does not allow to achieve higher benefits in form of avoided GHG emissions.

Taking the result of Figure 5.3 as a representation of the baseline scenario (S0) in terms of carbon footprint, alternative scenarios are proposed aiming at carbon neutrality (no net GHG emitted, or even GHG emissions avoided). A prerequisite was that any improvements would not consist on alterations to the actual infrastructures at the MBT, but else, scenarios should be built based on an increased performance of the MSW management system, namely:

- Increase in separate collection of recyclable materials by adoption of pay-as-throw tariffs
- Increase in biogas output by introducing in the anaerobic digesters a less-contaminated organic material
- Introduction of adequate management options for non-recyclable materials: RDF

Each of these three improvements was gradually introduced on the original baseline scenario in order to test their influence on the system performance – adjusting in every case the composition of the landfilled refuse, thus generating respectively three improved scenarios (S1, S2 and S3), discussed in the following sub-sections.

5.5.2.1. Improved scenario S1: increased separate collection

As first improvement introduced, it was considered how the carbon footprint of the MSW management system will be mitigated if the separate collection of MSW on the studied area increased until the highest possible separation level of the three recyclable MSW streams, without altering the overall MSW composition and without introducing separate collection of other MSW streams (namely biowaste). In practice, this means increasing the source separation percentage from the previous 12% until 34%: 73.4 tonnes of paper and cardboard instead of the current 28.5, 66.8 tonnes of plastic and metal packaging instead of 19.9 and 29.9 tonnes of glass instead of 5.7. Consequently, the resources required for the separate collection of the increased amounts have been also accordingly modified in the LCI model.

Although this percentage of separation should be interpreted as a theoretical maximum level, it may be regarded as typical of the most advanced cases of successful application of incentives for source separation of recyclable materials, many of them related to the adoption of PAYT pricing schemes; see for instance: Zero Waste Europe (2019), Morlok et al. (2017), BiPRO GmbH & Copenhagen Resource Institute (2015). However, in this context of maximum source separation, it is expected that, as an undesired side effect, contamination of separate collection fluxes would be increased until 30% for plastic and metal packaging, 10% for paper and cardboard and 5% for glass (Algar, 2018; Martinho et al., 2017). The corresponding carbon footprint of the scenario generated (S1) is presented in Figure 5.4.

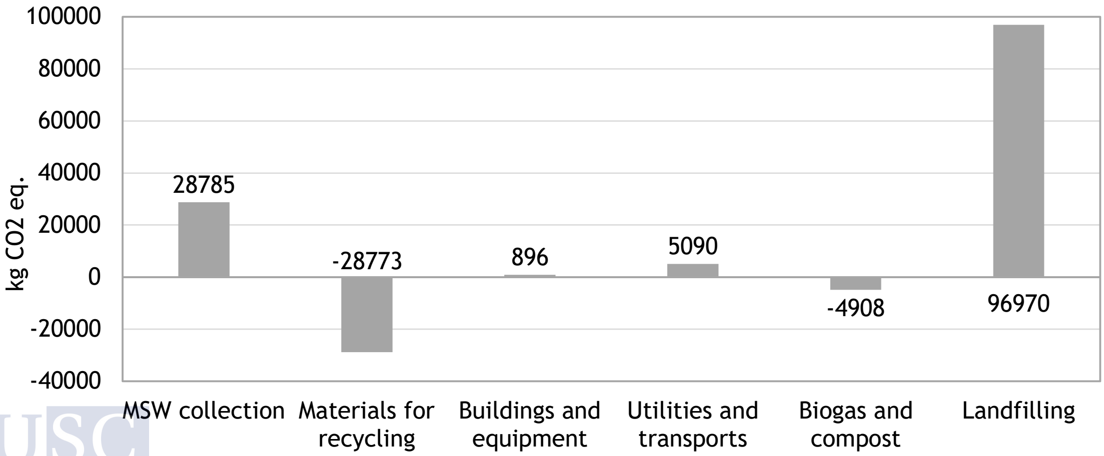

Figure 5.4. Global warming contribution of scenario S1 (global carbon footprint: 98,059 kg CO_2 eq. emitted per FU)

Global GHG emissions in this scenario S1 are still considerable: +98,059 kg CO_2 eq., but have been already reduced a 40% with respect to the baseline S0, thus showing the decisive importance of enhancing the recovery of recyclable materials.

5.5.2.2. Improved scenario S2: increased separate collection (S1) + increased biogas output

If non-biodegradable materials are removed from mixed MSW through separate collection, then the heterogeneity of this waste flux will be reduced and the relative content in biowaste would be higher. This implies that mechanical separation of MSW fractions in the MBT facility will improve its performance, so that the substrate introduced in the anaerobic digesters will be of better quality – even more if separate collection of biowaste were introduced, which currently is not the case. Hence, biogas and compost production will arguably be higher. Based on this consideration, a second alternative scenario (S2) was generated, where besides the increased separate collection previously proposed in S1, biogas and compost production will be increased up to a 95% utilisation of all biowaste found in mixed MSW. That is: 242 t of biowaste will be sent to biologic treatment, instead of current 126 tonnes (Figure 5.1), generating 30 tonnes of biogas – where the methane content was set at the typical 65% in volume – and subsequently, 114 tonnes of compost, of which 73 tonnes will be assigned to agricultural application, whereas the rest will be disposed of in the sanitary landfill (41 tonnes; the same ratio with respect to landfilled MSW as in S0). The resulting new carbon footprint scenario S2 is shown in Figure 5.5.

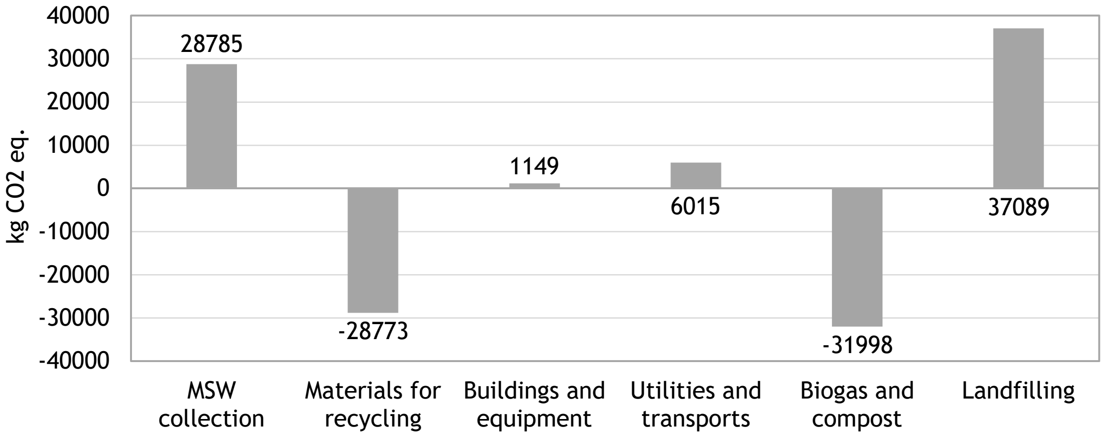

Figure 5.5. Global warming contribution of scenario S2 (global carbon footprint: 12,267 kg CO_2 eq. emitted per FU)

The improvement in biogas and compost has significantly contributed in S2 to reduce GHG emissions almost 8 times regarding the previous scenario S1, with a global result of +12,267 kg CO_2 eq. emitted per FU. This number still represents a net impact in carbon footprint, although near of neutrality if compared with the baseline situation S0. The combination of recycling and biogas production, along with the reduction of the landfilled MSW amount, has contributed to reverse the carbon footprint result to a large extent. This scenario might be reached with a better separation of the most valuable fractions of MSW. To further improve the carbon footprint balance, measures for the rest of MSW components should be taken. This is discussed in the next scenario S3.

5.5.2.3. Improved scenario S3: increased separate collection (S1) + increased biogas output (S2) + introduction of RDF

Once the recovery of recyclable materials and valorisation of biowaste have been addressed, the only option left for achieving a neutral or positive carbon footprint in the analysed case of MSW management is to focus on the valorisation of non-recyclable materials. In fact, the current MSW management system in the city envisions the production of RDF as a secondary valorisation treatment. However, this alternative is currently not usual in Portugal due to low price RDF concurrence from foreign countries, namely UK: provided that landfill taxes are higher there, this turns waste exportation into a financially more attractive option. Nevertheless, RDF has potential to be used as fuel, for instance in cement kilns – an industry with significant presence in Portugal. In a new ideal scenario for MSW management, rejected materials from separate collection, along with wood and textile materials recovered from mixed MSW might be suitable for RDF production – instead of being landfilled. In this new scenario 71 tonnes of RDF would be produced, with calorific power set at 12 MJ/kg (as communicated by the MSW management company), thus equivalent to 21951 m^3 of natural gas which will be replaced by the alternative fuel. Added to the other improvements previously introduced in S1 and S2, the final alternative generated scenario (S3) is presented in Figure 5.6.

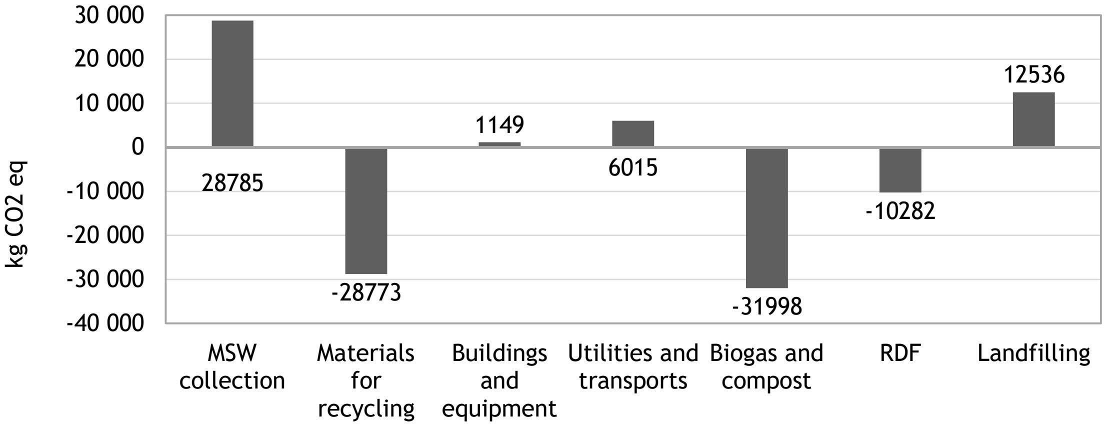

Figure 5.6. Global warming contribution of scenario S3 (global carbon footprint: -22,568 kg CO_2 eq. emitted per FU)

In this scenario S3 GHG emissions showed a negative value: -22,568 kg CO_2 eq., or -45 kg CO_2 eq. / t; this value of carbon footprint implies a beneficial outcome, since GHG emissions are avoided. This can be verified comparing the rest of selected impact categories in the whole impact assessment, after applying the ReCiPe methodology to the new improved scenario. The results of the compared assessment between the baseline S0 and the improved scenario S3 are presented in Figure 5.7.

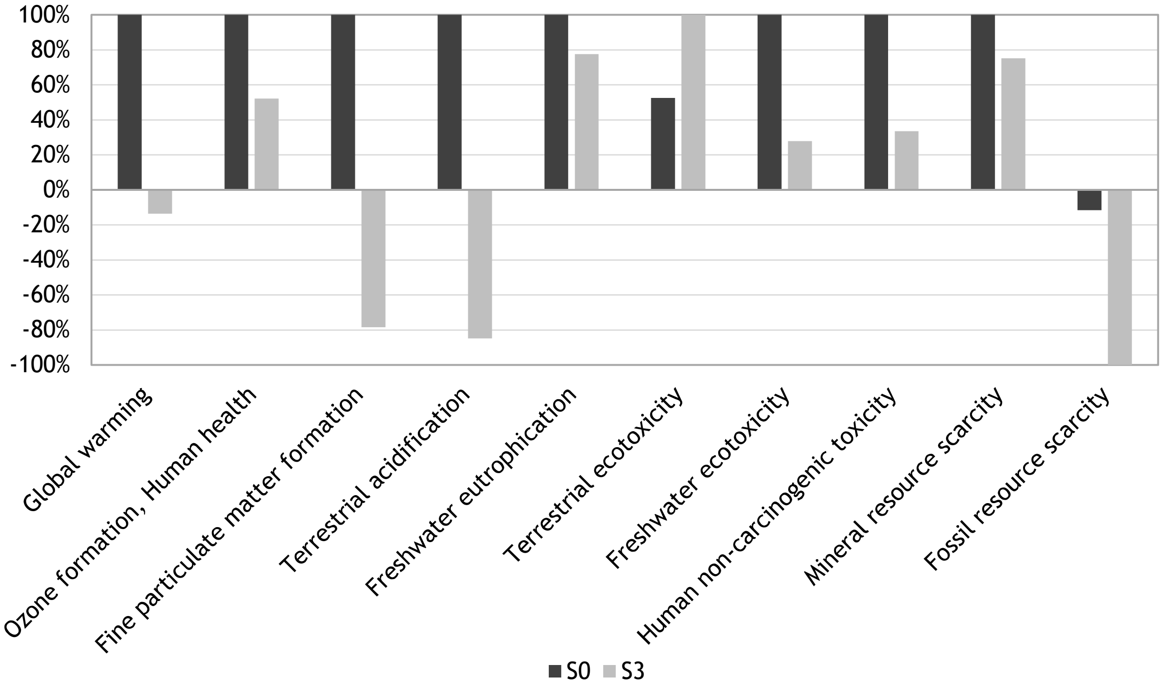

Figure 5.7. Compared LCIA of S0 and S3 MSW management scenarios

Almost all categories in Figure 5.7 progressed in a more environmentally beneficial direction, motivated by the changes introduced: less landfilling, more recycling and enhancing of valorisation treatments. There is an exception, however, in terrestrial ecotoxicity, which is higher in the improved scenario due to the increased recycling activities. Nevertheless, this undesired consequence is compensated by the outcome in the other categories, especially in the others related to toxicity issues. The same happens with the emissions generated by MSW collection, now higher due to the increased separate collection effort (see Figure 5.6), but anyway offset by the avoided emissions in the other processes of the system. The changes introduced in the mass balances of the system are summarised in Table 5.8.

