UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

Departamento de Enxeñaría Química

Development and integration of environmental evaluation tools for the ecodesign of sustainable processes and products

Memoria presentada por:

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Para optar ao grao de Doutora pola Universidade de Santiago de Compostela.

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Que a presente memoria titulada "Development and integration of environmental evaluation tools for the ecodesign of sustainable processes and products", presentada por Marta Herva Iglesias para optar ao grao de Doutora pola Universidade de Santiago de Compostela, ten sido realizada baixo a nosa dirección no Departamento de Enxeñaría Química da Universidade de Santiago de Compostela.

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Os directores, A doutoranda,

Enrique Roca Bordello Amaya Franco Uría Marta Herva Iglesias

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Summary

Industry is recognized as one of the main sources of environmental pollution and resource depletion, both causing environmental degradation; nonetheless, its contribution to development and wealth creation is also acknowledged. Therefore, the identification of sustainable options in this area is a key factor. Nowadays, the attitude towards pollution prevention and control and cleaner production is not just a response to emerging environmental laws and regulations (Registration, Evaluation, Authorization and Restriction of Chemicals -REACH-, Integrated Pollution Prevention and Control –IPPC- Law, Integrated Product Policy –IPP-), but also a matter of corporate responsibility. Further, it has proved to be a way to increase profits. The sustainability definition has received certain criticism for its vagueness, ambiguity and difficulty to translate this concept at different levels. To overcome the difficulties of its implementation, a wide variety of indicators have been developed and applied over the years, providing metrics essential at the action level.

This thesis poses a contribution to the development of environmental evaluation tools adapted to particular production sectors, aiming at providing metrics to guide decision making for the ecodesign of sustainable processes and products. Integrative frameworks that combine methodologies of different nature were proposed as the most suitable way to achieve comprehensive evaluations. At the same time, the simplicity of tools was pursued to make its application easier and more attractive for enterprises, avoiding the need of in depth training.

Chapter 1 presents a review of the state of the art of indicators concerning environmental issues, under a process- and product- oriented approach. Indicators of different nature were reviewed, from those with a territorial dimension (Ecological Footprint, Environmental Space, Dissipation Area Index) to the more generic material and energy flows (Energy, Exergy, Emergy, Water Footprint, Rucksacks), life-cycle (Carbon Footprint) or environmental risk indicators (Hazard Quotient, Cancer Risk). Their usefulness, drawbacks and applications were discussed. It was observed that the different indicators provided complementary information about the environmental performance of processes and products; thus, if used together, a more comprehensive evaluation could be obtained, overcoming the weaknesses of single indicators. As a consequence, the combination of complementary indicators was found

particularly interesting and beneficial. The advantages of integrative frameworks and the effectiveness of different multi-criteria analysis methodologies to obtain a fair and simplified final result of the sustainability appraisals were also reviewed.

The usefulness of Multi-criteria Analysis (MCA) methodologies to aid decision making by handling at a time indicators from the different dimensions of sustainability was also discussed in Chapter 1. The ranking or the single comparable index obtained when different indicators are combined helps to better interpret the results and to simplify the evaluation problem; albeit, uncertainty associated to imprecision of data, weighting schemes or aggregation methods must be taken into account. The most relevant MCA methodologies for the environmental field were also presented: 1) Multi-attribute utility theory: AHP, MACBETH; 2) Outranking methods: ELECTREE, PROMETHEE, NAIADE; 3) Fuzzy Logic based MCA. Further, applications found in the literature were summarized.

Chapter 2 provides a general description of the materials and methods employed during the development of this thesis. The chapter was divided into three main sections. The first one included the methodologies regarding the environmental evaluation of products and processes, namely Ecological Footprint (EF), Life Cycle Assessment (LCA), Environmental Risk Assessment (ERA) and Energy and Material Flow Analysis (EMFA). The second part of the chapter dealt with the multi-criteria analysis, describing those tools that were selected to be applied in the different chapters: Analytic Hierarchy Process (AHP), Fuzzy Logic techniques, ELECTRE III and PROMETHEE/GAIA. The third section collected those statistical analysis tools that were applied to support, complement and give consistency to the different estimates conducted, namely sensitivity analysis and statistical correlations.

The materials mostly referred to software tools and database sources. A number of software tools were studied and applied during the development of the thesis, all of them linked to a specific methodology from the three sections established in this chapter. Hence, SIMAPRO® was employed to conduct LCA studies, Umberto® for EMFA, Matlab® and specifically the Fuzzy Logic Toolbox for multi-criteria analysis, Decision Lab® for PROMETHEE/GAIA, Crystal Ball® for sensitivity analysis and MS Excel® and SPSS® for general statistical analyses. MS Excel® was also used to implement the simplified tools derived from the adaptation of EF, LCA and ERA to the production processes studied. Regarding databases, several sources were consulted to collect all the parameters necessary to conduct the analysis proposed. These were more specifically detailed in the chapters where they were used.

The work developed in **Chapter 3** constituted a pioneering application of the EF to a production process. Until then, the EF had mostly been used as an indicator of environmental sustainability applied to individual lifestyles, regions, nations or even the world. Although its application to businesses and industry had been suggested, scarce initiatives existed in this field. In the study presented here, a textile tailoring plant was analyzed. The textile sector is situated among those more outstanding in the Spanish industrial structure, specially the tailoring sub-sector. In Galicia (NW Spain), the fashion industry has acquired especial importance in the last years due to the presence of several designers of national and international renown. This caused a strong development of this industry, which generated a great impact in the Galician economy. As a consequence, it was particularly interesting to develop tools to properly assess the environmental burdens associated to this sector.

The overall purpose of Chapter 3 was to develop a tool useful for evaluating the environmental impact evolution due to the performance of the plant, as well as for comparing the environmental behavior of different tailoring processes. Therefore, the EF methodology was adapted to the case study and the selected data were those from the manufacturing process. Data were divided into three main categories: energy, resources and waste. The principal contribution to the final EF (expressed in hectares of land) was the resources category, mainly due to the high value associated to the cloth. The consumed energy was the second contributor, while the waste category remained in third place. The final outcomes were divided by the production rates to obtain a comparable relative index, easy to be interpreted by the different stakeholders. This was of special importance for a company involved in Corporate Social Responsibility (CSR) meant to have a general communication strategy.

Although the EF has grown in interest and popularity over the years, it has also been the subject of criticism and controversy. The advantages of an aggregated indicator are often overshadowed by the shortcomings of its corresponding methodology. Hence, the application of the EF to a production process in Chapter 3 revealed that a complete measure of the environmental impact of the activity could not be provided. As a response, later research was focused on finding solutions to overcome the major critiques of the indicator, which prevented from considering the EF as an appropriate and useful indicator to be used at process or industrial level. One of the flaws of the EF methodology more frequently treated in the literature is the fact that it does not account for toxic or hazardous pollutants and wastes, which cannot be part of a closed biological cycle. This poses a major problem when evaluating the environmental burdens of an industrial process, where these kinds of flows are expected to be generated.

In this respect, the methodology developed in **Chapter 4** estimates the EF of toxic and hazardous wastes considering a closed cycle modeled through a plasma process; a phenomenon that naturally occurs in stars and volcanoes. A simultaneous dual reaction process takes place in a plasma reactor: the organic compounds are thermally decomposed into their constituent elements (syngas with more complete and advantageous conversion of carbon into gas than in incinerators), while the inorganic materials are melted and converted into a dense, inert and nonleachable vitrified slag, which does not require controlled disposal. Therefore, it can be viewed as a totally closed treatment system. Syngas (mainly composed of CO and H_2) can be used to generate electrical power. Vitrification is the result of the interaction between the plasma and inorganic materials. Because the inert fraction is vitrified and harmful substances can barely leach from the lava, this product can be used for road construction or as a building material.

Wastes from industry can be treated in a thermal plasma gasification process and, by developing a methodology based on a model to describe this process, the EF of hazardous wastes was estimated. Nonetheless, this does not mean that plasma treatment was considered as a panacea to deal with waste management problems; however, it was employed as a closed cycle model existing in nature for methodological purposes. The developed tool was tested with the case study presented in Chapter 3, both for hazardous and non-hazardous wastes generated in the tailoring factory. For the latter a value of 56.5 gha was obtained, a figure on the same order of magnitude as that obtained in Chapter 3 where the conventional EF methodology was applied.

The methodology presented in Chapter 4 was focused on solid waste flows; however, there are other waste flows that can be stem from a production process, e.g., emissions to air and water. **Chapter 5** deals with the former, since the EF is often criticized for not including emissions other than CO_2. In this respect, other greenhouse gases (GHG) were considered and the effect of incorporating acidifying emissions by considering a critical load was assessed. A ceramic industry (manufacture of bricks in baked clay) typical from Galicia was selected as study case to analyze the impact of emissions derived from the burning of fuel oil or natural gas during the drying and firing stages, as well as their influence in the EF figure.

Another controversial aspect is the use of global or local factors in EF assessments, leading to a discussion on the priority of assuring comparability among studies from all over the world or accuracy on estimates by adjusting to regional conditions. In Chapter 5, a specific CO_2 absorption rate was appraised for Galicia (NW Spain) on the basis of the capacity of the forests to act as a carbon pool. Two different methodologies to assess biomass were applied, using data from the second and third Spanish forest inventories: a) Biomass Expansion Factors (BEF); b) Allometric equations. The main species present in the Galician forests were considered: *Pinus pinaster*, *Eucalyptus globulus*, *Quercus pyrenaica*, *Quercus robur*, *Pinus radiata*, *Castanea sativa* and *Pinus sylvestris*. The results were: a) 3.83 t CO_2 ha^{-1} yr^{-1} for BEF and b) 4.33 t CO_2 ha^{-1} yr^{-1} for allometric equations. The former is in quite good agreement with the world-average one (3.67 t CO_2 ha^{-1} yr^{-1}) used in Living Planet Reports, but the latter was 18% higher. Allometric equations can be considered as a more accurate method, as the different parts of the tree are assessed independently for each species. Besides, for the majority of the species, site specific correlations for the area were available. A sustainable management of forests could lead to maintain a good carbon stock capacity in Galicia, thus contributing to mitigate climate change. This would be reflected in a reduction of EF values.

The tailoring factory analyzed in Chapter 3 was taken up in **Chapter 6**. To extend and make the evaluation more comprehensive, the new methodological proposals from Chapters 4 and 5 were incorporated. Further, apart from the EF, other environmental evaluation methodologies, namely EMFA and LCA, were applied to provide a complementary perspective. The dressmaking process was modeled using Umberto® 5.5 and an analysis was conducted on the basis of material and energy flows. The updated results for the EF were presented and air emissions initially excluded from the EF assessment (Chapter 3) were evaluated under two approaches: 1) using characterized categories from LCA as complementary environmental indicators; 2) incorporating emissions into the EF by means of absorption factors including or not weighting factors, as discussed in Chapter 5.

Finally, sensitivity analyses based on Monte Carlo simulations were carried out to assess the influence of variability in the input variables.

The higher energy consumption in the production process occurred during the cutting stage, where minimization strategies had already been implemented. General activities like lighting and heating of the factory represented the second major energy consumption. Hence, actions like installing low power consumption lights, constraining the lighting to the areas that strictly need it or regulating the use of heating would lead to a significant decrease in energy consumption. Further, the mass balance conveyed that gasoil, in spite of its low contribution to energy flow, was one of the main sources of pollution of the factory. As a consequence, it was recommended its substitution for cleaner sources of energy. In addition to these aspects strictly related to the performance of the factory (e.g. energy consumption patterns already discussed), the management policies were strongly related to the incorporation of environmental criteria in the design of products. Therefore, the materials consumption should be minimized and the selection of fabrics should be based on environmental aspects rather than on tendency patterns.

After dealing with the environmental evaluation of production processes in previous chapters, **Chapter 7** focused on the ecodesign of products. Ecodesign may be defined as the systematic introduction of environmental concerns during product design and development. This means to bear in mind the environmental impacts at all stages of the product life cycle, starting at the design and development phases. The objective is to create sustainable solutions that satisfy human needs and desires, pursuing a compromise solution among environmental, technical, functional, ergonomic, aesthetic or economic criteria.

When more than one indicator is handled at a time, the difficulty arises when a decision has to be made based on the information provided by all of them. As previously introduced, MCA methodologies have proved to be efficient in the definition of integrative frameworks. In this case, Fuzzy Logic (FL) techniques, commonly used to address uncertainty matters, were selected. The use of FL techniques allows obtaining a quantitative approach using a qualitative representation; thus, it is able to simultaneously handle numerical data and linguistic knowledge. In this sense, an ecodesign tool integrating the criteria provided by three environmental evaluation methodologies, namely EF, LCA and ERA, was developed on the basis of FL reasoning and features. This idea enabled the decision making at process and product level taking into account the values of the different indicators at a time. The relative importance of each of them was established through the definition of membership functions as inputs to the fuzzy inference reasoning in the case of a specific product. As a result, a Fuzzy EcoDesign Index (FEcoDI) was obtained.