Table 5.8. Changes in final destination of MSW between scenarios S0 and S3

MSW final destinations	% mass	
	S0	S3
Recycling	13	34
Biogas	3	6
Compost	1	15
RDF	----	14
Direct emissions	11	14
Landfill	72	17
TOTAL	**100**	**100**

5.5.3. Economic assessment

The aggregation of the monetary inputs and outputs listed in Table 5.6 results in a negative economic balance: -31444 € in current base scenario S0, for the functional unit considered. Therefore, it can be concluded that revenues generated from MSW in form of recovered resources are not enough to compensate the necessary expenses for MSW management activities, thus incurring in a global net cost. This gap in the recovery of expenses is actually borne by the citizens, through the MSW management fee. In analogous manner to the environmental analysis made in the previous section, an equivalent question arose concerning the feasibility for MSW management in the studied area to become economically sustainable. The alternative improved scenario S3 defined in subsection 5.5.2.3 was transformed into its monetary equivalent, using the same unitary prices previously applied in Section 5.4 for the baseline scenario. Only the price for RDF, introduced as a new valorisation option, was now set as 10 €/tonne as typical value – although this is highly dependent on the calorific power and the market fluctuations. The resulting balance is presented in Table 5.9.

Table 5.9. Costs and revenues inventory of scenario S3

Costs			Revenues		
Expenses	**Unitary cost**	**Total per FU**	**Incomes (sales)**	**Unitary price**	**Total per FU**
Municipal administrative staff	2.89 €/tonne	1456.86€	Beverage cardboard	564.00 €/tonne	3282.52€
Mixed MSW collection	46.45 €/tonne	15478.13€	Glass	36.00 €/tonne	1032.95€
Gate fee for mixed MSW	27.29 €/tonne	9092.91€	Paper/cardboard	173.00 €/tonne	11882.02€
Separate MSW collection and sorting	225.00 €/tonne	38269.24€	Fe metals (from separate MSW)	649.00 €/tonne	4462.37€
Waste management tax	5.39 €/tonne	1796.02€	Non-Fe metals (from separate MSW)	761.00 €/tonne	495.46€
			Plastics (from separate MSW)	545.00 €/tonne	20332.82€
			Compost	9.82€/tonne	717.97€
			Electricity production	114.75 €/MWh	4975.28€
			RDF	10.00 €/MWh	708.57€
	TOTAL COSTS	**66093.16€**		**TOTAL REVENUES**	**47889.97€**

The gap between expenses and revenues now equals -18203 € for scenario S3. The value is again negative, although it has been reduced in more than 40% with respect to baseline scenario S0. However, it has to be noticed that collection costs were kept constant when calculating the economic assessment of the alternative scenario. This assumption would be strictly valid only if the collection scheme was already working in a fully optimised manner, therefore making scale economy not applicable to the case. While this assumption may hold for the mixed MSW collection, it is less probable for separate collection, as shown by the much higher costs presented in Table 5.2, which are related to the low amounts of recyclables collected in comparison with mixed MSW. It is to some extent reasonable to expect that costs associated to separate collection and sorting of recyclable materials might be lower if source separation is increased, thus improving the economic performance of the system, and even reaching complete economic sustainability.

5.6. CONCLUSIONS

The performed LCA tests allowed to check to which extent would be required to improve the performance of the system – in terms of separately collected materials for recycling and energy recovery trough production of biogas and refuse-derived fuel (RDF) – in order to reach the neutrality of GHG emissions. In the resulting alternative scenario S3 finally generated (Figure 5.7), this condition was attained when all materials suitable for recycling were sent to the respective separate collection schemes, along with almost full recovery of biowaste and proper valorisation of the refuse in form of refuse-derived fuel (RDF). However, economic sustainability was still not achieved, although the balance looks considerably better than in the baseline scenario.

In fact, the result allows to understand the different influence of the system elements, depending on the analysed perspective: the driving factors in environmental performance are different from those governing the economic outcome. For instance: collection of mixed and separated MSW accounts for more than half of the total costs, with labour costs being the largest expense – as shown in a detailed study on waste collection costs by Sousa et al. (2018). On the contrary, collection has only a slight influence on the environmental assessment, contributing to 11% of total GHG emissions. On the other hand, the recovery of recyclable materials is the main source of income, as well as the most positive process regarding GHG emissions outcome. But this recovery alone is not enough to overcome the negative gap. Other valorisation activities, such as the electricity generation derived from biogas, compost and RDF production, also contribute to avoid landfilling of MSW and reduce the consumption of primary materials. Even if these secondary processes – except electricity generation – are not so profitable in economic terms, they are necessary in order to, as much as possible, close life-cycle loops of products in the best available way – e.g. biowaste – or, at least, mitigate its harmful effects – the case of RDF.

In terms of life-cycle assessment, this would require a further analysis performed from a consequential perspective, suitable for a future study. The same consideration would be expected for the performance of the MBT facility: less presence of materials other than biowaste would contribute to a better separation of the biowaste, which would thus be sent to anaerobic digestion process with less contamination and hence, the biogas production yield may be increased.

Nevertheless, it can be concluded that it is technically viable to achieve sustainability of MSW management even in complicated situations like the one analysed in this work. Notwithstanding, an optimal result would be only obtained through a careful combination of all the available options to reach this goal.

5.7. ACKNOWLEDGEMENTS

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Section III.

Modelling PAYT waste management with a consequential perspective

6. Consequential life cycle assessment as a governance tool for waste management

Álvaro Fernández Braña

ABSTRACT

A consequential Life Cycle Assessment (c-LCA), previously coupled to an application of Agent Based Modelling (ABM) was developed with the goal of assessing the evolution of environmental impacts derived by a change in MSW management pricing policy. The assessment was based in a case study for implementation of a unit based pricing (UBP) scheme for MSW management, which took place in a neighbourhood of the Portuguese city of Aveiro. The ABM was modelled taking into account the demographic characteristics of the population and their disposition to adopt more sustainable practices related to MSW management, namely: separating recyclable materials, reducing consumption and practising home composting. The model was calibrated to match the results actually obtained in the real case study, regarding the number of agents (equivalent to households) adopting each of the practices mentioned. Those numbers corresponded to a reduction of overall MSW generation, as well as an increase in separate collection, which in the c-LCA meant that a decrease of the environmental impacts due to landfilling, but an increase in the impacts due to MSW collection and recycling. Further hypothetical scenarios generated with the model showed that successive increases in the tariff pressure would persuade more residents to adopt more sustainable practices, although the environmental impacts associated would not follow a proportionally linear trend due to changes in the waste composition and scale economies.

6.1. INTRODUCTION

The success of a given public policy ultimately relies on the willingness of citizens to adapt their daily behaviour to the goals intended by such policy. When addressing policies dealing with municipal solid waste (MSW), a major issue arises regarding the most convenient pricing policy to be applied. The choice of MSW pricing policy is a relevant aspect to be considered, provided that it may become a powerful tool towards achieving the goals stressed by the currently demanding sustainability paradigm: increasing the rate of recycling – in the short term – and reducing the overall MSW generation – in the medium and long term.

A particular approach for establishing a MSW pricing policy is that generically known as unit-based pricing (UBP) or "pay-as-you-throw" (PAYT), that is: a variable fee relying on the actual quantity of residual (i.e. mixed or unsorted) MSW discarded to the collection system by a particular household or establishment, measured either by volume or weight. This kind of policies have already demonstrated their ability to increase the source-separation rates of recyclable materials by citizens – since these are excluded from billing –, and even reduce the overall MSW generation (Reichenbach, 2008, Elia et al., 2015, Wright et al., 2019).

The main reason for explaining the success of PAYT/UBP refers to this kind of policies being perceived as a fair method of charging waste generation (Batllevell and Hanf, 2008; Batllevell et al., 2008; Dahlén and Lagerkvist, 2010; Triguero et al., 2016), according to the "polluter pays" principle (PPP): the more waste a user produces, the more the user will pay. In parallel, it provides for the population an (economic) incentive to improve their management of household waste. Some authors (Bel and Gradus, 2016; Petryk et al., 2019) have investigated the effect of prices on MSW management performance. However, the question remains open regarding whether or not the potential money saving enabled by variable pricing represents an incentive strong enough for inducing a behavioural change of the population about their respective waste generation and management practices. In fact, the published research on UBP applied to MSW has shown that money saving alone – even though not without a significant role – is not the only "driving force" governing the behaviour of citizens when PAYT policies are adopted. For instance, Dijkgraaf and Gradus (2009, 2015) assessed in their studies for the Netherlands how the most environmentally-friendly communities – which would be indeed the "best performers" in terms of recycling rates and reduced MSW generation – are more likely to promote UBP for charging MSW, thus demonstrating that environmental awareness has as much influence – or even more – than price for determining behaviour towards household waste. The same issue was addressed by other works like those of Allers and Hoeben (2010) and Bueno and Valente (2019).

Besides price, environmental awareness and other "classical" socio-demographic factors already studied such as age, housing type or income – see for instance the works by Grazhdani (2016) and Triguero et al. (2016) –, the other main driving force governing household waste behaviour practices is that corresponding to social norms. As pointed by some authors (Jones et al., 2010; Boetziagas et al., 2020), the acceptability and willingness to collaborate with PAYT schemes depends on what they defined as "social capital", a concept which encompasses the level of trust between citizens themselves and towards governmental institutions. In communities with low "social capital", citizens tend to justify the disrespect of norms, then favouring undesired behaviours: the inappropriate disposal of household waste (open dumping, burning, "waste tourism", etc.); however, it is also possible that initial reluctance towards PAYT schemes is later mitigated in view of the benefits brought by this system (Dunne et al., 2008; Brown and Johnstone, 2014). The study of Thøgersen (2003) offers an interesting overview from a psychological perspective of possible behavioural responses to UBP, in which two main motivations to collaborate are defined: those “extrinsic”, aimed at obtaining a perceptible outcome – e.g. saving money – and those “intrinsic” – i.e. not directly related to a reward but justified because “it is the right thing to do”. Among the latter, and in coincidence with Dunne et al. (2008), it is possible to further distinguish between “personal norms”: those based on moral reasons, internalised as an own responsibility – for example: as a consequence of environmental awareness – and “social norms”: those prescribed by social pressure, expecting that others will act in the same way. This social pressure can be either real (in form of laws and regulations) or imagined (behave like other community members just to prevent conflicts).

Most of the published literature mentioned above focus its research efforts on applying statistical methods to demonstrate the benefits achieved by PAYT/UBP in form of mixed MSW reduction for specific study cases, trying to elucidate the relative contribution for each one of the behavioural driving forces in action. Based on this previous knowledge, this paper tries to perform a simulation of the future evolution of a PAYT/UBP application, making use for this purpose of the Agent-based Modelling (ABM) procedure. ABM is a modelling tool which allows to simulate complex interdependent systems, where the core of the simulation relies on the behaviour of a set of individuals (“agents”), who are able to take their own decisions as an adaptive response to the changing circumstances of the environment; for a full description of the methodology see, for instance, Macal and North (2005).

ABM has been applied in a wide range of research areas (Macal and North, 2005), many of them approaching the decision-taking process by human communities subject to particular circumstances. These works encompass also the field of waste management, where ABM has been used as a tool for optimising the planning of MSW collection routes (Nguyen-Trong et al., 2017), analysing the influence of economic incentives in waste treatment market (Skeldon et al., 2018), or assessing the performance of MSW management plans (Miranda de Souza et al., 2021). But, the most interesting for the case here addressed is the application of ABM from a sociological perspective. Meng et al. (2018) developed for the Chinese city of Suzhou an ABM model representing recycling, which citizens were assumed as agents who would adopt different individual attitudes regarding MSW source separation depending on their willingness to spend money and time to do so, but also on the attitude taken by their neighbours. The model developed in this work follows a similar structure, but here focused on the analysis of the specific effect of an UBP tariff, which as far as the authors know, is a particular application of ABM not appreciably published up to this date.

Nonetheless, the work here presented entails a second aspect, aimed at the quantification of the environmental effects derived from the adoption of the UBP tariff due to the change of behaviour induced on citizens. Given that this change has an intrinsically dynamic nature, Consequential Life Cycle Assessment (C-LCA) was selected as an adequate methodology to be applied. The ability of C-LCA to assess the long-term environmental consequences of strategic decisions makes this analytical procedure a rather suitable tool for assistance in the design of waste management policies.

The application of C-LCA for analysis of MSW waste management systems has been gaining presence in the published research, as shown by the review of Bernstad Saraiva et al. (2018). The use of C-LCA in this context is justified by the need of evaluating the future impact of taking a given decision – e.g. a change of policy, such as introduction of UBP – in a particular moment, which will alter the outcomes of MSW management, further inducing consequences beyond the scope of the MSW management system. Provided that, in this case, the environmental consequences derived from PAYT/UBP, have their roots in the behaviour adopted by citizens when dealing with their household waste, the combination of ABM with C-

LCA seemed an adequate procedure to address the issue. Firstly, the application of ABM modelling allows the simulation of every individual citizen behaviour, thus generating a possible scenario of MSW management after the introduction of PAYT. Then, secondly, the environmental implications of that given scenarios are assessed by means of C-LCA. This is the combined approach suggested by Querini and Benetto (2014, 2015), in their case applied to analyse the population willingness to buy an electric vehicle. The same methodology was selected by the authors to be applied in a particular study case regarding the potential adoption of a PAYT/UBP tariff scheme for MSW. Its development and results will be presented in the following sections of this work.

6.2. AGENT-BASED MODEL

6.2.1. Case study

The case study selected for the testing of the model developed correspondend to the neighbourhood in the Portuguese of Aveiro which was selected as pilot area for the testing of a variable PAYT/UBP pricing scheme for MSW within the scope of the European LIFE PAYT project. The population consists mostly of medium-high income families. The number of inhabitants was counted in 1153 in the last available Portuguese census – which took place in 2011, so the number is likely to have changed. However, no more recent data have been still made public, so that it should be taken as an approximate estimate. The inhabitants are distributed into 441 inhabited households, thus resulting in an average occupation rate of 2–3 people per household.

The large majority of households (91%) corresponded to flat apartments located in block buildings with different heights. The remaining 9% corresponds to detached houses, which usually possess a garden as part of the premises.

Regarding the age distribution, 33% of the inhabited households have some member or more under 15 years old, while in 16% there is at least one person above 65 years old. In the remaining households all members are within both ages. The distribution of these age subgroups with respect to housing type is almost the same than that mentioned before for the total population.

6.2.2. Agent-based model structure

The agent-based model designed in this study was programmed and implemented making use of the open-source ABM software NetLogo – version 6.2.0 – originally authored by Uri Wilensky (Wilensky, 1999). The modelling development procedure followed is explained with detail in the next subsections.

6.2.2.1. Modelling of agent population

The first step when applying the ABM methodology corresponds to the definition of the “agent” itself. The whole set of agents constitutes the virtual “population”, which is the object of study. Every agent is characterised by a set of attributes, which are common for all agents but take different values for each of them. The agent is able to take decisions with autonomy, but conditioned by the particular features attributed. Obviously, these attributes should consist on properties and/or variables which are relevant to the issue studied, here: generation and management of MSW.