A well-known case study was used to support the development of the tool and to test it. In this respect, different packaging materials for a beverage bottle were considered to identify the most environmentally friendly option. After refinement on the basis of the feedback from this first case study and following the same protocol and features, the tool was enhanced and further developed to be applied in the ecodesign of footwear. Four models of children shoes were analyzed and compared according to the FEcoDI obtained. The tool properly identified those proposals of design that should be rejected (mainly because of the likely damage to human health during use) and provided a ranking based on their more or less suitability from an environmental and safety point of view. This information was considered by the design team to incorporate the environmental perspective in their decisions.

Finally, **Chapter 8** addressed the problem of Municipal Solid Waste (MSW) management. One of the major challenges for municipalities in the 21st century is to collect, recycle, treat and dispose of these increasing quantities of solid waste. Waste causes a number of impacts on the environment, including pollution of air, surface and groundwater; meanwhile, valuable space is taken up by landfills and poor waste management causes risks to public health. The waste hierarchy defined in the Directive 2008/98/EC on waste establishes the following priority order to be considered in waste prevention and management legislation and policy: 1) prevention; 2) preparing for re-use; 3) recycling; 4) other recovery, e.g. energy recovery; and 4) disposal. Nonetheless, this new Directive also addresses the possibility of altering the stated hierarchy in a specific situation, if justified by a life-cycle thinking study.

Therefore, the choice of a MSW treatment option is a complex process in which a widespread set of criteria must be taken into account. Additionally to economic, geographical situation or social aspects, the decision process should consider the environmental perspective. With the purpose of quantifying these environmental burdens, a wide variety of environmental and sustainability indicators have been developed in the last years. However, integrative frameworks have been signaled as the best option to achieve more comprehensive assessments.

In the first part of this chapter, a case study extracted from the literature was analyzed to prioritize among four different options of Municipal Solid Waste (MSW) treatment processes: landfilling with energy recovery; incineration with energy recovery; biological treatment of the organic fraction of MSW (OFMSW) with energy recovery from refuse derived fuel (RDF); thermal plasma gasification. In a first approach, the EF was applied as single indicator; latter, more indicators were included and a ranking of alternatives was established using MCA methodologies (AHP, ELECTREE family and PROMETHEE family). The ranking was (from best to worst): 1) thermal plasma gasification, 2) biological treatment of organic fraction with energy recovery from refuse derived fuel, 3) incineration with energy recovery and 4) landfilling. The results were compared to the commonly recommended hierarchy for waste management, conveying a good agreement. Also, in this case, the EF proved to be an efficient screening indicator to aid decision making.

In the second part of the chapter, the real case of LIPOR - Intermunicipal Waste Management of Greater Porto (Portugal) – was assessed. A joint application of EF and LCA was proposed with this purpose. LIPOR is the entity in charge for the management, recovery and treatment of the MSW produced by the eight partner municipalities: Espinho, Gondomar, Maia, Matosinhos, Porto, Póvoa de Varzim, Valongo and Vila do Conde. The main activities of the integrated management system of LIPOR, namely multi-material valorization, organic valorization, energy valorization and landfilling, were analyzed; consequently, stages that presented a major contribution to environmental burdens could be identified. The energy recovery plant was evaluated as the most environmentally friendly treatment plant from the EF approach. However, other environmental burdens as water consumption and environmental impacts associated to air emissions signaled this treatment process as the most pollutant. On the other hand, unexpectedly, the composting plant obtained one of the worst evaluations from the EF and water consumption point of view. Nonetheless, the release of air emissions may also occur in other MSW treatment process, although this was not reported by LIPOR. Therefore, faired conclusions would be achieved if these data were also included.

Resumen

La industria es una de las principales responsables de la contaminación ambiental y del agotamiento de los recursos naturales, siendo ambos factores causa de la degradación ambiental; no obstante, también se reconoce su contribución al desarrollo y a la creación de riqueza. Por tanto, la identificación de opciones sostenibles en este campo es un factor clave. Hoy en día, la actitud adoptada frente a la prevención y el control de la contaminación y la producción limpia no consiste simplemente en una reacción a la aparición de leyes y regulaciones (Registro, Evaluación, Autorización y Restricción de Químicos –REACH-, Ley de Prevención y Control Integrados de la Contaminación –IPPC-, Política de Producto Integrada –PPI-), sino que es una cuestión de responsabilidad corporativa. Además, está demostrado que puede suponer una forma de incrementar los beneficios económicos. La definición de sostenibilidad ha recibido cierta crítica debido a su vaguedad, ambigüedad y la dificultad para trasladar este concepto a los diferentes niveles. Para superar la dificultad de su implementación, a lo largo de los años se han desarrollado una gran variedad de indicadores que proporcionan medidas necesarias para la toma de decisiones.

Esta tesis supone una contribución al desarrollo de herramientas de evaluación ambiental adaptadas a sectores particulares de producción, con el objetivo de proporcionar indicadores que guíen la toma de decisiones para el ecodiseño de procesos y productos sostenibles. Se ha propuesto la integración y combinación de metodologías de diferente naturaleza como la forma más adecuada de alcanzar evaluaciones completas. Al mismo tiempo, se buscó mantener lo máximo posible la simplicidad de las herramientas para facilitar su aplicación y hacerlas más atractivas para las empresas, evitando la necesidad de tener que llevar a cabo un entrenamiento excesivamente complejo.

El **Capítulo 1** presenta una revisión del estado del arte de indicadores relacionados con aspectos ambientales bajo una perspectiva de procesos y productos. Se revisaron indicadores de diferente naturaleza, desde aquellos con una dimensión territorial (Huella Ecológica, Espacio Ambiental, Índice de Área de Disipación) hasta los más genéricos de flujos de materia y energía (Energía, Exergía, Emergía, Huella Hídrica, Mochila Ecológica), ciclo de vida (Huella de Carbono) o indicadores de riesgo ambiental (Índice de Riesgo, Índice Carcinogénico). Se discutió su utilidad, inconvenientes y aplicaciones. Se observó

que los diferentes indicadores proporcionaban información complementaria sobre el comportamiento ambiental de procesos y productos; por tanto, si se usaran conjuntamente, se obtendrían evaluaciones más completas, superando así las debilidades de la aplicación de un solo indicador. Como consecuencia, se encontró interesante y beneficiosa la combinación de indicadores complementarios. También se revisaron las ventajas de marcos de trabajo de integración y la efectividad de diferentes metodologías de análisis multi-criterio de cara a poder expresar los resultados de las evaluaciones de sostenibilidad de una forma adecuada pero a la vez simplificada.

Así, en el Capítulo 1 se trató la utilidad del Análisis Multi-criterio (AMC) como soporte en el proceso de toma de decisiones al permitir considerar al mismo tiempo indicadores de las distintas dimensiones de la sostenibilidad. La obtención de un orden de prioridad de alternativas o un índice único tras la combinación de distintas metodologías de evaluación ambiental facilita la interpretación de resultados y simplifica el proceso; no obstante, la incertidumbre asociada a la imprecisión de los datos, establecimiento de pesos de importancia o los métodos de agregación también se deben tener en cuenta. Se introdujeron las metodologías de AMC más relevantes en el campo de la evaluación ambiental: 1) Teoría de utilidad multi-atributo: AHP, MACBETH; 2) Métodos de ordenación: ELECTREE, PROMETHEE, NAIADE; 3) Técnicas basadas en lógica borrosas. Además, se revisaron las distintas aplicaciones encontradas en la bibliografía científica.

El **Capítulo 2** proporciona una descripción general de los materiales y métodos empleados durante el desarrollo de la tesis. El capítulo se dividió en tres secciones principales. La primera recopila las metodologías relacionadas con la evaluación ambiental de procesos y productos: Huella Ecológica (HE), Análisis de Ciclo de Vida (ACV), Evaluación de Riesgo Ambiental (ERA) y Análisis de Flujos de Materia y Energía (AFME). La segunda parte trata el análisis multi-criterio, describiendo aquellas metodologías que fueron seleccionadas para su aplicación en diferentes capítulos de la tesis: AHP, técnicas de lógica borrosa, ELECTRE y PROMETHEE/GAIA. La tercera sección recoge las herramientas de análisis estadístico, principalmente análisis de sensibilidad y correlaciones estadísticas) que fueron aplicadas para asistir, complementar y dar consistencia a las distintas evaluaciones llevadas a cabo.

Los materiales empleados durante el desarrollo de la tesis se refieren principalmente a paquetes informáticos y bases de datos. Se estudiaron y emplearon diversos programas asociados a las metodologías de evaluación aplicadas y que fueron anteriormente relacionadas. Así, se utilizó SIMAPRO® para los estudios de ACV, Umberto® para AFME, Matlab y específicamente la ToolBox de Lógica Borrosa para el análisis multicriterio, Decision Lab® para PROMETHEE/GAIA; Crystal BAll® para los análisis de sensibilidad y SPSS® para análisis estadísticos generales. También se usó MS Excel® para implementar las herramientas simplificadas derivadas de la adaptación de HE, ACV y ERA a los procesos de producción estudiados. En relación a las bases de datos, se han consultado distintas fuentes para recopilar todos los parámetros necesarios para llevar a cabo el análisis propuesto. Estos se especifican más en detalle en cada uno de los capítulos en los que fueron usados.

El trabajo desarrollado en el **Capítulo 3** constituyó una aplicación pionera de la HE a un proceso productivo. Hasta entonces la HE había sido utilizada principalmente como indicador de sostenibilidad ambiental aplicada a estilos de vida individuales, regiones, países o el mundo. Aunque se había sugerido su aplicación a negocios y la industria, en el momento de la realización de este trabajo las iniciativas a este respecto eran muy escasas. En el estudio que se presenta aquí se analizó una planta de confección textil. El sector textil se sitúa dentro de aquellos más destacados en la estructura industrial española, especialmente en lo que se refiere al sub-sector de la confección. En Galicia (NO España) la industria de la moda ha adquirido una especial relevancia en los últimos años debido a la presencia de varios diseñadores de reconocido prestigio a nivel nacional e internacional. Esto causó un fuerte desarrollo de esta industria, lo que generó un importante impacto en la economía gallega. Por tanto, el desarrollo de herramientas para evaluar de forma adecuado las cargas ambientales asociadas a este sector resultaba particularmente relevante.

El objetivo global del Capítulo 3 era el desarrollo de una herramienta útil para evaluar la evolución del impacto ambiental debido a la actividad desarrollada en la fábrica de confección, así como para poder establecer comparaciones con otras plantas de similares características. Por tanto, se adaptó la HE al caso de estudio y se recopilaron los datos de inventario derivados del proceso de producción. Éstos se dividieron en tres categorías principales: energía, recursos y residuos. La principal contribución a la HE total (expresada en hectáreas de territorio) fue la de la categoría de recursos, principalmente debido a los elevados requerimientos de terreno para la producción de las telas de origen no sintético. El consumo de energía fue la segunda contribución más importante, mientras que la categoría de residuos permaneció en tercer lugar. Los valores finales obtenidos se dividieron entre los valores de producción (número de piezas de ropa confeccionadas) para calcular indicadores relativos más fácilmente comparables y también fácilmente interpretables por las distintas partes interesadas. Esta última característica es de especial importancia para una compañía que elabora memorias de Responsabilidad Social Corporativa (RSC) y que por tanto debe contar con una estrategia de comunicación.

Aunque el interés por la HE y su popularidad han ido creciendo a lo largo de los años, también ha sido objeto de crítica y controversia. Las ventajas de un indicador agregado se ensombrecen frecuentemente por las debilidades de su metodología. Así, la aplicación de la HE al proceso de producción del Capítulo 3 puso de manifiesto la imposibilidad de alcanzar una evaluación ambiental completa. Como respuesta, la investigación posterior se centró en encontrar soluciones para superar las principales críticas del indicador, que impedían considerar la HE como un indicador apropiado y útil para ser aplicado a procesos o a nivel industrial. Una de las carencias de la HE que se trata con mayor frecuencia en la bibliografía es que no tiene en cuenta aquellos contaminantes y residuos tóxicos o peligrosos que no pueden formar parte de un ciclo biológico cerrado. Esto supone un gran problema cuando se evalúan las cargas ambientales de un proceso industrial, en el que se generen este tipo de sustancias.

En relación a esto, la metodología desarrollada en el **Capítulo 4** estima la HE de residuos tóxicos y peligrosos considerando un ciclo cerrado modelado mediante un proceso de plasma, fenómeno que tiene lugar de forma natural en volcanes y estrellas. En un reactor de plasma tiene lugar una reacción dual simultánea: los compuestos orgánicos se descomponen térmicamente en sus elementos constituyentes (gas de síntesis con una más completa y ventajosa conversión del carbón en gas que en las incineradoras), mientras que los materiales inorgánicos se funden y se convierten en una escoria densa, inerte y no lixiviable, que no requiere de un vertido controlado. Por tanto, se puede considerar que es un sistema de tratamiento cerrado. El gas de síntesis (compuesto principalmente por CO y H_2) se puede usar para generar electricidad. La vitrificación es el resultado de la interacción entre el plasma y los materiales inorgánicos. Dado que la fracción inerte se vitrifica y que es muy difícil que se puedan desprender las posibles sustancias peligrosas que pueda contener, este producto se puede usar para la construcción de carreteras o para edificación.

Los residuos industriales se pueden tratar mediante un proceso de gasificación térmica por plasma y, mediante el desarrollo de una metodología basada en un modelo que describa este proceso, se estimó la HE de los residuos peligrosos. Sin embargo, esto no significa que se proponga el uso de la tecnología de plasma como la solución ideal para el tratamiento de todo tipo de residuos, sino que se utilizó como recurso metodológico para simular los ciclos cerrados de la naturaleza. La herramienta desarrollada se validó con el caso de estudio presentado en el Capítulo 3, tanto para residuos peligrosos como no peligros generados en la fábrica de confección. Para los no peligrosos se obtuvo un valor de 56.5 gha, que se encuentra dentro del mismo orden de magnitud que el que se había estimado en el Capítulo 3 usando la metodología convencional de HE.

La metodología presentada en el Capítulo 4 se centraba en los residuos sólidos; sin embargo, existen otros flujos que se pueden generar en un proceso de producción como pueden ser las emisiones a la atmósfera y al agua. El **Capítulo 5** trata sobre las primeras, dado que la HE se critica frecuentemente por no incluir otras emisiones aparte del CO_2. En este sentido, se consideraron otros gases de efecto invernadero (GEI) y, adicionalmente, se evaluó el efecto de incorporar emisiones acidificantes transformadas en requerimientos de superficie por medio de valores de carga crítica. Como caso de estudio en este caso se seleccionó la industria cerámica (fabricación de ladrillos de arcilla cocida) típica de Galicia para analizar el impacto de las emisiones derivadas de la combustión de fuel o gas natural durante las etapas de secado y cocción, así como su influencia en el valor de HE.

Otro aspecto controvertido es el uso de factores globales o locales en las evaluaciones de HE, lo que conlleva una discusión sobre si es preferible asegurar la comparabilidad entre estudios llevados a cabo en distintas partes del mundo, o realizar evaluaciones más exactas y ajustadas a las condiciones regionales. En el Capítulo 5 se estima la capacidad de absorción de CO_2 para Galicia (NO España) basándose en su capacidad para actuar como sumidero de carbono. Se emplearon dos metodologías diferentes para evaluar la biomasa utilizando datos del segundo y tercer inventarios forestales nacionales: a) Factor de expansión de biomasa (FEB); b) Ecuaciones alométricas. Se tuvieron en cuenta las principales especies presentes en los bosques gallegos: *Pinus pinaster*, *Eucalyptus globulus*, *Quercus pyrenaica*, *Quercus robur*, *Pinus radiata*, *Castanea sativa* and *Pinus sylvestris*. Los resultados obtenidos fueron: a) 3.83 t CO_2 ha^{-1} yr^{-1} para FEB y b) 4.33 t CO_2 ha^{-1} yr^{-1} para las ecuaciones alométricas. La primera mostraba una similitud razonable con el factor medio global (3.67 t CO_2 ha^{-1} yr^{-1}) utilizada en los informes de Planeta Vivo, pero la segunda resultó un 18% mayor. Las ecuaciones alométricas se pueden considerar como un método más exacto, puesto que las diferentes partes del árbol se evalúan de forma independiente para cada especie. Además, para la mayoría de las especies consideradas, se disponía de correlaciones desarrolladas en bosques gallegos. Una gestión sostenible de los bosques podría llevar a mantener una buena capacidad de asimilación de carbono en Galicia, contribuyendo así a mitigar el cambio climático. Esto se vería reflejado en la reducción de los valores de HE.

El análisis de la planta de confección presentada en el Capítulo 3 se retomó en el **Capítulo 6**. Para extender y hacer la evaluación más completa se incorporaron las nuevas propuestas metodológicas de los Capítulos 4 y 5. Por otro lado, además de la HE, se aplicaron otras metodologías de evaluación ambiental (AFME y ACV) para proporcionar una perspectiva complementaria. Se modeló el proceso usando el programa Umberto® 5.5 y se llevó a cabo un análisis basado en los flujos de energía y materiales. Se presentaron los resultados actualizados de la HE y se evaluaron las emisiones atmosféricas inicialmente (Capítulo 3) excluidas del análisis de HE mediante dos enfoques: 1) usando las categorías de impacto de ACV caracterizadas como indicadores complementarios; 2) incorporando las emisiones dentro de la HE mediante factores de absorción, incluyendo o no factores de ponderación, tal y como se discutió en el Capítulo 5. Finalmente, se llevaron a cabo análisis basados en simulaciones Monte Carlo para estudiar la influencia en el valor final del indicador en base a la variabilidad de las variables de entrada.

El mayor consumo de energía del proceso de producción se producía durante la etapa de corte, para la que ya se habían implementado estrategias de minimización. Acciones generales como el alumbrado o la calefacción de la fábrica representaban el segundo mayor consumo. Así, para reducir la demanda energética deberían tomarse medidas como la instalación de lámparas de bajo consumo, limitar el alumbrado a las zonas en las que es estrictamente necesario o regular el uso de la calefacción. Además, el balance de masa reveló que el gasoil, a pesar de su baja contribución al suministro energético, era una de las principales fuentes de contaminación de la fábrica. Por tanto, se recomendó su sustitución por fuentes de energía más limpias. Además de estos aspectos estrictamente relacionados con el comportamiento de la planta (p.ej. la regulación de los usos de energía anteriormente discutidos), las políticas de gestión estaban fuertemente relacionadas con la incorporación de criterios ambientales en el diseño de los productos. Así, debería minimizarse el consumo de materiales y la selección de telas debería basarse en aspectos ambientales en lugar de en tendencias de moda.

Después de tratar la evaluación ambiental de procesos productivos en capítulos anteriores, el **Capítulo 7** se centra en el ecodiseño de productos. El ecodiseño puede definirse como la introducción sistemática de aspectos ambientales durante el diseño y desarrollo del producto. Esto implica tener en cuenta los impactos ambientales en todas las fases del ciclo de vida del producto, empezando en las etapas de diseño y desarrollo. El objetivo es crear soluciones sostenibles que satisfagan las necesidades y los deseos humanos, buscando una solución de compromiso entre los criterios ambientales, técnicos, funcionales, ergonómicos, estéticos o económicos.

Cuando se maneja más de un indicador al mismo tiempo, surgen dificultades a la hora de tomar decisiones basadas en la información proporcionada por todos ellos. Como se ha indicado anteriormente, se ha demostrado la eficacia de las metodologías de AMC en la definición de marcos de trabajo de integración. En este caso se seleccionaron las técnicas de Lógica Borrosa (LB), comúnmente utilizadas para abordar problemas de incertidumbre. El uso de técnicas de LB permite obtener un enfoque cuantitativo utilizando una representación cualitativa; así, son capaces de manejar al mismo tiempo datos numéricos y conocimiento lingüístico. En este sentido, se desarrolló una herramienta que integraba los criterios proporcionados por tres metodologías de evaluación ambiental (HE, LCA, ERA) en base a la aplicación del razonamiento y características de LB. Esta idea permitió la toma de decisiones a nivel de proceso y producto teniendo en cuenta al mismo tiempo los valores de los distintos indicadores. Para establecer la importancia relativa de cada uno de ellos se definieron funciones de pertenencia como entradas al motor de inferencia borroso en el caso de un producto específico. Como resultado se obtuvo un Índice Borroso de EcoDiseño.

Se utilizó un caso de estudio ampliamente conocido para dar soporte al desarrollo de la herramienta y para probarla. En este sentido, se consideraron distintos materiales para envases de bebidas de cara a identificar la opción medioambientalmente más beneficiosa. Después de refinar la herramienta en base a la retroalimentación obtenida del primer caso de estudio, y siguiendo el mismo procedimiento y características, se adaptó la herramienta para su aplicación en el ecodiseño de calzado. Se analizaron cuatro modelos de calzado infantil y se compararon de acuerdo con el índice de ecodiseño obtenido. La herramienta identificó de forma adecuada aquellas propuestas de ecodiseño que deberían ser rechazadas (principalmente debido a la probabilidad de que pueda ser perjudicial para la salud humana durante su uso) y proporcionó una ordenación de alternativas basada en si su uso era más o menos apropiado desde un punto de vista ambiental y de seguridad. Esta información fue considerada por el equipo de diseño para incluir criterios ambientales en su toma de decisiones.

Finalmente, el **Capítulo 8** aborda la problemática de la gestión de Residuos Sólidos Urbanos (RSU). Uno de los principales retos del siglo XXI para los municipios es la adecuada recolección, reciclaje, tratamiento y eliminación de cantidades de residuo sólido en continuo crecimiento. Los residuos generan una serie de impactos en el medio ambiente, incluyendo la contaminación atmosférica, del agua superficial y subterránea; además, los vertederos ocupan un espacio muy valioso y, si se gestionan de forma inadecuada, puede causar riesgos en la salud de la población. La jerarquía de residuos definida en la Directiva 2008/98/EC sobre residuos establece el siguiente orden de prioridad para ser considerado en la definición de políticas y legislación sobre la gestión y prevención de residuos: 1) Prevención; 2) Preparación para la reutilización; 3) Reciclado; 4) otro tipo de valorización (p.ej. la valorización energética; 5) eliminación. No obstante, esta nueva Directiva también contempla la posibilidad de alterar esta jerarquía en situaciones específicas si así se justifica mediante un estudio de ciclo de vida.

Por tanto, la elección de opciones de tratamiento de los RSU es un proceso complejo en el que se deben tener en cuenta una gran variedad de criterios. Además de la perspectiva económica, situación geográfica o factores sociales, la toma de decisiones debe considerar el enfoque ambiental. Se han desarrollados muchos indicadores ambientales y de sostenibilidad con el objetivo de cuantificar estas cargas ambientales. Sin embargo, la integración de indicadores se ha considerado como la mejor opción para alcanzar evaluaciones más completas.

En la primera parte de este capítulo se analiza un caso de estudio extraído de referencias bibliográficas para priorizar entre cuatro opciones de tratamiento de RSU: vertedero con recuperación de energía; valorización energética de los RSU; tratamiento biológico de la fracción orgánica (FORSU) con valorización energética de combustible derivado de residuo (CDR); gasificación térmica por arco de plasma. En un primer enfoque se aplicó la HE como indicador individual; posteriormente, se añadieron más indicadores y se aplicaron metodologías de AMC (AHP, ELECTREE I y III, PROMETHEE I y II) para establecer una ordenación de las alternativas. Las opciones de tratamiento se ordenaron de la siguiente manera (de mejor a peor): gasificación por plasma, tratamiento biológico de FORSU con valorización energética de CDER, valorización energética de los RSU y finalmente el vertedero. Estos resultados mostraron concordancia con la jerarquía establecida en la Directiva. Se pudo comprobar que la HE había proporcionado una buena orientación previa sobre la evaluación de las alternativas.

En la segunda parte del capítulo se estudió un caso real de gestión de residuos urbanos: LIPOR – Servicio Intermunicipal de Gestión de Residuos del Gran Porto, aplicando conjuntamente HE y ACV. LIPOR es la entidad encargada de la gestión, recuperación y tratamiento de los RSU producidos en los ocho municipios que son socios de la mismo: Espinho, Gondomar, Maia, Matosinhos, Porto, Póvoa de Varzim, Valongo and Vila do Conde. Se analizaron las principales actividades del sistema integrado de gestión de LIPOR: valorización multi-material, valorización orgánica, valorización energética y vertedero. De este modo se pudieron identificar las partes del sistema que presentaban una mayor contribución a la contaminación ambiental derivada del sistema. La planta de valorización energética obtuvo la mejor evaluación desde el punto de vista de la HE; sin embargo, otros impactos ambientales como los derivados de las emisiones atmosféricas o el consumo de agua identificaban esta fase del tratamiento como más contaminante. Por otro lado, de forma inesperada, la planta de compostaje obtuvo la peor evaluación desde el punto de vista de la HE y del consumo de agua por kg de residuo tratado. No obstante, la emisión de contaminantes a la atmósfera se puede producir en otras etapas del sistema integrado de gestión además de en la valorización energética, aunque éstas no se hayan cuantificado por parte de LIPOR. Por tanto, para poder extraer conclusiones definitivas deberían incorporarse estos otros aspectos junto con otros inicialmente excluidos del análisis (p.ej. tratamiento de aguas residuales).

Resumo

A industria é unha das principais responsables da contaminación ambiental e do consumo dos recursos naturais, sendo ambos factores causa da degradación ambiental; non obstante, tamén é recoñecida a súa contribución ao desenvolvemento e á xeración de riqueza. Por tanto, a identificación de opcións sustentables neste campo é un factor clave. Hoxe en día, a actitude adoptada fronte ao control e prevención da contaminación e a produción limpa non consiste simplemente nunha actitude reactiva á aparición de leis e regulacións (Rexistro, Avaliación, Autorización e Restrición de Químicos –REACH-, Lei de Prevención e Control Integrados da Contaminación –IPPC-, Política de Produto Integrada –PPI-), senón que é unha cuestión de responsabilidade corporativa. Ademais, está demostrado que pode supoñer unha forma de incrementar os beneficios económicos. A definición de sostenibilidade ten recibido certa crítica debido á súa vaguidade, ambigüidade e á dificultade para trasladar este concepto aos diferentes niveis. Para superar a dificultade da súa implantación, ao longo dos anos téñense desenvolvido unha grande variedade de indicadores que proporcionan medidas necesarias para a toma de decisións.