In this case, it was established that the virtual population should correspond to the population of the studied neighbourhood, so that an individual agent is equivalent to a household – understood as the group of people living on the same home. That means that the total number of agents equals 441; i.e. the number of inhabited households in the neighbourhood.

For the sake of simplicity, it was chosen that the set of attributes given to the agents consisted on just three relevant variables, easily obtained from the Portuguese census. Each of these variables is allowed to take only two different values, namely:

- Is there any member of the household under 15 years old? – either “yes” or “no”.
- Is there any member of the household above 65 years old? – either “yes” or “no”.
- Type of dwelling – either an “apartment” (in a multi-storey building) or a “single house” (i.e. a detached or semi-detached building containing just one household, typically including a garden).

The influence of these demographic variables on the attitude of the household regarding MSW generation and management is attested by the published literature, as well as the own research experience of the authors. Even though the presence of children in the household may bring greater environmental sensibility, families with children tend to incur in higher consumption habits – and consequently, higher MSW generation – while in parallel, their housework practices are constrained by limited time availability (Beaumais and Prunetti, 2018). On the other hand, people in the age of retirement enjoy plenty of time availability and, although they might be perceived in general as more reluctant to behaviour changes, their acquaintance with a more frugal lifestyle and minimum consumption habits makes them be familiar with the reuse and recycling culture (Davies et al., 2005; Triguero et al., 2016; Bueno and Valente, 2019). Lastly, households consisting on a single house dwelling usually have more favourable conditions for adopting environmentally-friendly practices: more available surface area for storing separated fractions of household waste and the possibility of practising home-composting in the garden (Ando and Gosselin, 2005; Martin et al., 2006). Finally, it is noteworthy to remark that average income of households – another relevant variable for MSW generation – has not been included in this model, since it is considered that there is no great difference to this respect between the households of the neighbourhood, which correspond to a medium level of income.

6.2.2.2. Engagement levels

Once the agents of the model have been defined, the next step is to establish which is the decision that they will be forced to take when running the model. Here, the question to answer consists on how will the agents manage or dispose of their generated household waste; that is: they must choose one of the various management options available to them.

Each of these options requires a different degree of effort – “effort” means here in practice: time availability, adequate conditions in the household, personal capacities to learn and to physically perform the task and even money to buy equipment. The more effort an agent is willing to do for these activities, the more “engaged” is considered to be with the goal of proper household waste management. Attending to the practices available, the agents (households) are classified into four “engagement levels”. These levels range from the least engaged behaviour to the maximum commitment, namely:

- Level 0: No engagement

This level represents the absence of compromise with a sustainable MSW management, corresponding to the disposal of household in the easiest possible way, that is: in the form of mixed MSW without any effort to recover valuable materials. In a “flat fee” pricing system without any kind of incentives – such as PAYT – this behaviour is considered with no difference respect to more sustainable practices, being actually in some manner subsidised – since the other practices require more effort.

- Level 1: Source-separation

In contrast to level 0, this level 1 represents the minimum commitment currently expected from citizens in this study case: the separation at home of those recyclable materials subject to a specific collection circuit – namely three fractions: glass, paper and cardboard and plastic and metal packaging. As previously explained for level 0, since the current pricing system does not include incentives for sustainable practices, this behaviour relies on a merely voluntary effort. In parallel, this activity might be constrained by the available surface area, thus prejudicing the smallest households.

- Level 2: Reduced consumption

While level 1 is regarded as the basic contribution to sustainable MSW management which can be at least expected from citizens, level 2 involves a more advanced engagement, consisting on reducing the consumption habits of the households and, consequently, generate less waste – e.g. buying products with less packaging, or larger volume units of a product to optimise the use of packaging. This level is not expected to be generalised in the population without a strong policy of specific incentives.

- Level 3: Home composting

Following the progression, this level represents the maximum level of commitment with sustainability: the own management of biowaste through the practice of home-composting. This activity has been considered to demand a significant effort from the household members – including the acquisition of equipment. Moreover, it is constrained by the household typology and size, regarding whether there is or not a garden to place the composter and where to apply the obtained compost.

In every run of the model, each agent / household must choose one of the four options – levels of engagement – to dispose of their waste. According to the changing circumstances, the households adapt themselves to the most suitable level for their particular situation.

6.2.2.3. Driving forces

Following the definition of the agents and the establishment of different options for taking decisions, the subsequent stage corresponds to the characterisation of the factors that lead the decision-taking process. Considering the previous research already described on the Introduction, it was assumed that the behavioural response of citizens affected by the implementation of a PAYT/UBP policy for the tariff scheme of MSW generation is dependent on a combination of three main driving forces – both extrinsic and intrinsic –, which have been summarised as follows:

- Economic incentive (dependent on price).
- Personal commitment (motivated by environmental / civic awareness).
- Social pressure (determined by the attitude taken by neighbouring individuals).

The first (economic incentive) and second (personal commitment) driving forces are respectively represented in the model by means of two internal variables which are assigned to every agent: the first variable, *budget*, represents the sensibility of the agent to the driving force of price: agents with a higher value are able to pay more; the second variable, *willpower*, can be considered as a proxy for representing the environmental sensibility of the agent: agents with higher values are able to engage into levels which require more effort, like composting biowaste at home. The variables are given an initial random value (between 0 and 100 for willpower and between 0 and 500 for budget), which is partially consumed – i.e. decreased – every time the agent is forced to take a decision about how to manage household waste, thus corresponding to an iteration of the model. Once the decision is taken and executed, the variables are randomly recharged – in the best case, allowing the agent not to lose any point – and a new iteration of the model is ready to begin.

The decision adopted by an agent depends on the value taken by the variables at the moment of decision. Four equal ranges were set between the minimum and maximum possible values. According to the four levels of engagement previously defined in 6.2.2.2: for the budget variable, the higher its value, the less engaged will be the agent – the ranges going from level 3 to level 0 – and for the willpower variable, exactly the opposite: the higher its value, the more engaged will be the agent from level 0 to level 3. Each decision taken implies a “cost” for the agent, since the values of the variables will be decreased, however this penalisation varies depending of the level of engagement where the agent is acting: in the case of the willpower variable, the higher the level of engagement of the agent, the faster that the variable will be depleted, since advancing in a more sustainable lifestyle requires more effort. The effect in willpower is independent of the existence of a PAYT tariff, since this variable is only related to personal beliefs of the individual. This is, obviously, not the case when it comes to the budget variable: if a PAYT/UBP policy is not in place, the variable is not affected by the decisions taken, since the agent has no control on the tariff paid. However, if such a pricing policy is implemented, then the corresponding effect appears: the lower the level of engagement is, the faster that the budget variable will be depleted; that is: less sustainable behaviours are more penalised by higher tariff prices.

Regarding the third driving force acting in this context, social pressure, a different approach was followed in NetLogo to model its influence on decisions. As the agents move freely and randomly all over the field defined in NetLogo interface, it happens at every model run, that two or more agents may find themselves in the same place. This can be considered a social interaction, and hence the model establishes a random possibility that, if two agents are acting in different levels of engagement, one of the agents might be able to persuade the other to adopt its same level of engagement, representing then a kind of “word of mouth” effect.

6.2.2.4. Agent behaviour profiles

The particular circumstances and characteristics of every individual agent play a role in the decision-taking process. That means, consequently, that not all the agents react in the same way to the changes introduced; in practice: not all the agents have the same sensibility to the defined variables. To take into account this variability, four different profiles of agents have been considered in the model, attending to their established behaviour:

- Insensitive: they are not affected by any variable and not engaged (i.e. they choose level 0 by default). They represent people not concerned by the waste issue, reluctant to be influenced by any policy adopted. Only an intense social pressure by their peers, making them to feel strongly criticised by their attitude, would be a driver for a behaviour change.
- Civic-minded: they represent people who are aware of the problems related to waste and hence they act according to their willpower variable.
- Saver: this group is sensitive to the money they spent, so they act guided by their budget variable.
- All-sensitive: this last profile is a combination of civic-minded + saver profiles; they are sensitive to both variables budget and willpower.

Moreover, the properties of each agent influence also their response to the driving forces, in the way already explained in subsection 6.2.2.1:

- Families with children (under 15) are penalised when advancing to better practices, due to time and money constraints.
- Families with older people (above 65) are, on the contrary, favoured due to their time availability and reduced consumption habits.
- Families who live in detached houses are also favoured for adopting more sustainable practices, especially composting, due to their increased space availability.

6.3. CONSEQUENTIAL LIFE CYCLE ASSESSMENT

6.3.1. Goal and scope

The Life Cycle Assessment here presented is an update of the work published by the authors in two previous publications (Fernández-Braña et al. 2019, 2020). Since the primary objective was then to evaluate the situation of the MSW management for the target area, prior to the introduction of a PAYT/UBP tariff, an attributional modelling framework was selected – although system expansion was also applied to account for the environmental credit obtained for the recovery of resources an energy from MSW. Now, the same MSW system was adapted to the new situation required by the implementation of a PAYT/UBP tariff scheme and further expanded to include all the relevant effects derived from this change. Although the system here analysed as a case study is small, the consequences derived from a specific policy adoption might be large, so that according to the ILCD Handbook, this corresponds to a type B goal decision-context situation (JRC–IES, 2010), hence concordant with a consequential approach (Laurent et al., 2014).

Since the main function provided by the system here analysed remains the same as in original assessments – providing an adequate MSW management service to the population – (Fernández-Braña et al. 2019, 2020), the functional unit (FU) remains also the same: total mass of MSW collected in the target area during one year (a complete cycle of seasonal variation of the MSW generation pattern), encompassing mixed (unsorted) MSW and the three fractions separately collected: glass, paper/cardboard and plastic/metal packaging. Other separate collection fluxes were excluded from the scope of this study. Regarding biowaste, a dedicated separate collection for this fraction does not currently exist in the target area – although it is likely to be implemented in the near future –, so for this study it was treated as part of the mixed/residual MSW.

As cut-off criteria for the analysis of the unit processes, it was decided that materials sent to recycling as end-of-life (EoL) treatment were to be considered as avoided waste, i.e. no environmental burdens were allocated to the system originating that material.

6.3.1.1. Data collection and assessment methods

The data concerning MSW generation in the studied area were obtained in field characterisation campaigns, supported by the project. These data were later complemented with the historic register of MSW collection at city level, kept by the municipality, and the declarations of the responsible company for MSW management. The municipality provided as well the description of the collection system – equipment and operating parameters – which was deepened with the information from equipment producers, where possible. The modelling of the MBT and sorting facility was based on the information publicly declared by the managing entity – including a summary of mass balances and operating performance. For further situations where primary sources of information were not available, the modelling process was chiefly based on ecoinvent 3.5® life cycle database (ecoinvent Association, 2018). Other assumptions are explained hereinafter where appropriate.

The modelling of the Life Cycle Inventory and subsequent environmental assessment was performed with the SimaPro commercial software – version 9.0.0. – (PRé Sustainability, 2019). A set of relevant impact categories was defined for the characterising the assessment: global warming potential (carbon footprint), contribution to photochemical ozone formation, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, human non-carcinogenic toxicity, mineral resource scarcity and fossil resource scarcity and, finally, land use. The selection was based on common practices found in similar LCA studies of MSW management (Andreasi Bassi et al. 2017; Bernstad Saraiva et al., 2017; Pires et al. 2017; Ripa et al. 2017). Additionally, the land use category was included attending to its importance for assessing processes concerning biomass valorisation and paper recycling (Laurent et al., 2014). The impact assessment method chosen ReCiPe Midpoint (hierarchist perspective) Version 1.03 (Huijbregts, et al., 2017), justified by its completeness in coverage on the categories of interest. An exception was made for carbon footprint, where the 2013 IPCC characterisation methodology for a 100-year period (version 1.02), was applied – since it is more recent than that included in ReCiPe methodology.

6.3.1.2. Definition of the studied system: boundary setting and affected processes

The MSW management system in target area consists in a Mechanical-Biological Treatment (MBT) facility for treatment of mixed MSW collected from households through collective street bins and transported to the facility by MSW collection and compaction vehicles. In the MBT, recyclable materials – namely metals and some plastics, mostly polyethylene (PE) and polyethylene terephthalate (PET) – are recovered for recycling in a first pre-treatment stage. Thereafter, the remaining wastes are processed by means of anaerobic digestion, degrading most of the organic matter to obtain biogas, mainly composed of carbon dioxide (CO_2), methane (CH_4) and water (H_2O). The methane content makes biogas a suitable fuel for feeding a set of gas engines coupled to electricity generators, which provide all the electricity required for the MBT itself, as a well as an energy surplus which is externally sold and delivered to the national power distribution grid. The heat produced by the biogas combustion is not externally employed out of the MBT. The semi-solid residue of anaerobic digestion, still rich in organic matter, is dehydrated through centrifugation and subjected to a two-stage composting process: firstly, closed tunnel composting with forced aeration and secondly, a maturation phase in indoor windrows with natural ventilation and regular turning. The resultant compost is allowed for field application as a soil amendment and hence, sold to local farmers. On the contrary, the liquid extracted from centrifugation is not reused for fertilisation like in other similar facilities, but either reinjected in the anaerobic digester or sent to the wastewater treatment plant to be later discarded in the environment.

Regarding the recyclable materials which are separately collected, this are also sent to the same facility, where the sorting process and packing of the useful materials to be later delivered to recycling industries takes place. The rejected materials from the MBT itself and from the several sorting processes are currently disposed of in a sanitary landfill next to the facility; although the original intention for this rejects was to produce a refuse-derived fuel (RDF) to be sold, for instance, to the Portuguese cement industry, this is not currently happening due to concurrence of foreign RDF – namely, from the United Kingdom – where waste exportation is cheaper than paying landfill taxes.

The whole system to be analysed is represented in Figure 6.1 (including also MSW collection within. The previous description comprised the unit processes labelled as *foreground system*, that is: in this case, those intrinsic to MSW management activities and under direct control of the system managers (JRC–IES, 2010). Solid arrows represent the main reference material flow constituted from MSW collected and delivered to the MBT facility for treatment. After progressing through several unit processes representing valorisation and recovery activities, MSW is transformed into several products: electricity from biogas and compost, both obtained from biological treatments of organic matter, and materials for recycling (glass, metals, plastics and paper/cardboard) which were separately collected or recovered from mixed MSW. Otherwise, rejected MSW is disposed of in the sanitary landfill, whereas another fraction represented by a dotted arrow is released to the environment in form of greenhouse gas (GHG) emissions, (chiefly CO_2, CH_4, H_2O) resulting from biologic degradation and also treated wastewater.