Esta tese supón unha contribución ao desenvolvemento de ferramentas de avaliación ambiental adaptadas a sectores particulares de produción, co obxectivo de proporcionar indicadores que guíen a toma de decisións para o ecodeseño de procesos e produtos sustentables. Propúxose a integración e a combinación de metodoloxías de diferente natureza como a forma máis axeitada de acadar avaliacións completas. Ao mesmo tempo, buscouse manter o máximo posible a simplicidade das ferramentas para facilitar a súa aplicación e facelas máis atractivas para as empresas, evitando a necesidade de ter que recibir un adestramento excesivamente complexo.

O **Capítulo 1** presenta unha revisión do estado da arte de indicadores relacionados con aspectos ambientais baixo unha perspectiva de procesos e produtos. Revisáronse indicadores de diferente natureza, dende aqueles cunha dimensión territorial (Pegada Ecolóxica, Espazo Ambiental, Índice de Área de Disipación) ata os máis xenéricos de fluxos de materia e enerxía (Enerxía, Exerxía, Emerxía, Pegada Hídrica, Mochila Ecolóxica), ciclo de vida (Pegada do Carbono) ou indicadores de risco ambiental (Índice de Risco, Índice Carcinoxénico). Discutiuse a súa utilidade, inconvenientes e aplicacións. Observouse que os diferentes

indicadores proporcionaban información complementaria sobre o comportamento ambiental de procesos e produtos; polo tanto, si foran empregados conxuntamente, obteríanse avaliacións máis completas, superando así as debilidades da aplicación dun so indicador. Como consecuencia, atopouse interesante e beneficiosa a combinación de indicadores complementarios. Tamén se revisaron as avantaxes de marcos de traballo de integración e a efectividade de diferentes metodoloxías de análise multi-criterio de cara a poder expresar os resultados das avaliacións de sostenibilidade dunha forma axeitada pero á vez simplificada.

Así, no Capítulo 1 tratouse a utilidade da Análise Multi-criterio (AMC) como soporte no proceso de toma de decisións ao permitir considerar ao mesmo tempo indicadores das distintas dimensións da sostenibilidade. A obtención dunha orde de prioridade das alternativas ou un índice único tras a combinación de distintas metodoloxías de avaliación ambiental facilita a interpretación de resultados e simplifica o proceso; non obstante, a incerteza asociada á imprecisión dos datos, establecemento de pesos de importancia ou os métodos de agregación tamén deben ser tidos en conta. Presentáronse as metodoloxías de AMC máis relevantes no campo da avaliación ambiental: 1) Teoría de utilidade multi-atributo: AHP, MACBETH; 2) Métodos de ordenación: ELECTREE, PROMETHEE, NAIADE; 3) Técnicas baseadas en lóxica borrosa. Ademais, resumíronse as distintas aplicacións atopadas na bibliografía científica.

O **Capítulo 2** proporciona unha descrición xeral dos materiais e métodos empregados durante o desenvolvemento da tese. O capítulo dividiuse en tres seccións principais. A primeira delas recompila as metodoloxías relacionadas coa avaliación ambiental de procesos e produtos: Pegada Ecolóxica (PE), Análise de Ciclo de Vida (ACV), Avaliación de Risco Ambiental (ARA) e Análise de Fluxos de Materia e Enerxía (AFME). A segunda parte trata da análise multi-criterio, describindo aquelas metodoloxías que foron seleccionadas para a súa aplicación en diferentes capítulos da tese: AHP, técnicas de lóxica borrosa, ELECTRE e PROMETHEE/GAIA. A terceira sección recolle as ferramentas de análise estatístico, principalmente análise de sensibilidade e correlacións estatísticas) que foron aplicadas para asistir, complementar e dar consistencia ás distintas avaliacións levadas a cabo.

Os materiais empregados durante o desenvolvemento da tese refírense principalmente a paquetes informáticos e bases de datos. Estudáronse e empregáronse diversos programas asociados ás metodoloxías de avaliación aplicadas e que foron anteriormente relacionadas. Así, utilizouse SIMAPRO® para os estudos de ACV, Umberto® para AFME, Matlab e especificamente a ToolBox de Lóxica Borrosa para a análise multi-criterio, Decision Lab® para PROMETHEE/GAIA; Crystal BAll® para as análises de sensibilidade e SPSS® para análises estatísticas xerais. Tamén se empregou MS Excel® para implantar as ferramentas simplificadas derivadas da adaptación de PE, ACV e ARA aos procesos de produción estudados. En relación ás bases de datos, consultáronse distintas fontes para recompilar todos os parámetros precisos para levar a cabo a análise proposta. Estes especifícanse máis en detalle en cada un dos capítulos nos que foron empregados.

O traballo desenvolvido no **Capítulo 3** constitúe unha aplicación pioneira da PE a un proceso produtivo. Ata ese momento a PE tiña sido empregada principalmente como indicador de sostenibilidade ambiental aplicada a estilos de vida individuais, rexións, países e o mundo. Aínda que se tiña suxerido a súa aplicación a negocios e á industria, no momento da realización deste traballo as iniciativas a este respecto eran moi escasas. No estudo que se presenta aquí analizouse unha planta de confección téxtil. O sector téxtil sitúase dentro daqueles máis destacados na estrutura industrial española, especialmente no que se refire ao sub-sector da confección. En Galicia (NO España) a industria da moda adquiriu unha especial relevancia nos últimos anos debido á presenza de varios deseñadores de recoñecido prestixio a nivel nacional e internacional. Isto causou un forte desenvolvemento desta industria, o que xerou un importante impacto na economía galega. Polo tanto, o desenvolvemento de ferramentas para avaliar de forma axeitada as cargas ambientais asociadas a este sector resultaba particularmente relevante.

O obxectivo global do Capítulo 3 era o desenvolvemento dunha ferramenta útil para avaliar a evolución do impacto ambiental debido á actividade desenvolvida na fábrica de confección, así como para poder establecer comparacións con outras plantas de similares características. Polo tanto, adaptouse a PE ao caso de estudo e recompiláronse os datos de inventario derivados do proceso de produción. Estes dividíronse en tres categorías principais: enerxía, recursos e residuos. A principal contribución á PE total (expresada en hectáreas de terra) foi a da categoría de recursos, principalmente debido aos elevados requirimentos de terreo para a produción das telas de orixe non sintética. O consumo de enerxía foi a segunda contribución máis importante, mentres que a categoría de residuos permaneceu en terceiro lugar. Os valores finais obtidos dividíronse entre os valores de produción (número de pezas de roupa confeccionadas) para calcular indicadores relativos máis facilmente comparables e tamén facilmente interpretables polas distintas partes interesadas. Esta última característica é de especial importancia para unha compañía que elabora memorias de Responsabilidade Social Corporativa (RSC) e que por tanto debe contar cunha estratexia de comunicación.

Aínda que o interese pola PE e a súa popularidade teñen crecido ao longo dos anos, tamén ten sido obxecto de crítica e controversia. As avantaxes dun indicador agregado ensombrécense frecuentemente polas debilidades da súa metodoloxía. Así, a aplicación da PE ao proceso de produción do Capítulo 3 puxo de manifesto a imposibilidade de acadar unha avaliación ambiental completa. Como resposta, a investigación posterior centrouse en atopar solucións para superar as principais críticas do indicador, que impedían considerar a PE como un indicador apropiado e útil para ser aplicado a procesos ou a nivel industrial. Unha das carencias da PE que se trata con maior frecuencia na bibliografía é o feito de que non inclúe os contaminantes e os residuos tóxicos ou perigosos que non poden formar parte dun ciclo biolóxico cerrado. Isto supón un grande problema cando se avalían as cargas ambientais dun proceso industrial, no que se espera que se xeren este tipo de sustancias.

En relación a isto, a metodoloxía desenvolvida no **Capítulo 4** estima a PE de residuos tóxicos e perigosos considerando un ciclo cerrado modelado mediante un proceso de plasma, fenómeno que ten lugar de forma natural en volcáns e estrelas. Nun reactor de plasma ten lugar unha reacción dual simultánea: os compostos orgánicos descompóñense por acción térmica nos elementos constituíntes (gas de síntese con un máis completa e avantaxe de conversión do carbón en gas que nas incineradoras), mentres que os materiais inorgánicos fúndense e convértense nunha escoura densa, inerte e non lixiviable, que non require dun vertido controlado. Por tanto, pódese considerar que é un sistema de tratamento pechado. O gas de síntese (composto principalmente por CO e H_2) pódese usar para xerar electricidade. A vitrificación é o resultado da interacción entre o plasma e os materiais inorgánicos. Dado que a fracción inerte se vitrifica e que é moi difícil que se poidan desprender as posibles substancias perigosas que poida conter, este produto pódese usar para a construción de estradas ou para edificación.

Os residuos industriais pódense tratar mediante un proceso de gasificación térmica por plasma e, mediante o desenvolvemento dunha metodoloxía baseada nun modelo que describa este proceso, estimouse a PE dos residuos perigosos. Sen embargo, isto non significa que se propoña o uso da tecnoloxía de plasma como a solución ideal para o tratamento de todo tipo de residuos, senón que se empregou como recurso metodolóxico para simular os ciclos cerrados da natureza. A ferramenta desenvolvida validouse co caso de estudo presentado no Capítulo 3, tanto para residuos perigosos como non perigosos xerados na fábrica de confección. Para os non perigosos obtívose un valor de 56.5 gha, que se atopa dentro da mesma orde de magnitude que o que se tiña estimado no Capítulo 3 usando a metodoloxía convencional de PE.

A metodoloxía presentada no Capítulo 4 centrábase nos residuos sólidos; sen embargo, existen outros fluxos que se poden xerar nun proceso de produción como poden ser as emisións á atmosfera e á auga. O **Capítulo 5** trata sobre as primeiras, dado que a PE é frecuentemente criticada por non incluír outras emisións ademais do CO_2. Neste senso, consideráronse outros gases de efecto invernadoiro (GEI) e, adicionalmente, avaliouse o efecto de incorporar emisións acidificantes transformadas en requirimentos de superficie por medio de valores de carga crítica. Como caso de estudio seleccionouse a industria cerámica (fabricación de ladrillos de arxila cocida) típica de Galicia para analizar o impacto das emisións derivadas da combustión de fuel ou gas natural durante as etapas de secado e cocción, así como a súa influencia no valor de PE.

Outro aspecto controvertido é o uso de factores globais ou locais nas avaliacións de PE, o que conleva unha discusión sobre se é preferible asegurar a comparabilidade entre estudos levados a cabo en distintas partes do mundo, ou realizar avaliacións máis exactas e axustadas ás condicións rexionais. No Capítulo 5 estímase a capacidade de absorción de CO_2 para Galicia (NO España) baseándose na súa capacidade para actuar como sumidoiro de carbono. Empregáronse dúas metodoloxías diferentes para avaliar a biomasa utilizando datos do segundo e terceiro inventarios forestais nacionais: a) Factor de expansión de biomasa (FEB); b) Ecuacións alométricas. Tivéronse en conta as principais especies presentes nos bosques galegos: *Pinus pinaster*, *Eucalyptus* *globulus*, *Quercus pyrenaica*, *Quercus robur*, *Pinus radiata*, *Castanea sativa* and *Pinus sylvestris*. Os resultados obtidos foron: a) 3.83 t CO_2 ha^{-1} yr^{-1} para FEB e b) 4.33 t CO_2 ha^{-1} yr^{-1} para as ecuacións alométricas. A primeira amosou unha similitude razoable co factor medio global (3.67 t CO_2 ha^{-1} yr^{-1}) empregado nos informes de Planeta Vivo, pero a segunda resultou un 18% maior. As ecuacións alométricas pódense considerar como un método máis exacto, posto que as diferentes partes da árbore son avaliadas de forma independente para cada especie. Ademais, para a maioría das especies consideradas, dispoñíase de correlacións desenvolvidas en bosques galegos. Unha xestión sustentable dos bosques podería levar a manter unha boa capacidade de asimilación de carbono en Galicia, contribuíndo así a mitigar o cambio climático. Isto veríase reflectido na redución dos valores de PE.

A análise da planta de confección presentada no Capítulo 3 retomouse no **Capítulo 6**. Para estender e facer a avaliación máis completa incorporáronse as novas propostas metodolóxicas dos Capítulos 4 e 5. Por outra banda, ademais da PE, aplicáronse outras metodoloxías de avaliación ambiental (AFME e ACV) para proporcionar unha perspectiva complementaria. Modelouse o proceso usando o programa Umberto® 5.5 e levouse a cabo unha análise baseada nas correntes de enerxía e materiais. Presentáronse os resultados actualizados da PE e avaliáronse as emisións atmosféricas inicialmente (Capítulo 3) excluídas da análise de PE mediante dous enfoques: 1) usando as categorías de impacto de ACV caracterizadas como indicadores complementarios; 2) incorporando as emisións dentro da PE mediante factores de absorción, incluíndo ou non factores de ponderación, tal e como se discutiu no Capítulo 5. Finalmente, leváronse a cabo análises baseadas en simulacións Monte Carlo para estudar a influencia no valor final do indicador en base á variabilidade das variables de entrada.