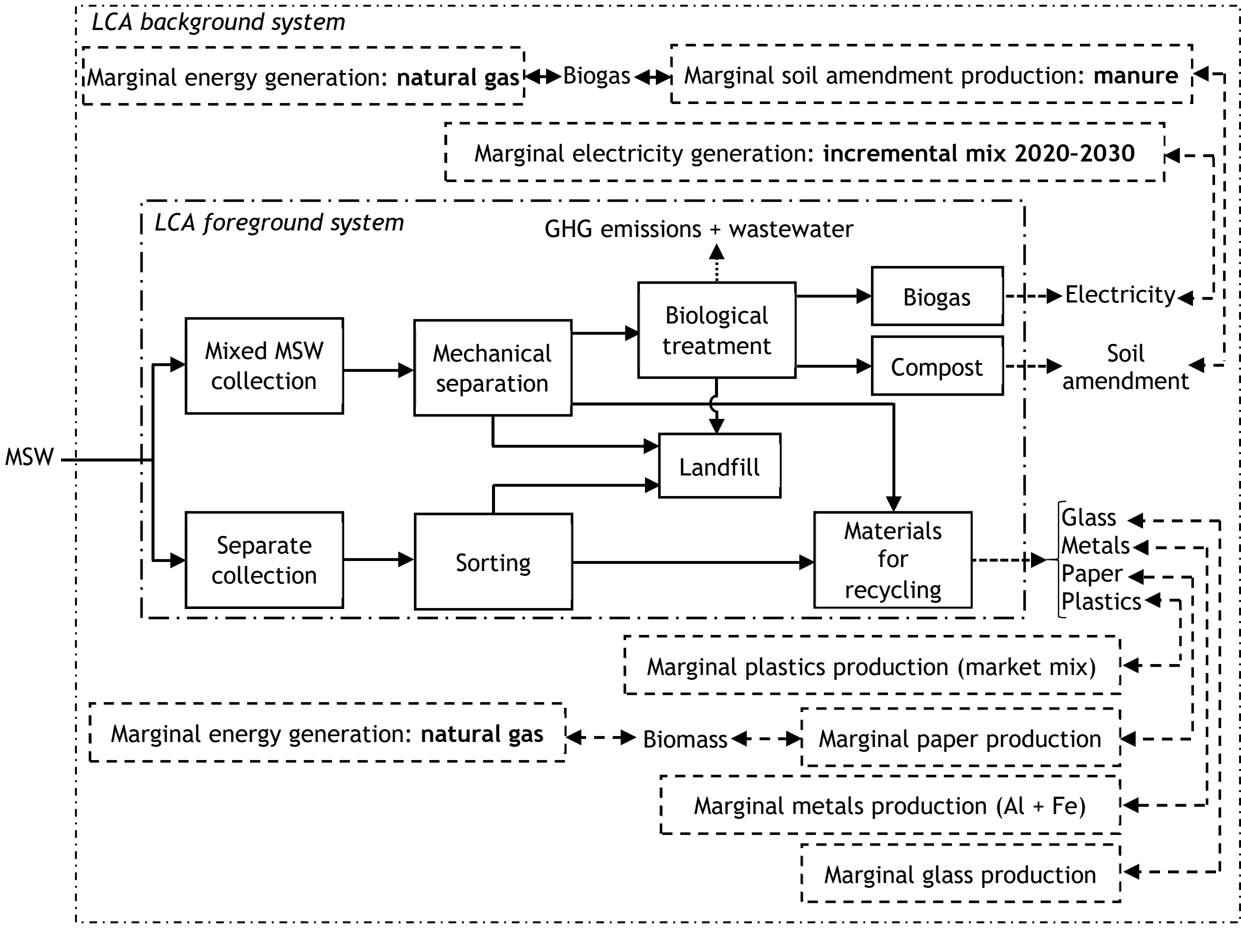

Figure 6.1. Representation of the analysed system with corresponding boundaries

Furthermore, the consequential point of view implies an expansion of the system assessed, in order to include in the assessment other unit processes directly and indirectly affected by changes of the functional unit induced in the foreground system (Ekvall and Weidema, 2004), but out of control by foreground system managers: these affected processes correspond to the *background system*. Each one of the affected processes respectively corresponds to the *marginal supply* displaced by a small (marginal) change in the demand of each one of the resources obtained from MSW valorisation. The interaction between resources and affected processes is represented by dashed arrows.

Compared to the whole background system, the changes introduced by the variation of MSW generation induced by the adoption of a PAYT/UBP tariff were considered small enough – even if such a tariff was to be generalised –, to correctly apply the methodological framework proposed by Weidema et al. (1999, 2009) for identification of the marginal suppliers affected: identifying scale and time horizon of potential changes, market limits, market trends and changes in supply and demand. This identification was performed for each of the displaced products presented in Figure 6.1, and explained with detail in the next section.

6.3.2. Life Cycle Inventory

Based on the framework previously explained in 3.1, 3.2 and 3.3, a Life Cycle Inventory (LCI) was developed from the data gathered, encompassing the environmental modelling of both the processes included in the foreground system – the main stages of MSW management: collection, transport, treatment, valorisation and final disposal – as well as the processes making part of the background system – those affected by the outputs of the foreground system, Since the modelling of the foreground system was already addressed in the previous published works already referred (Fernández-Braña et al., 2019, 2020) here it is only presented in a summarised manner (in 3.4.1 and 3.4.2), while more focus will be put on the description of the background processes (from 3.4.3 to 3.4.6).

6.3.2.1. Modelling of the MSW collection system

The modelling of MSW collection in the target area was structured according to: (i) the necessary equipment: household bins to collect the waste at home, plastic carrier bags to transport it, collective street containers for mixed MSW and the different separate collection fractions – three main fractions were considered for this case: glass packaging, paper/cardboard packaging and plastic/metal packaging and, finally the MSW collection vehicles; (ii) the resources needed for actual collection, namely: fuel (gasoil) consumed by collection vehicles. The amounts of materials required for producing the equipment are shown in Table 6.1, referred to the functional unit. Prior to the implementation of the experimental PAYT tariff the collection of mixed MSW fraction comprised on 26 standard street containers with 1100 L capacity, which upon the project were updated with an electronic access control based on user cards. Separate collection consisted on 5 bring banks, formed by three containers – one for each separate fraction. With the project, the number of bring banks was increased to 10, to receive a larger amount of separated materials.

Table 6.1. LCI for MSW collection equipment

Equipment	Lifespan	Materials	Mass per FU without PAYT	Mass per FU with PAYT
Carrier bags	----	HDPE	1435 kg	996 kg
Household bins	7 years	PP	335 kg	335 kg
Street containers for mixed MSW	14 years	HDPE	70.3 kg	70.3 kg
		Steel	8.5 kg	14.0 kg
		Rubber	2.8 kg	2.8 kg
		Electronics	----	4.6 kg
Street containers for separate collection	14 years	HDPE	163.0 kg	326.0 kg
		Steel	5.6 kg	11.2 kg
		Rubber	0.3 kg	0.6 kg

The modelling of the collection vehicle was not included in Table 6.1, since it was taken from ecoinvent database (Doka, 2007) and EMEP/EEA guidebook (EMEP/EEA, 2016) for indirect emissions, fixing its useful lifetime in 10 years. Fuel consumption is presented in Table 6.2; the value for mixed MSW was supplied by the municipality while those for separate collection are taken from the specific study of Moreira Monteiro (2013).

Table 6.2. Fuel consumption for collection of different MSW fractions

MSW fraction	Fuel consumption (L/tonne)	Distance travelled (km/tonne)
Mixed MSW	4.23	5.84
Separate glass	19.79	32.32
Separate paper/cardboard	23.77	44.17
Separate plastic/metal	50.55	101.47

6.3.2.2. Modelling of MSW treatment

The treatment phase of MSW was modelled according to the structure presented in Figure 6.1. Firstly, it was necessary to establish the waste composition matching the functional unit at both moments, before and after the implementation of the project, which, as already mentioned, came from field and registered data. The composition is shown in Table 6.3.

Table 6.3. MSW composition

MSW flow	Component	Composition before PAYT (% mass)	Mass per FU before PAYT (t/year)	Composition after PAYT (% mass)	Mass per FU after PAYT (t/year)
Mixed MSW	Biowaste	59.80 %	268.6	68.45 %	213.4
	Textiles	10.50 %	47.2	7.36 %	22.9
	Paper and cardboard	9.00 %	40.4	4.59 %	14.3
	Plastics	5.60 %	25.1	4.05 %	12.6
	Glass	5.11 %	22.9	3.03 %	9.4
	Hazardous and WEEE	1.27 %	5.7	0.94 %	1.4
	Composites	1.12 %	5.0	1.29 %	4.0
	Wood	0.98 %	4.4	0.77 %	2.4
	Metals	0.79 %	3.5	0.85 %	2.6
	Fines	2.63 %	11.8	7.43 %	23.2
	Others	3.20 %	14.4	1.24 %	3.9
	TOTAL	**100 %**	**449.1**	**100 %**	**311.7**
Glass packaging	**Glass**	**100 %**	**5.7**	**100 %**	**16.4**
Paper and cardboard packaging	Paper and cardboard	99.0 %	28.3	95.5 %	54.5
	Others (reject)	1.0 %	0.2	4.5 %	2.5
	TOTAL	**100 %**	**28.5**	**100 %**	**57.0**
Plastic and metal packaging	Plastics	58.8 %	11.7	49.2 %	15.4
	Composites	11.5 %	2.3	9.7 %	3.0
	Fe-metals	13.6 %	2.7	15.5 %	4.9
	Non Fe-metals	1.2 %	0.2	1.6 %	0.5
	Others (reject)	14.9 %	3.0	24.0 %	7.5
	TOTAL	**100 %**	**19.9**	**100 %**	**31.3**
TOTAL (FU)			**503.3**		**416.4**

In the next step, these compositions served as a basis for calculating mass balances along the different treatment stages, whose final outcomes in terms of materials and energy recovered are presented in Table 6.4.

Table 6.4. Outputs of MSW treatment

Output	Amount per FU (before PAYT)	Amount per FU (after PAYT)
Electricity (MWh)	11.2	7.3
Compost (t)	5.1	3.3
Replaced Fe-metals (t)	3.5	3.8
Replaced non-Fe metals (t)	0.5	0.5
Replaced glass (t)	4.8	13.8
Replaced paper (t)	13.4	25.4
Replaced plastics (t)	14.2	12.0

In the case of mixed MSW, it is estimated that from the initial input of 449 tonnes per year, there are 126 tonnes which enter the biological treatment stage, being mostly biowaste. Thereafter, 15 tonnes of biogas are obtained for electricity generation, while 5 tonnes of compost are sold for agricultural application – the amount of compost produced is much larger (51 tonnes), but a great part is internally used as landfill coverage. The rest of biomass is degraded and released in form of gaseous emissions of carbon dioxide and water. The electricity obtained from biomass corresponded to 18.1 MWh, which after discounting the own consumption in the facility results in a net production of 11.2 MWh, as in Table 6.4. The biological treatment process, as well as the sorting and recycling processes of recovered materials, were modelled according to ecoinvent 3.5 database.

Finally, all the remaining waste mass which has not been either recovered, valorised or degraded is sent to the sanitary landfill as a final refused residue, whose corresponding composition according to mass balances is shown in Table 6.5.

Table 6.5. Composition of landfilled refuse

Component	Composition before PAYT (% mass)	Mass per FU before PAYT (t/year)	Composition after PAYT (% mass)	Mass per FU after PAYT (t/year)
Biowaste	47.7 %	149.5	62.6 %	153.2
Textiles	15.0 %	47.2	9.4 %	22.9
Paper and cardboard	11.4 %	35.7	4.6 %	11.2
Plastics	5.5 %	17.2	4.4 %	10.7
Glass	7.3 %	22.9	3.9 %	9.4
Hazardous and WEEE	1.8 %	5.6	0.5 %	1.3
Composites	1.6 %	5.0	1.6 %	4.0
Wood	1.4 %	4.3	1.0 %	2.4
Metals	0 %	----	0.4 %	1.0
Fines	3.8 %	11.8	9.5 %	23.2
Others	4.6 %	14.4	1.6 %	3.9
TOTAL	100 %	313.7	100 %	244.7

The environmental effects of landfilling were modelled adapting the procedure in the ecoinvent database for municipal waste disposal in a sanitary landfill (Doka, 2007) to the composition of Table 6.5.

6.3.2.3. Substitution of electricity generation

According to the previsions publicly presented by the Portuguese government in 2019 aiming at achieving carbon neutrality by 2050, (Portuguese Ministry of Environment and Energy Transition, 2019), electricity demand in Portugal is expected to continuously raise in the next decades, making necessary an increase of the generation capacity of more than 100% until 2050. This growth will be based almost exclusively on renewable energy sources, mainly wind power and solar photovoltaic cells; on the contrary, coal-based power plants have been already shut down in 2021 while natural gas plants will be gradually reducing their share in the generation mix until their final phasing out. Focusing on the period 2020–2030, coincident with the time horizon of this study, the marginal mix for electricity generation in mainland Portugal was developed, following and adapting the approach suggested by Muñoz and Weidema (2021).

Table 6.6. Portuguese electricity marginal mix between 2020-2030

Energy source	Installed power in 2020 (GW)	Projected power in 2030 (GW)	Annual capital replacement rate (%)	Net annual growth (%)	Marginal mix 2020-2030 (%)
Biogas	0.1	0.3	-3.3%	21.6%	1.6%
Biomass	0.7	1.0	-2.2%	6.9%	3.9%
Coal	1.9	0	-1.7%	-8.3%	----
Fuel-oil	0.7	0.2	-1.7%	-5.5%	----
Hydro-power (reservoirs)	2.6	3.3	-1.0%	3.8%	8.0%
Hydro-power (run-of-river)	1.3	1.7	-1.0%	3.8%	4.0%
Hydro-power (pumped storage)	2.7	3.4	-1.0%	3.4%	7.5%
Natural gas	5.0	3.7	-3.3%	0.8%	3.1%
Solar photovoltaic (centralised)	1.4	4.7	-3.3%	27.1%	30.2%
Waste incineration	0.1	0.2	-2.2%	20.5%	1.1%
Wind power (on-shore)	5.4	7.4	-5.0%	8.6%	37.6%
Wind power (off-shore)	0	0.3	-5.0%	105.0%	3.0%

The capital replacement rates in Table 6.6 were calculated as the inverse of lifetime, taken from NREL (2004, 2010). Energy sources with a net growth being less than their capital replacement rate are deemed obsolete and hence excluded from the future marginal mix. Regarding solar photovoltaic, only the contribution from centralised generation was included in the marginal mix, since even though decentralised (small-scale) generation is also expected to significantly increase in next years, that growth is expected to be based on the initiative of individual investors looking for more energy autonomy, rather than strategic large-scale investments in response to the increasing electricity demand. Electricity imports from Spain to Portugal were also excluded, given that they are hardly predictable in the long-term – they are dependent on the climatic conditions required for renewable generation – and, in any case, they are expected to be not relevant, since the intention of Portuguese authorities is to gain more energy autonomy.