O maior consumo de enerxía do proceso de produción producíase durante a etapa de corte, para a que xa se tiñan implantado estratexias de minimización. Accións xerais como o alumeado ou a calefacción da fábrica representaban o segundo maior consumo. Así, para reducir a demanda enerxética deberían tomarse medidas como a instalación de lámpadas de baixo consumo, limitar o alumeado ás zonas nas que é estritamente necesario ou regular o uso da calefacción. Ademais, o balance de masa revelou que o gasoil, a pesar da súa baixa contribución ao subministro enerxético, era unha das principais fontes de contaminación da fábrica. Polo tanto, recomendouse a súa substitución por fontes de enerxía máis limpas. Ademais destes aspectos estritamente relacionados co comportamento da planta (p.ex. a regulación dos usos de enerxía anteriormente discutidos), as políticas de xestión estaban fortemente relacionadas coa incorporación de criterios ambientais no deseño dos produtos. Así, debería minimizarse o consumo de materiais e a selección de telas debería basearse en aspectos ambientais en lugar de en tendencias de moda.

Despois de tratar a avaliación ambiental de procesos produtivos en capítulos anteriores, o **Capítulo 7** centrouse no ecodeseño de produtos. O ecodeseño pode definirse como a introdución sistemática de aspectos ambientais durante o deseño e desenvolvemento do produto. Isto implica ter en conta os impactos ambientais en todas as fases do ciclo de vida do produto, comezando nas etapas de deseño e desenvolvemento. O obxectivo é crear solucións sustentables que satisfagan as necesidades e os desexos humanos, buscando unha solución de compromiso entre os criterios ambientais, técnicos, funcionais, ergonómicos, estéticos ou económicos.

Cando se manexa máis dun indicador ao mesmo tempo, xorden dificultades á hora de tomar decisións baseadas na información proporcionada por todos eles. Como se indicou anteriormente, as metodoloxías de AMC teñen demostrado a súa eficacia na definición de marcos de traballo de integración. Neste caso seleccionáronse as técnicas de Lóxica Borrosa (LB), comunmente empregadas para abordar problemas de incerteza. O uso de técnicas de LB permite obter un enfoque cuantitativo utilizando unha representación cualitativa; así, é capaz de manexar ao mesmo tempo datos numéricos e coñecemento lingüístico. Neste senso, desenvolveuse unha ferramenta que integraba os criterios proporcionados por tres metodoloxías de avaliación ambiental (PE, LCA, ARA) en base á aplicación do razoamento e as características de LB. Esta idea permitiu a toma de decisións a nivel de proceso e produto tendo en conta ao mesmo tempo os valores dos distintos indicadores. Para establecer a importancia relativa de cada un deles definíronse funcións de pertenza como entradas ao motor de inferencia borroso no caso específico dun produto concreto. Como resultado obtívose un Índice Borroso de EcoDeseño (IBED).

Utilizouse un caso de estudo amplamente coñecido para dar soporte ao desenvolvemento da ferramenta e para validala. Así, consideráronse distintos materiais para envases de bebidas de cara a identificar a opción medioambientalmente máis beneficiosa. Despois de refinar a ferramenta en base á retroalimentación obtida do primeiro caso de estudo, e seguindo o mesmo procedemento e características, mellorouse a ferramenta para a súa aplicación no ecodiseño de calzado. Analizáronse catro modelos de calzado infantil e comparáronse dacordo co IBED obtido. A ferramenta identificou de forma axeitada aquelas propostas de ecodiseño que deberían ser rexeitadas (principalmente debido á probabilidade de prexudicar a saúde humana durante o seu uso) e proporcionou unha ordenación de alternativas baseada en se o uso era máis ou menos axeitado desde un punto de vista ambiental e de seguridade. Esta información foi tida en conta polo equipo de deseño para incorporar a vertente ambiental na súa toma de decisións.

Finalmente, o **Capítulo 8** aborda a problemática da xestión de Residuos Sólidos Urbanos (RSU). Un dos principais retos do século XXI para os municipios é a axeitada recolección, reciclaxe, tratamento e eliminación de cantidades de residuo sólido en continuo crecemento. Os residuos ocasionan unha serie de impactos no medio ambiente, incluíndo a contaminación atmosférica, da auga superficial e subterránea; ademais, os vertedoiros ocupan un espazo moi valioso e, se se xestionan de forma inadecuada, pode causar riscos na saúde da poboación. A xerarquía de residuos definida na Directiva 2008/98/EC sobre residuos establece a seguinte orde de prioridade para ser considerada na definición de políticas e lexislación sobre a xestión e prevención de residuos: 1) Prevención; 2) Preparación para a reutilización; 3) Reciclado; 4) outro tipo de valorización (p.ex. la valorización enerxética; 5) eliminación. Non obstante, esta nova Directiva tamén contempla a posibilidade de alterar esta xerarquía en situacións específicas si así se xustifica mediante un estudo de ciclo de vida.

Por tanto, a elección de opcións de tratamento dos RSU é un proceso complexo no que se deben ter en conta unha grande variedade de criterios. Ademais da perspectiva económica, situación xeográfica ou factores sociais, a toma de decisións debe considerar o enfoque ambiental. Nos últimos anos téñense desenvolvido moitos indicadores ambientais e de sostenibilidade co obxectivo de cuantificar estas cargas ambientais. A integración de indicadores considerouse como a mellor opción para acadar avaliacións máis completas.

Na primeira parte deste capítulo analízase un caso de estudo extraído de referencias bibliográficas para establecer prioridades entre catro opcións de tratamento de RSU: vertedoiro con recuperación de enerxía; valorización enerxética dos RSU; tratamento biolóxico da fracción orgánica (FORSU) con valorización enerxética do combustible derivado de residuo (CDR); gasificación térmica por arco de plasma. Nunha primeira aproximación aplicouse a PE como indicador individual; posteriormente, engadíronse máis criterios e aplicáronse metodoloxías de AMC (AHP, ELECTREE I y III, PROMETHEE I y II) para establecer unha ordenación de las alternativas. As opcións de tratamento ordenáronse do seguinte xeito (de mellor a pero): gasificación por plasma, tratamento biolóxico de FORSU con valorización enerxética de CDER, valorización enerxética dos RSU e finalmente o vertedoiro. Estes resultados amosaron concordancia coa xerarquía establecida na Directiva. Ademais, neste caso púidose comprobar que a PE tiña proporcionado unha boa orientación previa sobre a avaliación das diferentes alternativas.

Na segunda parte do capítulo estudouse un caso real de xestión de residuos urbanos: LIPOR – Servizo Intermunicipal de Xestión de Residuos do Grande Porto. Con este obxectivo propúxose a aplicación conxunta da PE e ACV. LIPOR é a entidade encargada da xestión, recuperación e tratamento dos RSU producidos nos oito municipios que son socios da mesmo: Espinho, Gondomar, Maia, Matosinhos, Porto, Póvoa de Varzim, Valongo and Vila do Conde. Analizáronse as principais actividades do sistema integrado de xestión de LIPOR: valorización multi-material, valorización orgánica, valorización enerxética e vertedoiro. Deste xeito puidéronse identificar as partes do sistema que presentaban unha maior contribución á contaminación ambiental derivada do sistema. A planta de valorización enerxética obtivo a mellor avaliación desde o punto de vista da PE; sen embargo, outros impactos ambientais como os derivados das emisións atmosféricas ou o consumo de auga identificaban esta fase do tratamento como a máis contaminante. Por outro banda, de forma inesperada, a planta de compostaxe obtivo a peor avaliación desde o punto de vista da PE e do consumo de auga por kg de residuo tratado. No obstante, a emisión de contaminantes á atmosfera pódese producir noutras etapas do sistema integrado de xestión ademais de na valorización enerxética, aínda que estas non están recollidas nos informes de sostenibilidade da LIPOR. Polo tanto, para poder extraer conclusións definitivas deberían incorporarse estes outros aspectos, así como as necesidades de tratamento das augas residuais.

Chapter 1

Introduction

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Abstract

The sustainability definition has received certain criticism for its vagueness and ambiguity. To overcome the difficulties of its implementation, a wide variety of indicators have been developed and applied over the years, providing metrics essential at the action level. This chapter presents a review of the state of the art of indicators concerning environmental issues, under a process- and product-oriented approach. Indicators of different nature have been reviewed, from those with a territorial dimension to the more generic material and energy flows, life-cycle or environmental risk indicators. Their usefulness, drawbacks and applications have been discussed. It was observed that, the different indicators provide complementary information about environmental performance of processes and products; thus, when used together, a more comprehensive evaluation can be obtained, while the weaknesses of indicators are overcome. Thus, the combination of complementary indicators can be particularly interesting and beneficial. The advantages of integrative frameworks and the effectiveness of different multi-criteria analysis methodologies to obtain a fair and simplified final result of the sustainability appraisals are also reviewed.

Multi-criteria analysis methodologies (namely AHP, PROMETHEE or ELECTRE) are widely applied to handle at a time the indicators from the different dimensions of sustainability, thus aiding decision making. The ranking or the single comparable index obtained when different indicators are combined helps to better interpret the results and simplify the evaluation problem; albeit, uncertainty associated to imprecision of data, weighting schemes or aggregation methods must be taken into account. Therefore, additional research is needed to acquire further knowledge and understanding of different types of uncertainty inherent in environmental decision-making, and how they affect the quality of decisions when MCA is applied.

1.1. Introduction

Industry is recognized as one of the main sources of environmental pollution and resource depletion, both causing environmental degradation; nonetheless, its contribution to development and wealth creation is also acknowledged. Therefore, the identification of sustainable options in this area is a key factor (Azapagic and Perdan, 2000).

In a sustainable production, the conservation of energy and natural resources is pursued, as well as the minimization of pollution. Industrialized countries such as those of the European Union (EU) share a special responsibility and opportunity for addressing the challenge of a more efficient use of resources. Environmental management as a whole is a complex strategy concerning a wide variety of issues for which there is not a single approach to deal with. Economically viable, socially beneficial, safe and healthful are other desired characteristics for such processes and systems (Veleva and Ellenbecker, 2001).

In this respect, different attitudes have been adopted over the years (Sikdar, 2003a). At first, just corrective actions were carried out as a response to emerging environmental laws and regulations, but soon businesses realized that if pollution prevention and cleaner production policies were adopted, not only environmental improvements would take place, but also an increase in profits (Azapagic and Perdan, 2000). A change from a reactive to a more proactive attitude has succeeded thus avoiding or reducing human and ecological health impacts. Indicators can also provide an early warning, sounding the alarm in time to prevent economic, social and environmental damage. The major level of sustainability consciousness has been achieved by those businesses that do not only care about legal compliance, but consider the implementation of voluntary environmental management systems as a strategy to compete in markets. To translate this statement of intent into more realistic aims, better tools and methodologies that offer an objective evaluation of the real estate of natural capital reservoirs are required.

Since the United Nations (UN) and national governments worldwide have been the driving force behind sustainable development, most frameworks developed to assess sustainability have subsequently focused on a national, regional or community level (Labuschagne et al., 2005). The EU has demonstrated special interest in the development of indicators in the scope of sustainability appraisals, and especially in the efficient utilization of natural resources. Recently, particular attention has been focused on the capability of the Ecological Footprint (EF) to measure sustainable use of resources, as it can be inferred from the different reports released in the last years, e.g. Ecological Footprinting (ECOTEC, 2001), Ecological Footprint and Biocapacity (EUROSTAT, 2006) or Potential of the Ecological Footprint for monitoring environmental impacts from natural resource use (Best et al. 2008). The study by Best et al. also assessed how the EF could best be combined with other tools to meet the EU's desired monitoring objectives: Environmentally Weighted Material Consumption (EMC), Human Appropriation of Net Primary Production (HANPP) and Land and Ecosystem Accounts (LEAC).

Nonetheless, there are other contributions that involve other kind of indicators like those material flow derived (EUROSTAT, 2000) or the Drivers–Pressures–State–Impacts–Responses (DPSIR) framework (EEA, 1995; Holten-Andersen et al., 1995). The DPSIR framework is a widely accepted and commonly used framework for interdisciplinary indicator development, system and model conceptualization, and the structuring of integrated research programs and assessments (Svarstad et al., 2008).

It is remarkable the common awareness of the need to define a group of indicators in order to cover all the areas required to achieve a comprehensive environmental appraisal. Attending at the assessment of the environmental performance of processes and products, some of the available indicators reviewed can provide complementary information. Thus, the difficulty often arises from the lack of criteria to select the appropriate indicators to be used in each case, rather than from the scarcity of scientifically sounded methodologies (Niemeijer and de Groot, 2008).

This chapter aims at reviewing the different indicators developed in the last years to measure progress towards sustainability, particularly those concerning environmental issues, under a process- and product- oriented approach. Their limitations are highlighted and the need of their complementary use is proposed and the appropriateness of their application in diverse situations is assessed. Options of integration and the applicability of Multi-Criteria Analysis (MCA) methodologies as a tool to handle at a time a set of indicators were also explored and reviewed.

1.2. The need of indicators development

If certain vagueness or ambiguity can be assigned to the definition of sustainable development, these can be reduced or eliminated through the development and application of indicators, which provide metrics essential at the action level (Tibbs, 1999; Johnston et al., 2007).

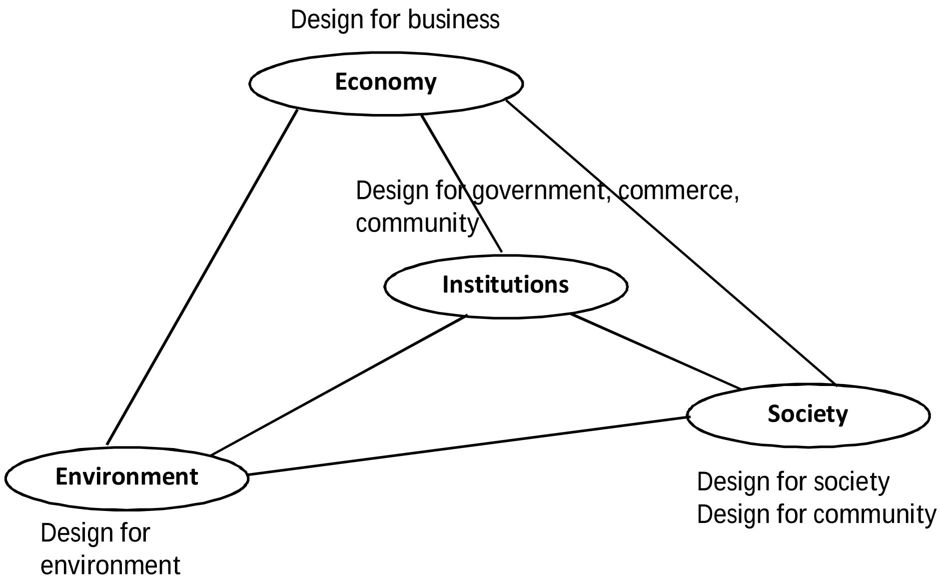

Figure 1.1. The four dimensions of sustainability.

Currently, sustainability is considered to comprise four dimensions: environmental, social, economic and institutional (Figure 1.1). For the former three, indicators have been developed in abundance, whereas for the institutional dimension indicator proposals are still quite rare (Spangenbergh, 2002). Hence, future steps should be addressed to the creation of comprehensive frameworks consisting of a generic set of indicators to enable a consistent comparison and identification of more sustainable options. This is in accordance with the underlying idea in the Triple Bottom Line concept (TBL), term coined by John Elkington (1994), consisting in the evaluation of the economic-financial, social and environmental elements in an integrated manner (Singh et al., 2009). This can be developed across a range of scales, namely the financial accounts of a firm, a service or a product (Foran et al., 2005).

The WCED (World Commission on Environment and Development) dissertation on sustainability considered the Planet Earth as a whole (WCED, 1987). However, there are different subsystems and levels at which sustainability can be addressed. For example, Batterham (2006) considered 5 levels ranging from global objectives to individual actions, while Sikdar (2003a) proposed a classification based on physical boundaries more than in organizational ones. In the context of production processes, it is a key issue to incorporate the sustainability concept into process and product design, manufacturing, and value chain management to prevent the consequences of unsustainable resource utilization and adverse environmental impacts. Thus, it is necessary to continue working on the development and improvement of indicators that can be used across all scales of application, in order to translate broad goals into decision-making processes.

The ultimate purpose of any performance measurement scorecard is to change behavior (Hussey et al., 2001). Unfortunately, many companies appear to view reporting as a sustainability strategy in itself, rather than as a tool to measure progress towards sustainability targets (Batterham, 2006). Albeit, when a metric is relevant, understandable and reliable, it can impact the consumer choice and ultimately influence legislative and regulatory action (MacLean, 2001; European Commission, 2003). For production processes and services, the availability of a set of indicators would allow comparison and benchmarking within the organization itself or against other companies, apart from evaluating environmental performance evolution (Azapagic and Perdan, 2000). In some cases, the lack of suitable evaluation tools makes necessary the development of comprehensive combined or integrated tools and indicators to evaluate the behavior of a particular process or product. Meanwhile, in other cases, an adapted and simplified tool expressing results in indexes easy to be interpreted by the different stakeholders is desirable. This is in accordance with the initiatives promoted by the Integrated Product Policy (IPP), which aims to support the implementation of environmental product innovations in order to achieve a broad reduction of all environmental impacts along a product's life cycle (European Commission, 2003).

1.3. Environmental indicators

According to the definition given by the European Environmental Agency (EEA), an environmental indicator is an observed value representative of a phenomenon under study (EEA, 1999). Indicators quantify information by aggregating different and multiple data (necessary to obtain reliable information); thus, they can be used to illustrate and communicate complex phenomena in a simpler way, including trends and progresses over a certain period of time (Roca et al., 2005). Actually, successful environmental policy requires that the condition of complex environmental systems is captured in one or more simple figures understandable to policymakers and the general public (Niemeijer, 2002). Besides, to be more effective, indicators should consider the inter-generational equity, one of the pillars of sustainability often disregarded by those indexes aiming at representing a fixed picture of the environmental status in a certain moment (Pan and Kao, 2009).

Indicators must provide information about the main characteristics that affect the suitability of products and processes from a sustainability viewpoint. These are: energy use per unit of economic value-added; intensity and type of energy used (renewable or non-renewable); materials use (or resource depletion); fresh water use; waste and pollutants production; environmental impacts of product/process/service; assessment of overall risk to human health and the environment (Sikdar, 2003b).

Next, a review of environmental indicators is presented, classified into the following categories: indicators of material and energy flows; indicators with a territorial dimension; indicators of environmental life-cycle assessment; indicators of environmental risk assessment (Figure 1.2).

1.3.1. Indicators of Material and Energy Flows

Flows of energy and material are valuable environmental indicators both at micro and macro scale. Actually, a key task of Industrial Ecology is to identify, trace and allocate energy and material flows throughout the system (Lou et al., 2004). Dematerialization is one of the mechanisms to deal with sustainability objectives. This means the reduction of material flows and substitution, i.e. exchange of type/quality of flows and/or activities, that can be planned in parallel and at different scales, e.g from changing amounts and types of fuel in the same process, through a more radical change of the whole process, to completely new and less resource demanding and more ecologically and socially sound ways of satisfying the same human need (Robèrt et al., 2002). Efficiency in resource use is directly related to the Factor X approach, i.e. by what factor can or should certain flows be reduced.

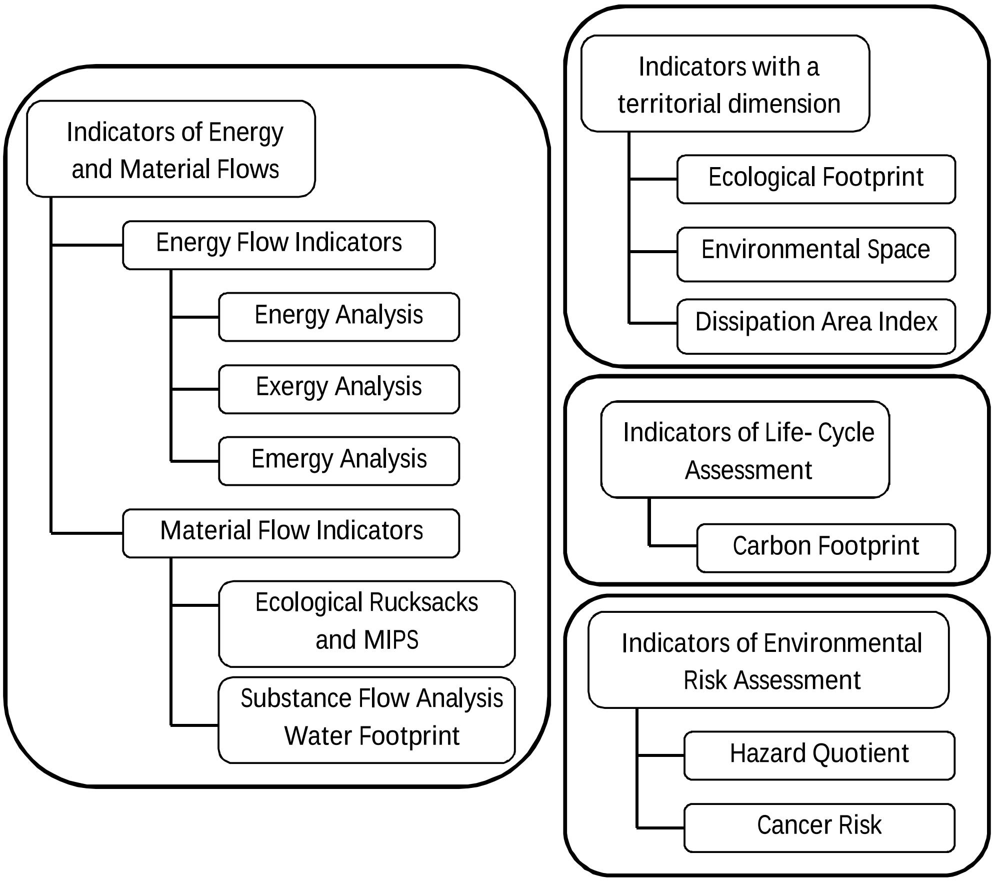

Figure 1.2. Indicators of the environmental dimension of sustainability.

Energy and Material Flow Analysis (EMFA) is an assessment methodology of environmental issues and a decision-support method that can be defined as a systematic appraisal of the flows and stocks of energy and material within a system defined in space and time (Torres et al., 2008). When applied to production processes, the main aim is to pursue reductions in the consumption of energy, raw material, water and in the discharge of effluents, emissions or wastes. Torres et al. (2008) applied the EMFA to reduce the environmental impact of the storage stage of clay in the roof-tile manufacture.

1.3.1.1. Energy flow indicators

Energy analysis

Energy analysis is the process of determining the energy required directly and indirectly to allow a system to produce a specified good or service (Nilsson, 1997; Herendeen, 2004). It accounts for the different types of energy in the same analysis. A key concept is the embodied energy, which is the direct and indirect energy required to produce a good or a service (Herendeen, 2004). Different studies have been conducted to assess energy consumption in production processes and energy embedded in products (Sakamoto et al., 1999; Bernard and Côté, 2005; Ramirez and Worrell, 2006; Neelis et al., 2007).

Exergy analysis

Exergy is an efficient indicator for energy policy making applications since it is a measure of quantity and quality of the energy sources, unlike energy which only informs about the quantity (Hovelius, 1997). From a thermodynamic point of view, exergy is defined as the maximum amount of work which can be produced by a system or a flow of matter or energy as it comes to equilibrium with a reference environment (Rosen and Dincer, 2001). Its application in the environmental impact evaluation of industrial processes has been explored (Hau and Bakshi, 2004a; Zhu et al., 2005), as well as its usefulness to measure the optimal use of energy in processes (Banat and Jwaied, 2008) or in buildings (Torío et al., 2009). It has also been employed to measure water quality (Huang et al., 2007) or to assess the efficiency of resources use and losses of quality during recycling processes (Castro et al., 2007).

Emergy analysis

Emergy, term introduced by H. Odum in the 1980's (Odum, 1988; Brown and Ulgiati, 2004), is defined as the solar energy directly or indirectly necessary to obtain a product in a process and it is expressed in solar emergy joules (seJ). To carry out the conversion into the solar equivalent, it is necessary to know the solar transformity, which is the emergy used to make a unit of available energy of a product or service and it is usually expressed in seJ/J (Herendeen, 2004; Pulselli et al., 2008a). The calculations consider different energy qualities and take into account the losses of energy in the energy transformation processes. The emergy concept can be better visualized through the emergy flow diagram as conceived by Odum (Figure 1.3). *R* stands for the local renewable resources (e.g. solar energy, rain, wind, tide) which enter to the process economy, *N* for the local nonrenewable resources (e.g. fossil fuel), *F* for the input purchase from market (e.g. electricity, equipment, service), *Y* for the product to be sold to the market and *W* for the waste released from the system.

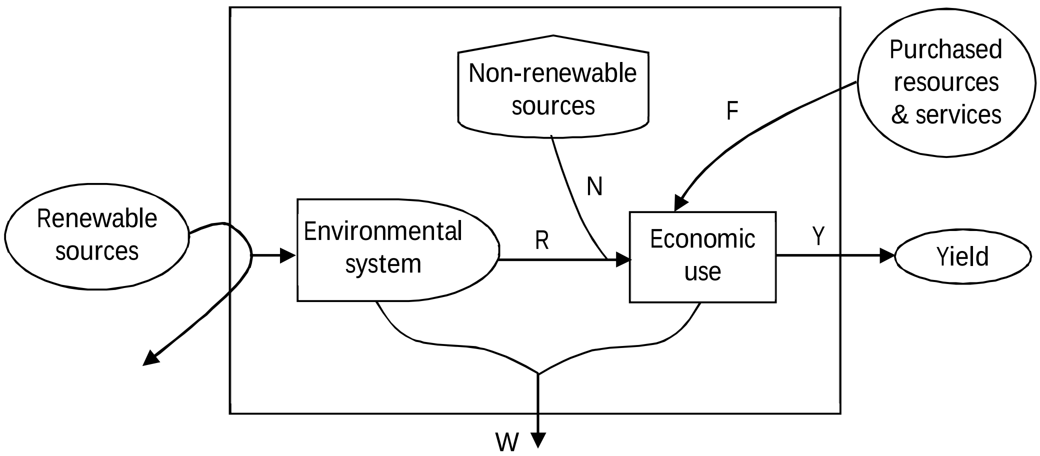

Figure 1.3. Emergy flow diagram (Lou et al., 2004).