6.3.2.4. Soil amendments

Compost has been typically addressed by many authors in LCA practice as a by-product of MSW management, whose main value relies on its fertiliser capacity, assumed as an alternative to conventional mineral fertilisers (Abeliotis et al., 2012; Beylot et al., 2015; Ripa et al., 2017). However, this fertiliser capacity, even though relevant, is rather low if compared to that of mineral fertilisers, due to the limited content in nutrients – namely, nitrogen (N), phosphorus (P) and potassium (K). To the knowledge of the authors, in the common agricultural practices found in Portugal, the main interest of compost application on land does not correspond to its nutrient content, but to its potential to provide organic matter to soil, i.e. as a structuring agent, or more generally a soil conditioner. Moreover, the acceptance of compost as a soil amendment is mostly explained due to it being an inexpensive product derived from biowaste, so that farmers are able to acquire the amount required for their needs without being constrained by its price. Notwithstanding, an economic constraint does actually exist if transport distance is taken into account, due to the relative low density of compost, as already pointed by Bernstad Saraiva et al. (2017). Consequently, the compost produced by MSW management facilities is usually distributed only to the local farmers located in the vicinity – the corresponding road transport distance has been set for this study at 50 km.

Some authors have "credited" the value of compost as soil conditioner in their LCA practices by attributing to compost the substitution of alternative products such as turf and straw (Boldrin et al., 2009, 2010; Hermann et al., 2011). Although correct from a theoretical point of view, since the functionality provided by these alternatives is roughly the same as compost, this substitution would be hardly applicable to real conditions, since at least in the context of Portuguese agricultural practices found in the vicinity of the case study area and familiar to the authors, it seems unrealistic that any farmer would ever consider turf or straw as soil amendments – as well as mineral fertilisers, as explained before –, due to evident price constraints: both are much more expensive materials than compost, with a more valuable field of applications than agricultural soil conditioning. Instead, it makes more sense to look at other inexpensive residual by-products, as alternatives to provide the same function as compost. For instance, manure originated in different kinds of animal farms has been a resource traditionally used as an organic soil conditioner and fertiliser, and hence suitable to be selected as a realistic alternative to compost. From a consequential perspective, animal manure is by its residual nature a constrained product, which is not produced in response to a demand for it but as waste originated by animal farming. A change in demand of manure as soil amendment would then result not in a change of its production, but in a change in the available offer of manure for other competing activities. The main alternative for manure valorisation is its processing through anaerobic digestion to obtain biogas suitable as fuel, most likely for power generation. In spite of this valorisation process not being currently very usual in Portugal, the national policies intended for achieving carbon neutrality promote this option as part of the better practices for sustainable management of farming effluents (Portuguese Ministry of Environment and Energy Transition, 2019); so it can be expected that in the near future this activity will be more present in Portuguese farms and hence, it could be considered as the marginal process affected by an increase in demand of manure for soil conditioning. Assuming the same composition of biogas substrate as in ecoinvent 3.5® database – 1 m^3 of biogas is generated by 4.8 kg liquid manure from cattle, 2.3 kg liquid manure from swine and 1.0 kg solid manure from cattle –, according to the previous function defined the substitution ratio respect of compost was based on the respective content of organic matter – taken from Sinaj et al. (2009), as shown on Table 6.7.

Table 6.7. Organic matter content in substrates (adapted from Sinaj et al., 2009)

Substrate	Relative composition of biogas substrate (mass/mass)	Organic matter content (kg OM/tonne substrate)
Bovine slurry (liquid)	55%	70
Bovine manure (solid)	12%	175
Swine slurry (liquid)	33%	36
Average biogas substrate	100%	71
Compost	----	214

In conclusion, the combination of manures selected contains about ⅓ of the organic matter found in the same amount of compost used as soil amendment and, if alternatively used for energy production, 37 kg of the manure mix would be needed to produce 1 m^3 of biogas, which in turn would replace 0.25 m^3 of natural gas (from ecoinvent 3.5) as marginal energy source, in view of the previsions shown before on Table 6.6.

6.3.2.5. Recycled paper and cardboard

The case of paper and cardboard recycling has been widely addressed in C-LCA, particularly regarding the alternative use of wood no longer derived for paper production (Bernstad Saraiva et al., 2018). In Portugal, almost all of the paper production is based on pulpwood obtained from *Eucalyptus spp.* plantations, located mainly in Portugal and neighbouring Spain – especially Galicia – making 89% of total; 73% is transported by road and 13% by railway (assuming an average distance of 200 km) –, and to a small extent imported from South America – the remaining 11%, transported by ship (assuming a distance of 6000 km). These data were extracted from the main Portuguese paper producer in 2020, others have similar supplier structures. For producing 1 kg of paper, 2.22 kg of wood are assumed to be necessary – from ecoinvent 3.5® database.

Being a fast-growing tree, *Eucalyptus spp.* constitutes also a suitable source of biomass fuel for energy production, which is expanding in last years in accordance with the perspectives shown in 3.1.1. It is reasonable to assume that *Eucalyptus spp.* biomass no longer demanded by paper industry will be available for biomass power plants, and consequently the marginal technology for energy production will be displaced; which, again accordingly to Table 6.6, corresponds to power plants based on natural gas. The equivalence here applied (from ecoinvent 3.5) is that 1 kWh of combined heat and power cogeneration requires 0.841 kg of wood biomass, as well as 0.107 m^3 of natural gas.

6.3.2.6. Recycled packaging materials: glass, metals and plastics

According to the modelling approach developed by Ekvall and Weidema (2004) for open-loop recycling crediting, the ability of a recycled material to replace either its non-recycled market equivalent or another equivalent recycled material with different origin depends on the price elasticity of the demand and supply for the material to be recycled. Actually, the situation in Portugal is that the price of materials recovered from MSW and suitable for recycling is fixed by government (Ministério do Ambiente and Ministério da Economia, 2016). Under this situation, there is no difference in price elasticities of demand and supply – both would be, indeed, infinite. In this particular case of equal elasticities, the result is that every amount of recycled material replaces a 50% mix of half non-recycled (primary) and half non-recycled (secondary) material, being both considered as marginal suppliers to the same extent. This theoretical assumption was considered as a reasonable simplification for this study. The affected processes for primary material production were selected from the ecoinvent 3.5® database, being respectively shown in Table 6.8.

Table 6.8. Primary products affected (replaced) by recycling

Material for recycling	Primary process affected	Quality loss factor
Glass	Global production of white packaging glass (without cullet).	1
Ferrous metals	Global market for pig iron from conventional blast furnaces.	1
Non-ferrous metals	European production for primary aluminium (cast alloy).	1
Paper and cardboard	Portuguese production of newsprint primary (virgin) paper.	0.83
Plastics	European production of a default (ecoinvent 3.5) mix made by: ▪ 46.1% HDPE ▪ 26.6% PP ▪ 18.3% PET ▪ 4.8% PS ▪ 4.1% PVC	0.75

In addition to the previous considerations, it was also decided to apply a penalising factor within the calculation of material substitutions, to account for the loss of quality experienced by some materials – namely, paper and plastics – when subject to open-loop recycling – the so called "downcycling". The approach is adapted from Bala Gala et al. (2015) and the values are also shown in Table 6.8.

6.4. RESULTS

6.4.1. Baseline: business-as-usual (BaU) scenario

The agent-based model was firstly initialised by modelling the target population, taking into account its demographic characteristics, as indicate by the survey made in the beginning of the project. The initial distribution, attending to the behaviour profiles defined in 6.2.2.4, is shown in Table 6.9.

Table 6.9. Initial behaviour distribution of population

Agent (household) type	Members age	Behaviour profile			
		Insensitive	Civic-minded	Saver	All-sensitive
Resident in apartment	Under 15	22	69	2	19
	Between 15-65	34	137	5	38
	Over 65	11	32	1	9
Resident in detached house	Under 15	2	6	0	6
	Between 15-65	4	19	1	15
	Over 65	2	4	0	3
TOTAL		75	267	9	90

Once the model is parametrised with these inputs, it was left to run without any tariff restriction in action – only the *willpower* was acting, since in baseline scenario, UBP has not yet been applied; thus corresponding to a business-as-usual (BaU) scenario, where no action has been taken yet. In these conditions, it was observed that the distribution of agents within the defined engagement levels tended to stabilise after some 15,000 iterations, as seen in Figure 6.2. This effect is indeed expectable, since once the agents are initially placed on their levels according to their variables, the only driving force able to provoke bulk movements is that coming from social interactions, which in turn decreases its influence over time, since less initially reluctant agents are left available to be persuaded.

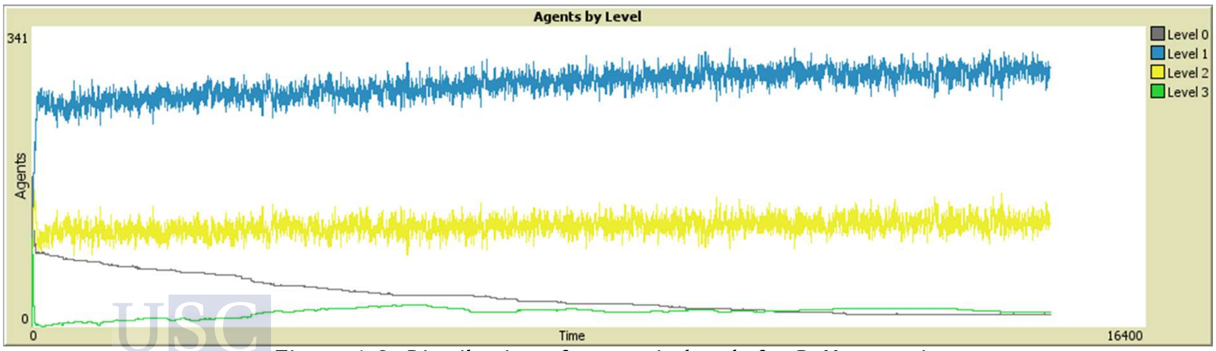

Figure 6.2. Distribution of agents in levels for BaU scenario

The relative distribution of agents in Figure 6.2 points to 3.2% of agents (roughly 14 households) acting in level 0, 66.0% in level 1 (~291 households), 26.8% in level 2 (~119 households) and, again 3.6% in level 3 (~16 households), a result which reasonably matches those given by the surveys made to the population, in which 3.8% placed themselves in level 0, 64.2% in level 1, 28.3% in level 2 and 3.8% in level 3. Actually, this approximation was achieved by refining the mechanism of depletion and regeneration of the internal variables as much as possible, in order to calibrate the model. The low numbers in levels 0 and 3 indicate that few people declared to be in the extreme positions: very few feel completely uninterested for the waste environmental, but also very few adopted the decision to compost their biowaste at home, besides the other environmentally-friendly practices. A large majority decided to contribute through the easiest option: separating recyclable materials, while a smaller part was available to modify their consumption habits in favour of generating less waste. This BaU scenario can be regarded as a sort of baseline representing the population behaviour, prior to the implementation of the PAYT/UBP tariff.

6.4.2. Scenario with PAYT

Surveys made after the implementation of the project showed a clear change in preferences of population: virtually all of the residents interviewed took some action to mitigate their mixed waste generation – that is, no one was classified in level 0 (0%). The preference between levels 1 and 2 had been to some extent inverted: 40.1% of households were just separating recyclable materials (level 1), but 54.0% had been able to reduce their consumption (level 2). Finally, home-composting (level 3) had increased, but only slightly, reaching 5.9% of households. The model was again parametrised, now incorporating the *budget* variable, until matching as close as possible the situation given by the survey, a result shown in Figure 6.3: 1.1% (5 households) in level 0 – 0 was not reached within 30,000 iterations, but the decreasing trend was constant towards extinction of this group, 37.9% in level 1 (~167 households), 54.4% (~240 households) in level 2 and 6.3% (~28 households) in level 3.

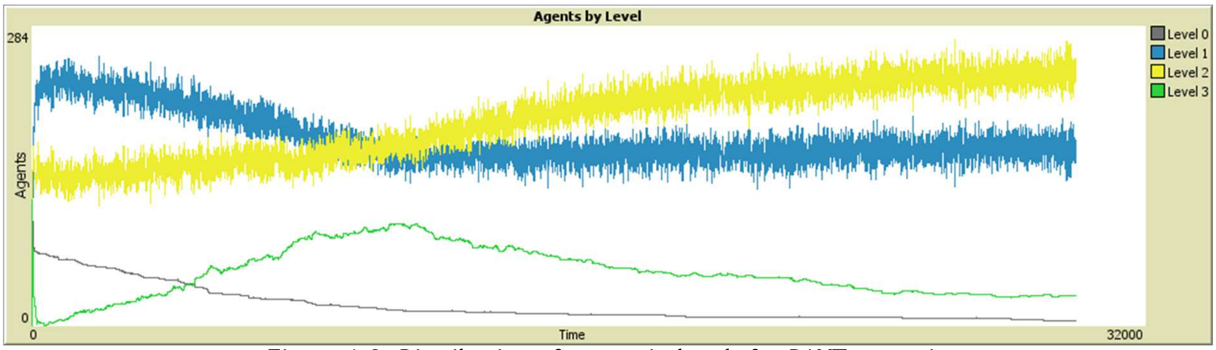

Figure 6.3. Distribution of agents in levels for PAYT scenario

6.4.3. C-LCA results

After the lasting period of the pilot experience with PAYT/UBP, the change observed in the amount and composition of household waste collected corresponded to that shown in Table 6.3 – reflecting the variation of the functional unit. Environmental assessments were conducted both in the beginning and in the end of the practical experience, based on the consequential LCI previously developed. The corresponding results are presented in Table 6.10 as global numbers, and in Figures 6.4 and 6.5 in relative form, showing the impacts distributed along the main stages of MSW management.