Emergy has been applied as environmental indicator in different fields: electricity production systems (Brown and Ulgiati, 2002); comparison of horse and tractor traction (Rydberg and Jansén, 2002); evaluation of building materials (Pulselli et al., 2008a) and their recycling options (Brown and Buranakarn, 2003); evaluation of a building (Meillaud et al., 2005); evaluation of eco-industrial park with power plant (Wang et al., 2005); production, processing and export of coffee (Cuadra and Rydberg, 2006); solar salt production process (Laganis and Debeljak, 2006); hydrogen production systems from biomass and natural gas (Feng et al., 2009).

Some studies have accomplished combinations of energy, emergy or exergy analyses of a studied system, most of them in the biomass production field

(Nilsson, 1997; Hovelius, 1997; Hovelius and Hansson, 1999; Franzese et al., 2009).

1.3.1.2. Material flow indicators

Traditional material flow indicators relate to input and output flows within specific geographical or politic boundaries (countries, regions, etc.), e.g. Direct Material Input –DMI-, Physical Trade Balance –PTB- or Domestic Processed Output –DPO- (EUROSTAT, 2000). However, other indicators, such as rucksacks, Material Input Per unit Service (MIPS) or Substance Flow Analysis (SFA), can be more suitable to deal with the evaluation of products and production processes. Besides, the usefulness of Material Flow Analysis (MFA) in sustainable materials management has also been stated (Allen et al., 2009).

<u>Ecological Rucksack and MIPS</u>

The Ecological Rucksack (ER), term coined by F. Schmidt-Bleek in 1993 in the Wuppertal Institute (Spangenbergh, 2002), represents the sum of all materials which are not physically included in the economic output under consideration, but have been necessary for production, use, recycling and disposal (including those consumed indirectly). Thus, by definition, the ER is the life-cycle-wide material input minus the mass of the product itself (Schmidt-Bleek, 2001; Spangenbergh, 2002). Economic, social and technical innovation is advocated such that population needs are satisfied using less natural resources -reduction of at least a factor 10 as established in the Brundtland Report (WCED, 1987)-, at the same time that the value and utility of goods produced are improved. This relation between material input and service obtained as an output is called MIPS (Material Input per Unit Service) and introduces the idea of resource-efficiency (Hille, 1997). The reference to an output flow provides a standardized reference and allows comparisons among different yet functionally equivalent products (Spangenbergh, 2002). Thus, MIPS is a resource-efficiency measurement for the micro level that helps in the design of industrial products and in the planning of environmentally friendly processes, facilities and infrastructures (Adriaanse et al., 1997; Hertwich et al., 1997). Sinivuori and Saari (2006) applied MIPS to analyze the natural resource consumption in two university buildings. The methodology showed a good potential to point out the measures that should be adopted to reduce natural resource consumption during the different phases of a building life cycle (namely planning, construction and usage).

Substance Flow Analysis. Water Footprint

Substance Flow Analysis (SFA) focuses on specific substances, either within a region or from “cradle-to-grave”. Typical examples can include studies of nitrogen flows in a local area or flows of a specific metal in a regional scenario (Finvedenn and Moberg, 2005). Albeit, it has also been applied to assess industrial processes (Antikainen et al., 2004) and in the waste management field (Brunner and Ma, 2008).

The Water Footprint (WF) is one of the more recently developed indicators. It was introduced in 2002 in order to have a consumption based indicator of water use that could provide useful information in addition to the traditional production-sector-based indicators of water use (Hoekstra and Hung, 2002). Developed in analogy to the Ecological Footprint (EF), although not expressed in area units (see subsection 1.3.2), the WF of a nation was defined as the total volume of freshwater that is used to produce the goods and services consumed by the people of the nation. Thus, it could be considered as a particular case of SFA. Since not all goods consumed in one particular country are produced in that country, WF consists of two parts: use of domestic water resources and use of water outside the borders of the country (Hoekstra and Chapagain, 2007). In any sense, this term complements the EF and supplies one of its limitations given that water consumption is not properly accounted for in EF estimates. Thus, the WF can be estimated for a nation, business or individual by calculating the total water used during the production of goods and services. Industry related applications are, for example, the water footprint of worldwide cotton consumption (Chapagain et al., 2006), tomato production in Spain (Chapagain and Orr, 2009) and biofuel production (Dominguez-Faus et al., 2009).

1.3.2. Indicators with a territorial dimension

During the 90’s, a particular tendency to provide flows of natural resources with a territorial dimension came up. Within this context, concepts like the Ecological Footprint (EF) or Environmental Space (ES) emerged; nonetheless, there was a precedent to this idea (Hornborg, 2006). In 1965, Borgström had explained the apparent excess in own resources (particularly referring to food) appropriation by alluding to the fact that nations had drawn upon on an “invisible” carrying capacity (i.e., located elsewhere on the planet). In opposition to the “visible acreage” (farm and pasture land within the nation’s borders), this was named as “ghost acreage” and divided into two components: “trade acreage” (fraction that comes from net imports of food) and “fish acreage” (food obtained from the sea) (Borgström, 1965).

The EF and ES, as well as the Dissipation Area Index (DAI), are treated in further detail in next paragraphs. They were collected in chronological order in Table 1.1.

Table 1.1. Indicators with a territorial dimension in chronological order of appearance in literature.

Indicator	**Date**	**Author/s**	**Country**	**Reference**
Ghost Acreage	1965	G. Borgström	U.S.A.	Borgström, 1965
Environmental Space	1980’s	H. Siebert and J.B. Opschoor	The Netherlands	Opschoor and Reinders, 1991
Ecological Footprint	1990’s	W. Rees and M. Wackernagel	Canada	Rees, 1992; Rees and Wackernagel, 1996
Dissipation Area Index	1995	M. Narodoslawsky and C. Krotscheck	Austria	Narodoslawsky and Krotscheck, 1995

<u>Ecological Footprint</u>

The EF indicator was mainly founded on the carrying capacity concept, which refers to the number of individuals who can be supported in a given area within natural resource limits, and without degrading the natural social, cultural and economic environment for present and future generations (Kratena, 2008; CCN, 2010). Thus, the EF determines the space required to support an activity by means of the area needed to provide the resources consumed and to absorb the wastes generated (Rees and Wackernagel, 1996; Monfreda et al., 2004; Kitzes et al., 2007; Venetoulis and Talberth, 2008). Major land use types in ecological footprint accounting are: cropland, grazing land, fishing grounds, forest area, built-up land

and carbon land (Kitzes et al., 2007). Originally, the EF was advocated to assess the level of sustainability of the urban development, lifestyles or regions. Rees, who coined the term in the nineties together with Wackernagel, used the EF to call people's attention to the fact that urban regions are exceeding the use of territory geographically allocated for them (Rees, 1992; Rees and Wackernagel, 1996). The Global Footprint Network (GFN) publishes every year in the Living Planet Report a list of the calculated EFs, as well as the biocapacity, of a large number of countries and regions (GFN et al., 2008). Biocapacity is the capacity of ecosystems to produce useful biological materials and to absorb waste materials generated by humans using current management schemes and extraction technologies (Kitzes et al., 2007). A comparison between the EF and the Biocapacity reveals whether existing natural capital is sufficient to support consumption and production patterns (Wackernagel and Yount, 2000; Monfreda et al., 2004). The ecological deficit occurs when the EF exceeds the available biocapacity.

The European Union has showed particular interest in evaluating the EF capability to measure sustainable use of resources (ECOTEC, 2001; EUROSTAT, 2006; Best et al., 2008). Currently, there are also a wide range of applications in the environmental evaluation of production processes and products (Kratena, 2008; Limnios et al., 2009), like in aquaculture processes (Kautsky et al., 1997; Muir, 2005), a water supplier company (Lenzen et al., 2003), mobile phones (Frey et al., 2006) or wine production (Nicolucci et al., 2008).

The Human Appropriation of Net Primary Production (HANPP) is strongly related to EF since both methodologies recognize the significance of surface areas for ecological processes (Haberl et al., 2004). It can be expressed as a material, substance or energy flow. However, in spite of being an area related indicator, HANPP is not expressed in area units as the EF. Furthermore, HANPP is focused on the assessment of land use on national territory; thus, its application in the evaluation of production processes or products is not that clear as in the case of EF.

Environmental Space

The notion of ES was first introduced by Horst Sieber in 1982 (Bührs, 2007), although further developed by J.B. Opschoor in the early 1990s (Opschoor and Reinders, 1991). The concept reflects that at any given point in time, there are limits to the amount of environmental pressure that the Earth's ecosystems can handle without irreversible damage to these systems or to the life support processes that they enable. This limited space must be distributed among stocks of resources and sinks to absorb waste and pollution (Hille, 1997). Besides the need to respect the ecological limits, the ES is also based on the principle of fair distribution of resources (Spangenberg, 2002). For resources which are considered to be too environmentally damaging or risky (such as chlorine and nuclear power), environmental space is set at zero, implying a phase-out.

The ES is similar to, and overlaps with, the concepts of carrying capacity and ecological footprint. However, there are differences in the underlying methodology and also in the way results are expressed, since the notion of environmental space usually uses a range of indicators for different resources, in contrast to the single-scored ecological footprint. Besides, ES uses a range of indicators for different resources, and is not normally expressed in a single or composite indicator like the EF (Bührs, 2007).

The quantitative targets for the permitted use of environmental space must be made measurable with a standardized methodology, delivering meaningful, transparent and replicable information about the total material activated by the production, use and disposal or recycling of a certain product including all ecological rucksacks. The ES has been used in urban sustainability and policy guidance, rather than to evaluate the environmental performance of production processes (Mittler, 1999).

Dissipation Area Index

The DAI originates from the concept of assimilation capacity: a certain part or compartment of the ecosphere can absorb only limited output flows from the anthroposphere without suffering irreversible damage. Instead of estimating the output-flows of human activities that can be tolerated with a given assimilation capacity, the assimilation capacity that would be necessary to cope with given output flows is calculated (Eder and Narodoslawsky, 1999). Narodoslawsky and Krotscheck (1995) developed a method to estimate the dissipation areas of output-flows. Then, a Sustainable Process Index (SPI) is appraised as a result of aggregating all the areas implied in a process: material resources, energy, personnel, process installation (e.g. machines for the production process), product dissipation (assessment of the waste quality and quantity of different material and energy flows) and emissions (Narodoslawsky and Krotscheck, 1996; Stoeglehner and Narodoslawsky, 2009).

The relation between assimilation capacity and dissipation area is equivalent to that between the ecological footprint and the carrying capacity. However, an important difference is that the dissipation area index considers the absorption of certain kind of substances excluded from EF because they are considered unsustainable and not belonging to closed cycles in nature (e.g. heavy metals). For carbon dioxide a dissipation area is considered only if the emissions stem from fossil sources. The DAI can be disaggregated for the different key production sectors in a region. Thus, the production activities with the highest potential of contributing to a steering process towards sustainability can be identified (Eder and Narodoslawsky, 1999).

1.3.3. Indicators of Environmental Life-Cycle Assessment

This framework is based on a life cycle approach which considers the full supply chains of materials and energy. The conventional philosophy underlying in environmental life cycle approach refers to a cradle to grave framework, although in recent years a cradle to cradle perspective has been introduced (McDonough and Braungart, 2002); however, when analyzing particular systems or production processes, a specific-boundary approach can be defined and a gate to gate assessment carried out. The main phases in Life Cycle Assessment (LCA) studies are: 1) Goal and Scope definition; 2) Inventory analysis; 3) Impact assessment; 4) Interpretation (ISO, 2006). Indicators usually originate from the impact assessment phase (Guineé, 2001). Some of the impacts have a local effect on the environment (e.g., photochemical smog and eutrophication) while the others are of a more global nature (e.g., global warming and ozone depletion) (Azapagic and Perdan, 2000).

Apart from the above mentioned impact categories, which relate to a mid-point perspective in LCA -e.g. the methodology by the Institute of Environmental Sciences (CML) of the Leiden University (CML, 2000)-, other indicators correspond to a higher level of aggregation, like the Ecoindicator 99, oriented towards damage estimation. In this case, three types of environmental damages (endpoints), namely human health, ecosystem quality and resources, are weighted to obtain a final single score (ecoindicator) (Goedkoop and Spriensma, 2001).

LCA has largely been applied in the environmental appraisal of processes (Barton et al., 1996; Burgess and Brennan, 2001; Wood et al., 2006; Cherubini et al., 2009) and products (Milà et al., 1998; Nieminen et al., 2007; Roy et al., 2009). Styles et al. (2009) developed the Environmental Emission Index (EEI) based on LCA methodology. This index provides an integrated measure of the environmental significance of various emissions reported by industrial installations and sectors licensed under the EU Integrated Pollution Prevention and Control (IPPC) Directive (European Union, 2008).