Table 6.10. Global C-LCA results

Impact category	Unit	Before PAYT	After PAYT
Global warming	kg CO2 eq.	168 447	143 557
Ozone formation, Human health	kg NOx eq.	197	301
Fine particulate matter formation	kg PM2.5 eq.	75	124
Terrestrial acidification	kg SO2 eq.	239	371
Freshwater eutrophication	kg P eq.	21.6	32.9
Terrestrial ecotoxicity	kg 1,4-DCB eq.	216 692	359 911
Freshwater ecotoxicity	kg 1,4-DCB eq.	172 330	93 332
Human non-carcinogenic toxicity	kg 1,4-DCB eq.	5 425 120	2 355 231
Mineral resource scarcity	kg Cu eq.	12.7	123
Fossil resource scarcity	kg oil eq.	1 935	7 613

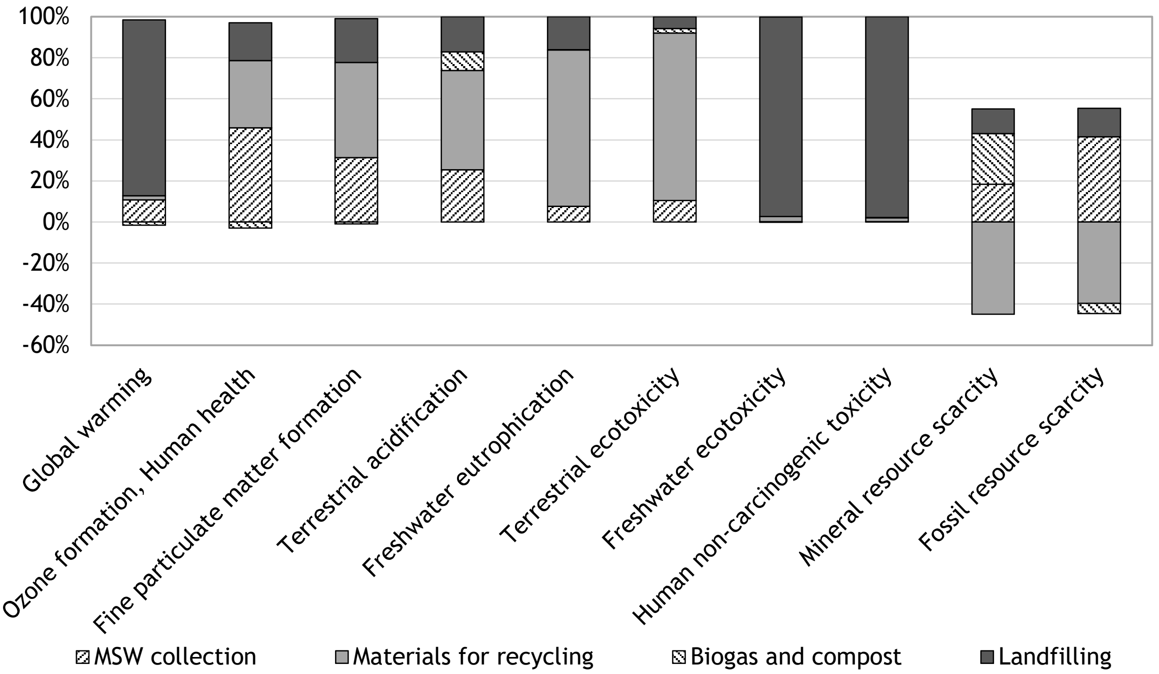

Figure 6.4. Relative distributed C-LCA results before PAYT application

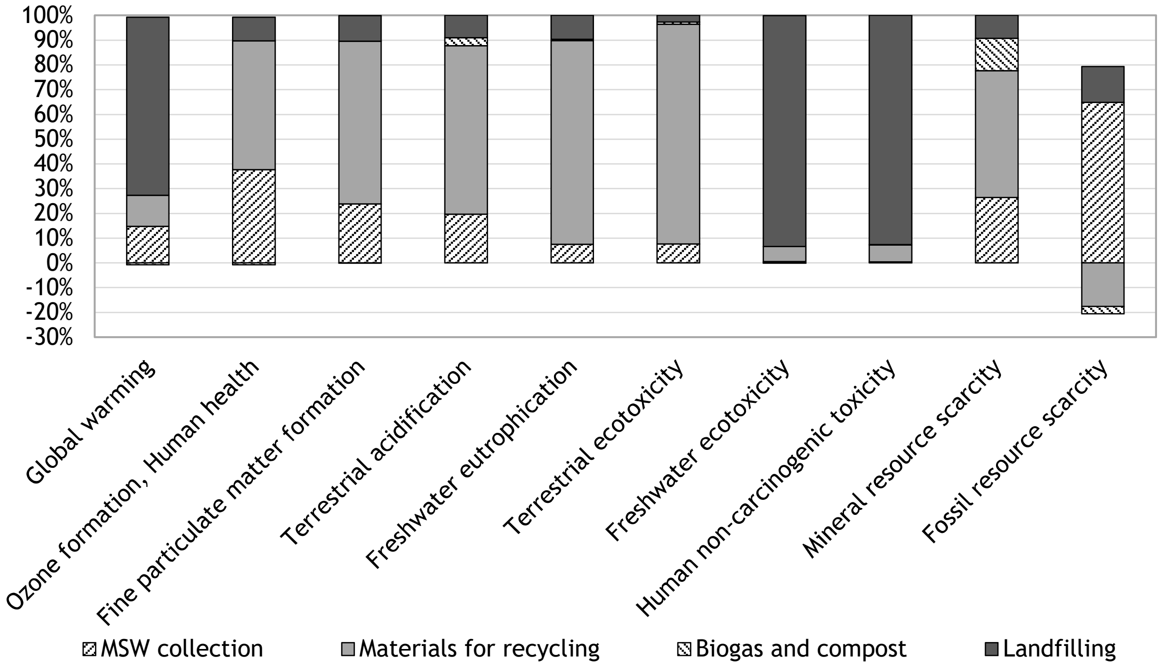

Figure 6.5. Relative distributed C-LCA results after PAYT application

As already pointed in Fernández-Braña et al. (2019), in the toxicity categories, the impacts are dominated by the harmful effects of landfilling, as well as in the contribution to global warming – due to methane fugitive emissions. MSW collection has its major influence in the fossil resources consumption, due to the fuel consumed by vehicles, while it is also relevant in categories referred to air quality, particularly in ozone formation, due to NO_x emissions from vehicles. In the rest of categories – remarkably in freshwater eutrophication and terrestrial ecotoxicity – the most relevant contribution is that coming from recycling activities. Given that the environmental credit attributed for the substitution of primary raw materials equalled only 50% of the recycled quantity, in that case the benefits obtained in form of prevented impacts were not enough to offset the actual impacts themselves originated by the recycling processes.

Furthermore, the replacement of future Portuguese electricity generation mix through use of biogas from biowaste revealed itself to be environmentally beneficial in all categories. However, the opposite was found when directly comparing natural gas-based generation replaced by either that biomass-based (from avoided paper production) or by biogas, the latter being only better regarding its contribution to global warming and consumption of fossil resources – otherwise an obvious result.

The overall comparison in relative terms for all categories between the two moments, before and after PAYT, is presented in Figure 6.6.

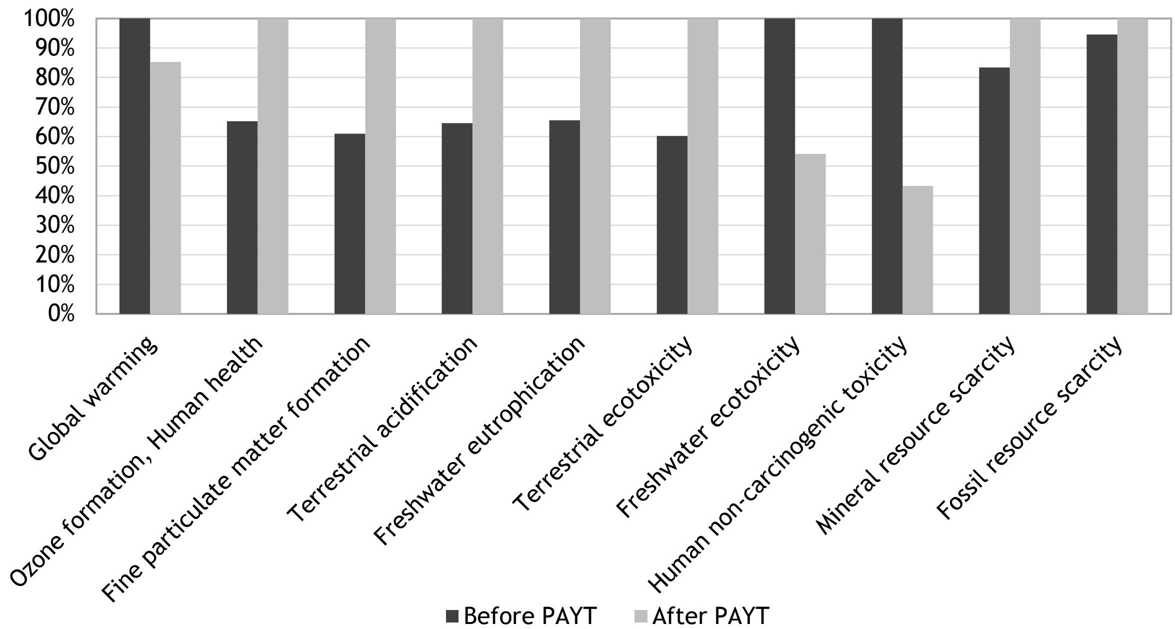

Figure 6.6. Relative comparison between C-LCA results before and after PAYT application

In this comparison, it can be observed that the decrease of mixed MSW collected allowed a reduction in the impacts caused by landfilling and biological treatments – especially remarkable in freshwater ecotoxicity and human non-carcinogenic toxicity, with roughly reductions to a half of the previous value –, but on the other hand, the increase of recyclable materials collected resulted in an expectable growth of the impacts associated to collection and recycling. Nevertheless, in normalised terms the global impact of recycling is always less harmful – thus, preferable – than that of landfilling.

6.4.4. Coupling ABM/C-LCA and generation of alternative scenarios

The results obtained in C-LCA can be linked with those scenarios generated in ABM for the corresponding initial and final moments of the practical experience conducted. Hence, the calibration of the model was completed, being then possible to generate new alternative scenarios by altering the parameters governing the behaviour of the agents.

Provided that the monetary value of the tariff is not by itself a critical factor for change – since its impact in the household budget would be anyway always low, the *budget* variable should be understood in a dimensionless sense. Actually, the new collection system was implemented and entered into action before any real alteration was introduced in the pricing scheme during the pilot experience, whereas the observed change in behaviour indeed happened. This reinforces the conclusion that it is not the price by itself the driving force, but the awareness regarding pricing: realising the fact that the waste generated is now charged according to the amount.

Regarding the model mathematical structure, if pricing is set as a linear function – since its dependence on the waste amount can be assumed as linear – and the minimum price is that assigned to the households in level 3, then the slope of the resultant line is the parameter that can be altered in order to take advantage of the population awareness regarding price: a higher slope means a higher increase of prices according to less sustainable behaviours, thus promoting a more forceful response from the agents. This finding is in line with the somewhat similar approach proposed by Ukkonen and Sahimaa (2021), in their case referred to service levels –

i.e. different levels of use of the collection service corresponding to different prices –, although in their model they included an internalisation of the price paid for the separately collected recyclable fractions which is not applicable to the usual scheme in Portugal.

A graphical representation of different tariff levels fitted to lines with varying slope is shown in Figure 6.7, taking as a base the line which corresponds to the situation after PAYT implementation of Figure 6.3.

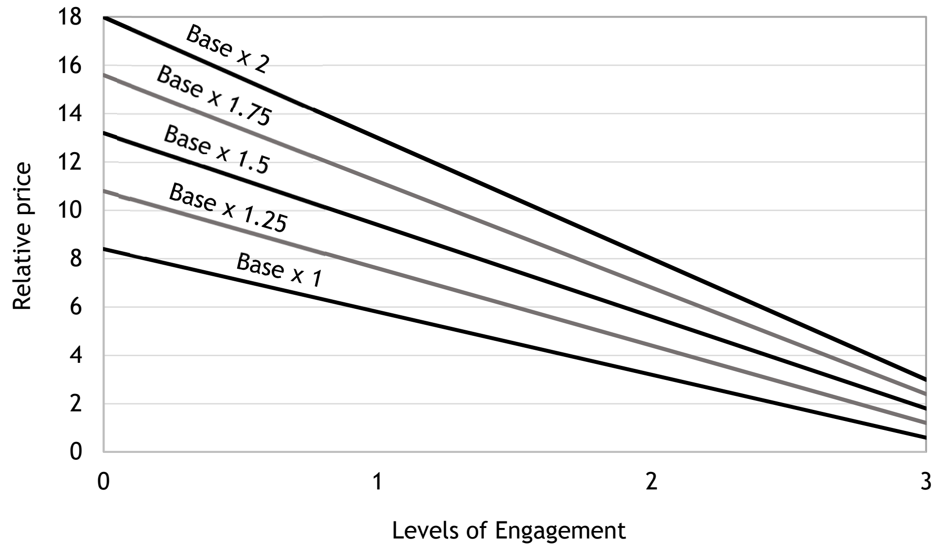

Figure 6.7. Progressive price slopes

If the desire of the system managers is to obtain a further reduction of MSW, then it is necessary to increase the tariff values, following the pattern of Figure 6.7, with the aim of getting more households advancing into higher levels of engagement. For instance, two model simulations were done, respectively selecting from Figure 6.7 the tariff slopes increased by a factor of 1.5 and by a factor of 2. The respective ABM results are presented in Figures 6.8 and 6.9.

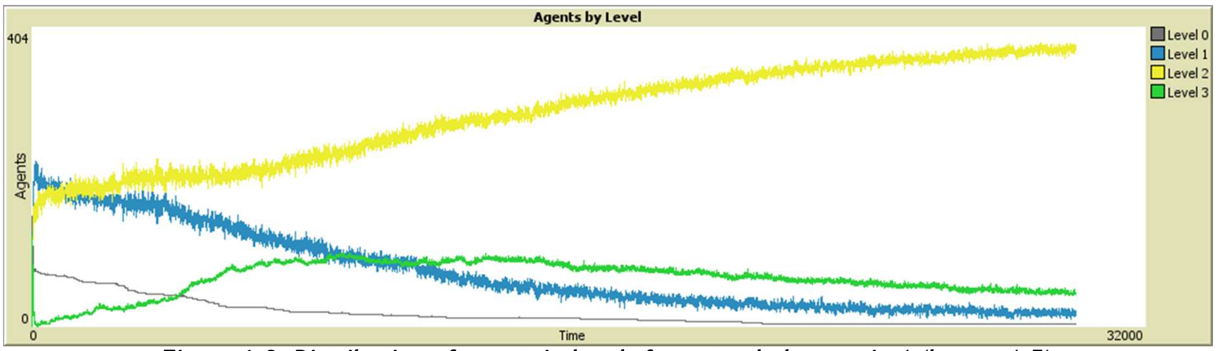

Figure 6.8. Distribution of agents in levels for extended scenario 1 (base x 1.5)

In Figure 6.8, the consequence of the change in the price slope is that the largest part (85%) of agents opted by act according to level 2 – i.e. reducing their consumption habits, while levels 0 and 1 have almost virtually disappeared, with 0.7% and 4.1% of agents each. However, the number of agents opting for level 3 still remains relatively low: only a 10.2% of agents. This cannot be regarded as unsatisfactory, since it corresponds to 45 households, a number which is actually higher than the total number of households in detached houses (40). The hard constraints set for the practise of home composting by agents living in apartments prevent this number of being significantly higher. Nevertheless, it was decided to check if a further increase in the price slope would force more households to reach level 3 – regardless of this option being perhaps somewhat unrealistic in practice. This is the result of Figure 6.9, with the slope increased by 2. In this case, level 2 still continues as the most common with 65% of agents, but a noticeable number of agents have now joined level 3, reaching 31%. Levels 0 and 1 remained in low numbers (1.4% and 3.2% respectively), as expected.

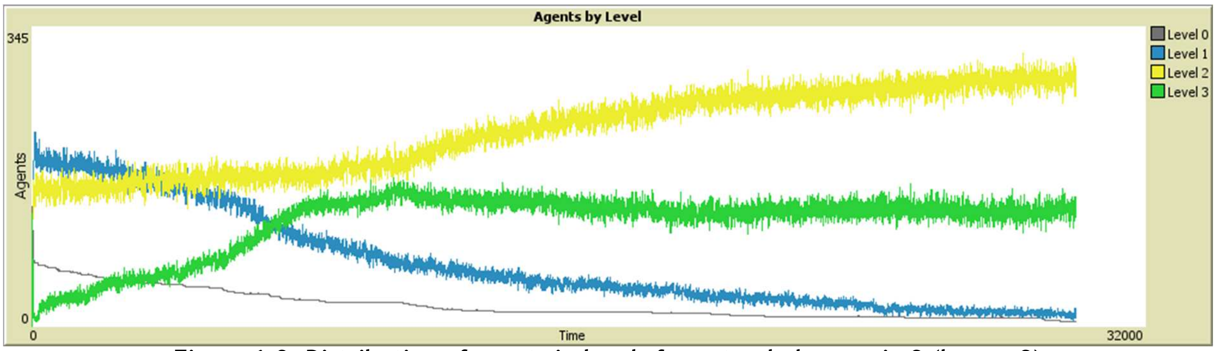

Figure 6.9. Distribution of agents in levels for extended scenario 2 (base x 2)

Basing on the assessment shown on Figure 6.6, now it is possible to translate the distributions obtained in Figures 6.8 and 6.9 into a respective environmental outcome which was added to the whole comparison in Figure 6.10.

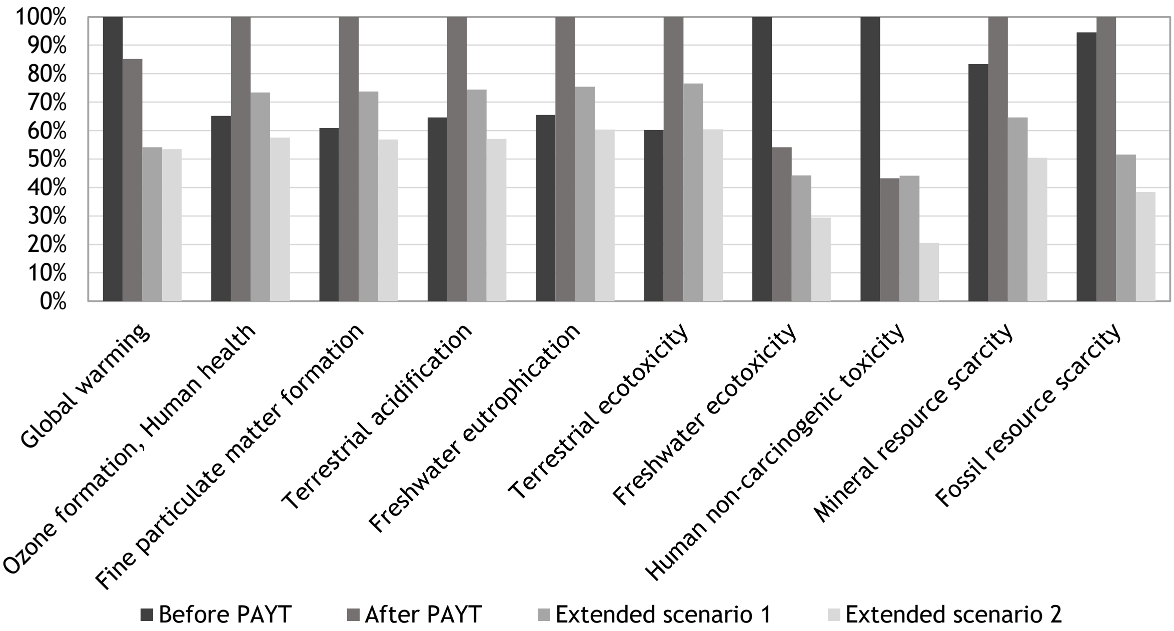

Figure 6.10. Relative comparison between C-LCA results before and after PAYT application and extended scenarios 1 and 2

The reduction in the categories related to toxicity is the most visible result in Figure 6.10, caused by the decreased in waste sent to landfill. The same effect is observed in the contribution to global warming, but the decrease is only relevant for extended scenario 1, being almost negligible when compared to extended scenario 2. The reason behind, is that the promoting of home composting diverts from mixed MSW – hence, from landfill – a considerable amount of biowaste which, otherwise, would have been in part landfilled and thus naturally degraded into biogas, suitable to be captured for generation of renewable energy. In other words: the less biowaste the landfilled waste material from refuse contains, the less is able to mitigate GHG emissions through the capture of landfill biogas.

In the other categories, the decrease in waste collected induced considerable reductions of the impacts. This is especially relevant in the two categories referred to consumption of resources (fossil and mineral); still, in the remaining categories, as long as the quantity of materials separately collected for recycling remained high compared to the initial BaU scenario, the efforts required for that separate collection and for the recycling processes themselves resulted in similar impacts as those obtained with BaU – specifically, somewhat higher in extended scenario 1 and somewhat lower in extended scenario 2.

6.5. CONCLUSIONS

The consequential model developed in this work with support of ABM was able to reasonably reproduce the results obtained in the PAYT pilot experience which took place in Aveiro. The model was useful to project hypothetical scenarios which represent possible suitable alternative implementation solutions for the operationalisation of a MSW pricing scheme based on variable charging. Further testing of the model with other similar experiences would be recommended for fully confirming its validity, but in the Aveiro case here addressed it has showed to work in satisfactory manner, despite some shortcomings found in the modelling process.

Firstly, the modelling of the landfill pollutant emissions was adapted in each scenario to a different composition of the landfilled refuse materials, according to the initial composition of MSW entering the MBT facility. However, uncertainty exist regarding the mass balances within the facility, since unfortunately, the responsible company does not make publicly available all the data regarding operational performance. This lack of information has more influence in the hypothetical scenarios, since there is no confirmation of how the internal processes in the facility would perform on such conditions – regarding efficiency of separation processes, biogas yield in the anaerobic digestors, etc.

Another source of uncertainty is found in the modelling of separate collection. As already mentioned in Chapter 5, mixed MSW is a much more optimised process than separate collection because it is favoured by scale economies, provided that the collected amount of mixed MSW is much larger than those of the different separate collection fractions. This disadvantage of separate collection is attested by the differences in fuel consumption required per tonne of collected material seen on Table 6.2. However, if more MSW are diverted to separate collection, it should be expected that these collection circuits would also be favoured by scale economies and, hence, more optimised in the future. But, as long as this remains only a hypothesis, the lack of reliable data difficults the modelling of that "improved" separate collection. Accordingly, the results presented in Figure 6.10 should be taken as conservative estimates.

It would be desirable for a future extension of this work, a testing with a larger sample – larger, at least, than one neighbourhood like that selected for this work – and also during a larger period – several years – in order to obtain a better assessment on long-term effects, based on extensive time series of data.

It has been demonstrated by the attitude taken by the residents that the monetary value assigned to the price is not the only driving factor governing the process of decision-taking – since the changes in behaviour took place actually before any change in tariff entered in force – but the concern regarding pricing serves as a trigger to activate the other driving forces present in the model. However, the discussion remains open regarding which of the acting driving forces is the most powerful compared to the other. The question is not easy to answer, since the decision taken is likely a combination of all factors involved, which are intrinsically mixed, and probably also not constant in time due to other changing circumstances; hence, a deeper sociological and psychological approach would be needed to address the question.

6.6. ACKNOWLEDGEMENTS

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7. Conclusions and future perspectives

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ABSTRACT

The work developed in this thesis has been able to show that the adoption of a variable PAYT/UBP pricing system for financing MSW management can result in significant behaviour modifications of the population involved regarding their practices of waste generation and management. The reason for this behaviour modifications do not depend only in the value of the price applied, but in the awareness and sensitivity towards the waste issue which were indirectly activated by the policy change. These modifications in behaviour consisted in adopting a more sustainable lifestyle which results in less waste being generated, as well as more materials being recycled or valorised. Besides the general reduction of environmental impacts due to the decrease of waste quantities produced, more particular impacts of MSW generation should be addressed, namely the impact of plastic bags for collection and also the possible ways for optimising the treatments operations. Notwithstanding, the most environmentally impacting activity of MSW management corresponded to the disposal in a landfill. This finding justified the inclusion of landfilled MSW as one of the key indicators for assessing the progress of a MSW management system towards sustainability, along with the other indicators proposed. Nevertheless, some uncertainties exist in the modelling processes which could be better handled in a future extension of this work, although they do not compromise the global trends and findings observed.

7.1. OVERVIEW

The main question to which this PhD. thesis has tried to give an answer consisted in gaining knowledge about the environmental impacts which can be expected from the application of a variable charging scheme (PAYT/UBP) as a pricing policy to be paid by citizens for financing MSW management. At the current moment, it is foreseeable that these kind of systems will experience a great development and expansion in southern European countries like Portugal, where they did not a significant presence until now. Many municipalities are facing the challenge of designing and implementing waste policies which allow them to fulfil the recycling and sustainability goals set by the national governments and EU for the next years, and the adoption of variable pricing schemes is increasingly regarded as a useful tool to become a decisive part of that effort; see for instance the report about waste management in the EU capital cities made by BiPRO GmbH & Copenhagen Resource Institute (2015), which placed the cities with PAYT schemes among the best performers. Therefore, a clear understanding of what PAYT pricing can – and what cannot – achieve will provide a better guidance for granting the success of an adequate application of this policy.

To achieve this objective, a pilot experience for testing a PAYT system developed in a particular neighbourhood of the Portuguese city of Aveiro under the scope of a LIFE project (LIFE PAYT), has been used as a benchmark for determining how the different MSW fractions changed regarding its composition and quantity, as a consequence of the reaction of the population to the new system implemented. The field data obtained along the experience were later analysed, along with other complementary data concerning the general performance and characteristics of MSW management facilities in the region, by making use of the following modelling and analysis methodologies and tools:

- Attributional life cycle assessment (LCA): applied to obtain an initial assessment of waste management in the pilot area, prior to PAYT implementation.
- Agent based modelling (ABM): used as a modelling tool to reproduce the behaviour of the population distributed in households (represented by the agents).
- Consequential life cycle assessment (C-LCA): applied to extend the environmental assessment including the comparison between the initial and the final moment, linking it to the scenarios generated by the ABM.

The whole array of tasks performed has made possible to obtain a complete and comprehensive assessment of the environmental outcome derived from the experience.

Firstly, the work addressed on Chapter 3 within Section I, focused on the definition of a suitable set of indicators for evaluating the progress made in relation to a theoretical sustainability horizon, allowed a better understanding of the key aspects of a waste management system which should be monitored, in order to establish whether the actions that have been put in practice can be regarded as satisfactory or not, to achieve that ultimate sustainability goal.

Nextly, the work performed in Section II with attributional LCA applied to the case study showed the most relevant environmental impacts caused by municipal waste management: in Chapter 4 the assessment was focused on municipal waste collection, highlighting the impact of carrier plastic bags and fuel consumption, while in Chapter 5 the overall assessment of the whole waste management system shifted the focus to the consequences derived from treatment, remarkably the high influence of the environmental impacts attributed to the practice of landfilling. Nevertheless, this chapter explored also the feasibility of turning waste management a carbon neutral activity, taking as base this particular case located in Aveiro. The findings obtained can be regarded as an advancement of the potential improvements in the environmental footprint of waste management which might be enabled by a successful and generalised application of policy strategies like PAYT/UBP.

Finally, the work performed within Chapter 6 in Section III represented the closing of the assessment, basing on the results previously obtained to characterise the consequences of the waste tariff change. Once the default scenario was fully defined – as that corresponding to Section II –, it served as a basis upon which the dynamic the model developed in Chapter 6 was tested, to simulate the reaction of the population. The model was able to reproduce the behaviour change which was observed along the pilot experience, and therefore enabled the establishment of projections of future hypothetical evolution of the system analysed.

A more detailed exposition on the specific findings and conclusions related to each of the sections is developed in the following sections.

7.2. SECTION I: DEFINITION OF INDICATORS

As stated in the Introduction, this part of the work was necessary to elucidate which are the most critical aspects to monitor in a waste management system at municipal level with reference to a sustainability paradigm – a horizon or an ideal situation towards which each municipality should be constantly progressing, not only in the environmental dimension but also in the economic and social. It is, however, clear that the environmental component adopted a more protagonist role in accordance with the framework of this thesis, focused mainly on environmental assessment. The two indicators selected to represent the environmental component – MSW generated and MSW landfilled – turned out to be the most influencing variables for the environmental assessments performed on the later chapters, thus confirming the importance of both metrics. This seemed obvious in the case of MSW generated, since this is the reason for waste management to exist, but for MSW landfilled is a more particular feature of the context here addressed: landfilling is, in Portugal and in other southern European countries, still the most common form of waste treatment and, as such, the main source for the environmental impacts caused by this activity and the main barrier to be overcome for advancing towards sustainability. Monitoring the progress in landfilling reduction corresponds to tracking the fastest way to reaching environmental sustainability for municipalities like the one here analysed. Hence, the pertinence of the indicator was later justified and confirmed by the findings of the LCA.

Moving to the other two components – economic and social –, it can be concluded that the framework proposed is mostly in accordance with the correspondent effects expected from PAYT/UBP. In the economic aspect, it should be remarked that the implementation of a PAYT system usually requires, at least in the initial stage, significant investments in new or adapted equipment and a higher effort in collection, but conversely the operating costs should be reduced with time as long as the collected quantity of waste decreases, an aspect also highlighted by the analysis of costs performed in Chapter 5. A close monitoring of costs is, therefore, recommended. Moreover, the response of citizens towards the new system will indeed affect the revenues obtained: they would be lower as long as less waste is generated. Regarding this issue, the simulations conducted in Chapter 6 as a forecast of population behaviour could be usefully linked with the sustainability indicator related to financial balance between costs and revenues.

With respect to the social component, the relevance of the two indicators proposed corresponded with trends and events which were actually observed during the pilot experience in Aveiro. On the positive side, the number of bring banks placed in the streets of the neighbourhood for separate collection of recyclable materials was doubled; this increase encouraged and facilitated the effort made by citizens to separate those materials at home, with the subsequent environmental effects perceived in the LCA. This fact attested the importance of an indicator for evaluating the accessibility to separate collection, as the one proposed in Chapter 3. On the negative side, the other social indicator proposed, referred to the number of registered complaints, proved also its importance: the implementation of the new system was not free from technical troubles and malfunctions, a situation reflected in the number of complaints laid by disappointed citizens. Monitoring the number and evolution of these complaints is hence an appropriate manner for assessing the acceptability and satisfaction of the users with the PAYT scheme.

As a final remark, it should be however stressed, that the framework of indicators proposed makes the most sense in the scope of the context found in Portugal, but may be limited beyond that geographic scope. The necessary information required for calculating the indicator values is entirely included as part of the annual reports on waste management performance which municipalities must publicly submit to the regulatory authority – ERSAR, the waste and water services governmental regulatory entity[6]. Consequently, the author is confident that the framework is feasible to be determined for every Portuguese municipality, but it might not be the case when applying it to other countries where the publicly availability of these data is not compulsory for municipalities, or even where a regulatory legal body does not exist in the same or approximate form.

7.3. SECTION II: ATTRIBUTIONAL ASSESSMENT

The works performed within Section II provided a full and comprehensive view of all the environmental consequences derived from a conventional MSW management system – comprising collection/transport (focus of Chapter 4) and treatment/disposal operations (focus of Chapter 5, but integrating also the results of Chapter 4 in a global assessment) –, predominantly based on mechanical-biological treatment, recycling and landfilling as final disposal destination.

7.3.1. Influence of carrier bags in the impacts of MSW collection

The main finding in Chapter 4 pointed to the highly relevant contribution of plastic carrier bags commonly used as a recipient for keeping and transporting household waste to the environmental impact attributed to the MSW collection process. It is remarkable that the impact of bags exceeded that of the fuel used by collection vehicles – the other top contributor to collection impacts – in the category referred to fossil resources depletion, where the oil-derived fuel has in principle its most direct impact. So, it can be concluded from this result that, while the use of fuel for MSW collection has been optimised to a considerable extent – i.e. bringing the amount of fuel consumed per mass of MSW collected to a minimum –, it does not happen the same with carrier bags.

The conclusion makes sense, as long as fuel consumption is a cost supported by the operator of collection system, and given the influence of fuel on the costs structure of MSW collection (Sousa et al., 2018), it is obviously of its main interest to keep this expense at a minimum. However, a similar reasoning is hardly applicable to carrier bags, which are individually acquired at low cost by every user of the waste collection system, and probably not

[6] https://www.ersar.pt/pt/publicacoes/relatorio-anual-do-setor

always employed in an optimised way, neither choosing the optimal bag size; that is: the users do not wait until completely filling the bag with rubbish, but discard it due to hygienic concerns when it is considered that enough time has passed before degradation of biowaste begins to be perceived. This practice may be partially related to MSW collection frequency, as mentioned on the results section of Chapter 4. Hence, waste carrier bags can be assumed as a consumable good with a short lifetime and only one use, and that is the explanation for its relative high impact, compared to other items of MSW collection.

As discussed in the same chapter, it could be arguably objected, though, that the previous reasoning is only applicable to throwaway bags purposely made for carrying waste, but not for reused bags. In the latter situation, a strictly attributional point of view for the LCA would require to split the environmental impacts by using some rule to allocate them to each of the uses the bag had. The approach has interest, taking into account that the implementation in Portugal of the tax penalising the purchase of lightweight plastic bags at stores has resulted in the spreading of reuse practices (Martinho et al., 2017). To some extent, this reaction may be regarded as a parallel behaviour to the one observed with PAYT application: the taxed price of bags is not by itself a hard barrier preventing its use – since the value is anyway low –, but it acts as a trigger of personal environmental awareness. This reuse trend, along with the enhancement of plastic film recycling, are regarded as the most current effective ways to prevent the environmental impacts of plastic bags, given that substitute bags based on other alternative materials such as biopolymers, do not constitute yet a more environmentally friendly option, as demonstrated by the LCA performed in this work.

7.3.2. Optimisation of MSW treatment

In Chapter 5, the main conclusion extracted from the work to the feasibility of greatly reducing the carbon footprint of MSW management – and, subsequently, other environmental impacts – and even, under some particular circumstances, turning it into a carbon neutral system. It is clear that the initial assessment of the base scenario showed a picture rather far from neutrality in carbon footprint, as well as in the other impact categories with the exception of the contribution to fossil and mineral resources scarcity, in which the *environmental credit* obtained by the recovery of materials for recycling allowed considerable savings of raw materials. But, the analysis performed was able to demonstrate that, with some strategic modifications in the distribution of waste flows collected and in the operating conditions of the treatment facility, then the outputs obtained from MSW treatment would be altered to the extent that the overall environmental impacts assessment would be substantially improved.

It is noteworthy to remark that, since this chapter is still dealing with the initial baseline assessment of the case study without any change having been yet introduced in the pricing system, the modifications experimented did not imply any change on the overall quantity and composition of the MSW generated in the neighbourhood – i.e. the functional unit of the LCA remained always the same – only its distribution between the material flows of mixed MSW and separate collection, as well as between the internal treatment processes, were modified. Nonetheless, an important question appeared in this point: would it be possible that these changes took really place without an external boost such as that represented by a change in pricing towards PAYT/UBP? Probably not, and here lies the main interest of the work presented: to unveil the potential of such a powerful tool to reverse the environmental outcome of MSW management. This is noteworthy, for instance, when addressing the issue of anaerobic digestion performance – in other words: biogas yield – within mechanical-biological treatment. To the knowledge of the author, it has been a recurrent complaint of MBT operators the low "quality" of the biowaste sent to the biological treatment stage as part of mixed MSW. The high amount of materials other than biowaste hinders their complete separation in the mechanical stage and thereafter negatively affects the performance and even the stability of the methanisation reaction in the anaerobic digesters. Besides the separate collection of biowaste, still pending to be massively introduced in Portugal, any additional diversion of materials from mixed MSW towards separate collection flows – like those promoted by PAYT – would result in performance improvements. Still, it should be said that it is hard to model the exact magnitude of the improvement due to an incomplete knowledge of the operating parameters of the MBT process here analysed – because the process operator did not make them available –, but the improvement is anyway realistic and was thus included in the assessment. Not so directly related with PAYT is the valorisation of the remaining refused waste as RDF, since it depends of external market circumstances, but it is nevertheless a desirable improvement of the system, and as such was also included.

Regarding the costs assessment which was also addressed, it was found that, not surprisingly, is directly related to the value of the outputs obtained from MSW management. As long as more quantity of the outputs is produced and sold – compost, electricity, sorted recyclable materials, RDF – the revenues obtained contribute to compensate the costs required by MSW management, which also increased globally, but a lower rate than revenues. Neither a balanced situation between both costs and revenues was ever reached in the scenarios proposed, nor was this a goal to be met, since it is not the function of MSW management to obtain a profit, but to fulfil a public necessity of society.

Other than the shortcoming already indicated in Chapter 5 regarding the modelling of separate collection costs, which cannot be adequately projected in the hypothetical scenarios due to lack of the data that would be required to build a more complex and specific cost model for that activity, the assessment did not include neither the investments that would be required for establishing a PAYT scheme, nor the expected evolution of revenues from MSW tariff, being thus limited only to the operating costs of the system. A more extensive economic study which included also these aspects was beyond the scope of the work here presented, but it would undoubtedly constitute an interesting continuation for the future.

7.4. SECTION III: CONSEQUENTIAL ASSESSMENT

In contrast to Section II, where an attributional approach was chosen for the environmental assessment of the scenario initially found in the area which is the object of study, in Section III it was decided that a consequential perspective would be more adequate for the modelling of the effects derived from the introduction of a new variable pricing system. The decision is fully justified because the process assessed fits into the application logic of consequential LCA: an initial stable situation which is affected by a punctual change, forcing the whole system to a reconfiguration in order to achieve stability again. In parallel, the advantage taken from application of Agent Based Modelling allowed to build, with a reasonable effort and a not too complex dataset, an operational simulation of the behaviour of the population involved in the PAYT experiment. Despite the necessary simplifications which are inevitable in a model which tries to reproduce the always more complex reality, the model built in Chapter 7 managed to reproduce, with reasonable accuracy, the adaptation to the new situation experienced by the agents – in this case: the participant households.

Even though the existence of undesired behaviours deviated from the norm – the most likely violation consisted in transferring the own waste to areas outside that of the pilot experience – cannot be excluded, it was attested by the experience that a large part of residents actively collaborated in the experience by regularly using the electronic containers connected to the new pricing system. The answers collected in the surveys made to the population also reflected a positive opinion regarding the PAYT concept – perceived as a fair way to pay for MSW management –, despite some problems and constraints detected during the implementation. It was demonstrated that changing the pricing scheme to put in place a variable charge induced a significant reaction of the residents, confirmed not only by the increase observed in separate collection – facilitated by the installation of new bring banks in the street – but resulting also in a net reduction of the overall MSW generation in the neighbourhood.

The reduction trend was actually observed even before the actual change in the tariff taking place. This finding confirmed the hypothesis proposed in the Introduction of this thesis, that it is not the monetary value by itself the variable which exerts the main pressure for changing behaviour, but the awareness that the value will be effectively paid and that the user has the power to modify it – to make it become lower – by means of her/his actions. It would be rather tempting to undertake an attempt to establish a kind of weighting between the three driving forces defined in Chapter 6 – civic/environmental awareness, price concern and social pressure –, in other words: to find how powerful is each force in comparison with the other. However, such an effort seems futile: what in ABM is modelled as a sum of rational decisions by the agents, may be more intertangled in reality, with more particular factors intervening and influencing each other (Heller and Vatn, 2017). This does not contradict the reliability of the model as a whole – as long as it was based on a real experience –, only makes difficult to isolatedly assess the role of each driving force. For instance, it is clear in the structure of the model that while the reactive response to environmental and price concerns was fast in the simulations, quickly placing each agent on a level after a few iterations, the effect of social pressure was felt during a much longer time, slowly vanishing as long as less agents were left available to be persuaded. Although the loss of engagement with time is consistent with other reported experiences (De Jaeger and Eyckmans, 2015), it can be arguably discussed if this lag in time between different driving forces is or not realistic. But nevertheless, the final result would have been ultimately always the same, no matter how long does it take to attain it.

Regarding home-composting, as the option more environmentally-friendly for biowaste household waste management, a slight increase in practitioners was observed during the pilot experience, referred in Chapter 6, but they remained anyway a minority within the whole population. According to the model developed, a more exigent increase in the waste tariff price will persuade more households to join this practice, but unfortunately neither the scope and length of the experience developed in this project did not allow to further explore this hypothesis, nor many other studies have been found to serve as precedents – see for instance Wu et al. (2019). In practice, for an expansion of this activity, a strong supporting effort by the municipality is recommended; for instance: providing equipment (home composters), offering assistance and training for interested citizens, etc. Community composting in public gardens or parks is also a good alternative for highly populated areas of apartment buildings, where individual composting at home is not easy to implement.

Furthermore, it has been found that the reduction in environmental impacts obtained from a decrease in waste generation was not so remarkable when the prevented waste input corresponds to biowaste. This consequence is explained by the fact that, even though biowaste is directly responsible for impacts like GHG emissions, it also serves as feed for energy recovery processes through its valorisation as biogas produced naturally in landfills, and industrially in anaerobic digestion reactors. If the biowaste input to these processes is diminished, the environmental credit obtained by replacing non-renewable energy sources becomes also smaller. Notwithstanding, the uncertainty in this aspect is significant, due to the difficulties for modelling the environmental impacts caused by landfilled waste in hypothetical scenarios: although the composition of landfilled residual material was adapted to each situation, it is always uncertain how the mass balances and the general operating conditions within the MBT facility would be altered in case of those scenarios taking place in the future. This gap, added to the already mentioned uncertainty in the modelling of hypothetically more generalised separate collection, contribute to some loss of robustness of the results – which maybe could have been mitigated with a geographically broader with a larger population sample and a longer in time field experience –; but, in a general sense, the overall result can be regarded as clarifying.

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Appendix

Index of Figures

Index of Tables

List of publications included

- Chapter 3

Title	Are municipal waste utilities becoming sustainable? A framework to assess and communicate progresses
Authors	1. Álvaro Fernández Braña (Universidade de Santiago de Compostela) Contributions: conceptualisation, data collection, performing calculations and manuscript writing.
	2. Vítor Faria e Sousa (Instituto Superior Técnico de Lisboa) Contribution: conceptualisation and revision.
	3. Célia Dias Ferreira (Instituto Politécnico de Coimbra / Universidade de Aveiro) Contribution: conceptualisation, revision and supervision.
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Impact Factor (JCR)	3.056 (2019)
Quality Index (JCR)	Q2: 62.83% percentile in category Environmental Sciences (2019)
DOI	10.1007/s11356-019-05102-4

- Chapters 2 and 4

Title	Looking beyond the banning of lightweight bags: analysing the role of plastic (and fuel) impacts in waste collection at a Portuguese city
Authors	1. Álvaro Fernández Braña (Universidade de Santiago de Compostela) Contribution: conceptualisation, data collection, performing calculations, interpretation of results and manuscript writing.
	2. Gumersindo Feijoo Costa (Universidade de Santiago de Compostela) Contribution: revision and supervision.
	3. Célia Dias Ferreira (Instituto Politécnico de Coimbra / Universidade de Aveiro) Contribution: revision and supervision.
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DOI	10.1007/s11356-019-05938-w

➢ Chapter 5

Title	Turning waste management into a carbon neutral activity: Practical demonstration in a medium-sized European city
Authors	1. Álvaro Fernández Braña (Universidade de Santiago de Compostela) Contribution: conceptualisation, data collection, performing calculations, interpretation of results and manuscript writing.
	2. Gumersindo Feijoo Costa (Universidade de Santiago de Compostela) Contribution: revision and supervision.
	3. Célia Dias Ferreira (Instituto Politécnico de Coimbra / Universidade Aberta) Contribution: revision and supervision.
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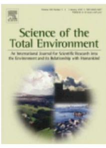

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This PhD. thesis has as main objective the evaluation of the environmental impacts derived from the application of unit base (variable) pricing systems for financing urban waste management. To this purpose, a pilot experience with variable pricing schemes which took place in the Portuguese city of Aveiro was chosen as case of study. The evaluation was mainly performed making use of Life Cycle Assessment (LCA) as an analytical tool for measuring environmental impacts. Both modelling approaches of LCA (attributional and consequential) were used in order to perform a comprehensive environmental comparison between the situation found in waste management before and after the pilot experience.