Issues of global warming and greenhouse gas emissions are increasingly becoming one of the major technological as well as important societal and political challenges. Although several carbon-related indicators have emerged to this respect, the Carbon Footprint (CF) is the most popular and widely used to raise awareness on this environmental impact (Hoffmann and Busch, 2008). Next, the CF is described in further detail.

Carbon Footprint

The largest single contributor to climate change is carbon dioxide, although other greenhouse gases have higher global warming potential (IPCC, 2007). Hence, a CF measures the total set of greenhouse gas emissions caused directly and indirectly by an individual, event, organization or product and is expressed in carbon dioxide equivalents. The Carbon Footprint (CF) is measured in mass units. Therefore, in spite of the "footprint" term, the CF is equivalent to the global warming characterized category in LCA studies, and it does not measure land requirement as in the case of the EF. A further step of transformation from mass to area units is required. However, the difficulty and controversy arises when trying to identify an average assimilation rate for the different substances. Albeit, when biofuel systems are considered in energy planning, not only reduction of CO_2 should be considered but also land availability constraints, especially when agricultural resources need to be used for both food and energy production (Foo et al., 2008). In this sense, Stöglehner (2003) has also proposed a modified model of EF, which does not only account for energy savings but also for the substitution of fossil through renewable energy carriers, to be used for energy planning.

Business can use carbon footprints to inform their internal environmental management. Furthermore, carbon labels are a way to communicate a summary of the carbon footprints (which is strongly related to the supply chain) of a product to the final consumers (Edwards-Jones et al., 2009). Carbon Trust, a British not-for-profit company, was a pioneer in the development of a carbon label for products. They were also involved, alongside the British Department for Environment, Food and Rural Affairs (DEFRA) and the British Standard Institute (BSI), in the launching of PAS 2050:2008 Standards (Specification for the assessment of the life cycle greenhouse gas emissions of goods and services). Currently, there is also an ISO standard under development (ISO/WD 14067-1, Carbon footprint of products - Part 1: Quantification).

The CF is generally applied in energy-related studies (Johnson, 2008; Perry et al., 2008; Foo et al. 2008).

1.3.4. Indicators of Environmental Risk Assessment

Over the last decades there has been an exponential increase in the level of pollution and in the quantity of toxic substances released to the environment. This circumstance has awakened awareness about potential exposure to contaminants and a considerable activity in the field of Environmental Risk Assessment (ERA) has been going on. This has mainly taken place in international bodies such as the Organization for Economic Co-operation and Development (OECD) or the World Health Organization (WHO). In this context, the REACH deserves a special remark as a recent European Community Regulation on chemicals and their safe use (EC 1907/2006) which entered into force on 1 June 2007 (European Commission, 2006). It deals with the Registration, Evaluation, Authorization and Restriction of Chemical substances and aims at improving the protection of human health and the environment through the better and earlier identification of the intrinsic properties of chemical substances.

Historically, risk assessments have primarily focused on risks to human beings. It has gradually become apparent, however, that the ecological implications of large-scale environmental pollution should also receive attention (Van Leeuwen, 2007). ERA takes many different forms, depending on its intended scope and purpose, the available data and resources, and other factors. Hence, the scope and nature of risk assessments can range from national to site-specific findings concerning the same chemicals. Besides, some assessments are retrospective, focusing on injury after the fact, while others seek to predict possible future harm to human health or the environment (Patton, 1993).

ERA is a standardized process for the estimation of the magnitude, probability and uncertainty of adverse effects on health derived from the exposure to substances present in the environment (US EPA, 2009; ORNL, 2009). It is part of the more global risk management process, which consists on taking measures based on risk assessments and considerations of a legal, political, social, economic and engineering nature. The entire risk management process consists of eight steps, in which the former four correspond to the risk assessment phase, while steps 5 to 8 are in the domain of risk management (Van Leeuwen, 2007). Risk assessment comprises hazard identification, exposure assessment and risk characterization, as it is shown in Figure 1.4. From this, two relevant indicators (Hazard Quotient HQ and the Cancer Risk factor CR) can be obtained.

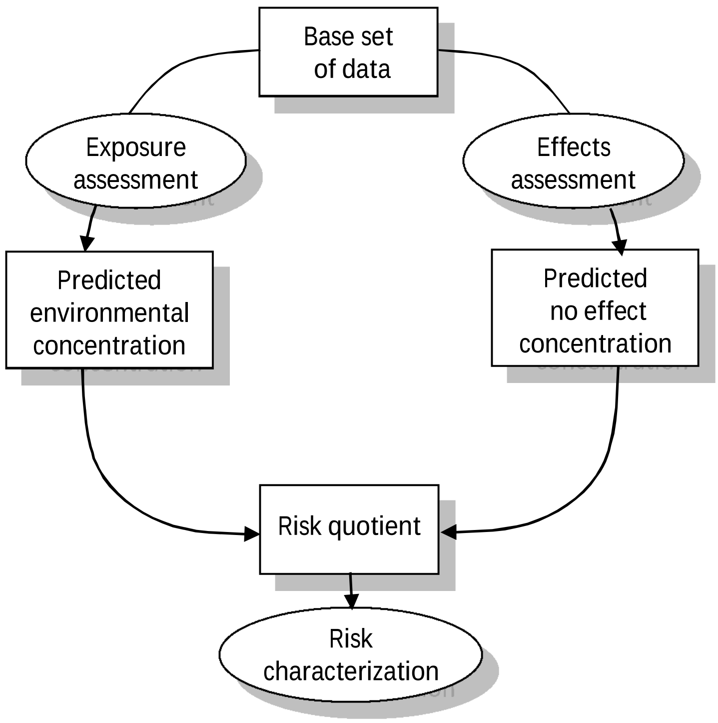

Figure 1.4. Risk characterization: a systematic procedure through estimation of exposure and effects (Van Leeuwen, 2007).

There are a great variety of models of a diversity degree of complexity for the assessment of distribution and exposition to hundreds of pollutants. These models are particularly useful to obtain a quick preliminary result providing information about the scenario. ChemCAN (Trent University, 2003), EUSES (European Union System for the Evaluation of Substances, European Commission, 2009), CalTOX (McKone and Enoch, 2002) and ACCHuman (Czub and McLachlan, 2004) are, among others, some of the most representative ones.

Risk assessment studies cover different areas, such as waste reuse scenarios (Franco et al., 2006; Muñoz et al., 2009); release of hazardous substances from products (Babich et al., 2004; Franco et al., 2007b) or occupational and home exposure to chemicals (Franco et al., 2007a; Ling and Hoang, 2000; Tsai et al., 2001; Hellweg et al., 2005).

1.4. Synergies and integration proposals

Material and energy flows can be considered as the basis on which all indicators are founded. They reflect the consumption of resources from nature and the emission of pollutants to the environment. These flows can be considered separately, certainly providing more detailed information, or aggregated, thus reducing the number of indicators to be handled. To this respect, bulk-MFAs are material flow analysis in which all materials flows are summed to generate single indicators of mass flow within an industrial economy. MIPS is bulk-MFA applied to a specific product or service, and could be considered as a simplified LCA in which the mass flows (including hidden flows) are used as an indicator of the environmental impact of a product or service (Kleijn, 2001).

Therefore, a strong link exists between EMFA and LCA, since inventories used for EMFA are generally based on a life-cycle perspective. However, EMFA fails at including all the information necessary to assess potential impacts on human health and the environment or energy and water consumption (Allen et al., 2009). Thus, EMFA and LCA indicators are not completely exchangeable but are likely to be integrated (Azapagic et al., 2007).

Many environmental issues are caused by or relate to the production, transformation and use of energy, e.g. ambient air quality, solid waste disposal, acid deposition, global climate change, etc. (Dincer, 2002). Thus, the minimization of energy flows is extremely important. Process Energy Analysis focuses on different processes and levels in the product life cycle and sums up the flows of energy use through each of the production process stages (Ness et al., 2007).

But attention must also be paid to the quality of energy. Thus, it has been observed that exergy exhibits a potential usefulness in addressing and solving environmental problems as well as attaining sustainable development. Increased efficiency can help to achieve energy security in an environmentally acceptable way by reducing the emissions that might otherwise occur (Dincer, 2002). Emergy has also found a good acceptance as environmental indicator, although, as other holistic approaches, it has encountered certain criticism mainly stem from the difficulty in obtaining details about the underlying computations (Hau and Bakshi, 2004b).

As it could be observed, the different kinds of available indicators can highlight different potential environmental problems, but none of them offers a comprehensive measure of the natural resources degradation effects originated as a consequence of anthropogenic activities. In some cases, the limited information provided by a single indicator can be assumed, while in other circumstances an integrative proposal needs to be considered. A commonly accepted feature of Integrated Assessments (IA) is that it is born from the joint contribution of several disciplinary fields, and that a useful IA should be able to cope with a plurality of perspectives on a particular issue (Gough et al., 1998).

The analysis presented next will focus on three complementary types of indicators, namely EF, LCA and ERA, representative of the different categories previously reviewed.

1.4.1. EF strengths and drawbacks

The EF provides valuable information about the degree of sustainability of a particular process since this indicator especially accounts for resources and energy consumption. Besides, it is especially helpful for communication purposes since it is a concept easy to be interpreted by the different stakeholders (Ferguson, 1999). However, some limitations were acknowledged for this methodology (Wackernagel and Yount, 2000; Kitzes et al., 2009), even though active development on EF methodology poses to continuous new proposals to overcome core critiques (Venetoulis and Talberth, 2008).

The EF does not capture most of the impact categories usually applied in life cycle analysis or it does not comprehensively take into account waste and emission flows. The argument is that for CO_2 emissions, a sufficiently sound method is available for calculating the land area required to absorb them, while this is not the case with other greenhouse gases. Actually, some greenhouse gases, such as hydrofluorocarbons, may be systematically excluded on the argument that these are synthetic gases for which no biological absorption rate can be defined (Kitzes and Wackernagel, 2009). In contrast, in the case of the Dissipation Area Index (DAI), considered as a modified version of EF, a list of relative factors is available for a variety of substances, apart from CO_2, for different compartments (Eder and Narodoslawsky, 1999). Moreover, the EF does not account for the depletion of non-renewable resources or does not reflect losses in biological diversity or environmental quality as such (Kitzes and Wackernagel, 2009), and the contribution of nuclear energy is still not clear (in fact, it is not included in National Footprint Accounts).

1.4.2. Benefits of a joint application of EF and LCA

Given the weaknesses stated for the EF, it results interesting to complement EF studies with certain LCA indicators. In addition to using LCA to assess aspects excluded from EF estimates (e.g. emissions different from CO_2 or hazardous wastes), its application was also proposed when a more in depth analysis was required for a particular functional unit, thus helping to identify more sustainable solutions or best available techniques (Azapagic, 1999). Besides, the existence of a relation between EF and LCA has been identified by Huijbregts et al. (2007) when the Ecoindicator (EI) 99 was employed to evaluate a large number of products and services consumed in the western economy. It was found that the majority of the products have an EF/EI ratio of around 30±5 m^2-eq yr/ecopoint, but deviations occurred when products with high mineral consumption and process-specific metal and dust emissions were evaluated. Therefore, even though the EF is recognized as a screening indicator for environmental performance, improvements in its methodology are recommended. EF is an indicator of minimum criteria for sustainability appraisal. This fact poses a drawback when different alternatives of processes or products are evaluated as it can lead to erroneous comparative analyses when a key environmental aspect is not taken into account by the EF methodology. This does not invalidate the application of EF, but the complementary use of other indicators would be recommended.

1.4.3. LCA and ERA: complementary approaches regarding toxicity

As it occurs in the case of EF and LCA, the LCA and ERA methodologies may benefit from a joint application in some cases. Their limitations when used alone and their complementary characteristics are exposed below.

1.4.3.1. LCA limitations

LCA is claimed to offer an integrative assessment of a process; however, the information provided regarding human and ecosystem toxicity, for example, is more incomplete than desirable. This means that it has a limited capacity to predict toxicity effects given that the fate of pollutants is usually not considered, so that the calculated impacts are potential rather than actual (Azapagic and Perdan, 2000).

1.4.3.2. Complementary characteristics of ERA

Risk assessment, on the other hand, provides an established methodology based on the assessment of different scenarios and events, distribution and transfer routs, exposure pathways, duration and frequency of the events that allows for a more rigorous and exhaustive evaluation. Nevertheless, assessments may need to integrate the risks from the entire life cycle of the chemical or product (Van Leeuwen, 2007). Therefore, LCA and ERA are methodologies that provide complementary indicators that can be integrated (Leet Socolof and Geibig, 2006).

1.4.3.3. Integration of LCA and ERA

Actually, motivated by the increasing release of pollutants in production processes, the European Union has carried out an integration of the risk and life-cycle assessment tools named USES (Uniform System for the Evaluation of Substances)-LCA (Huijbregts et al., 2000). As an example of its application, for the Human Toxicity category in the CML (Institute of Environmental Sciences of the Leiden University) 2 baseline 2000 life cycle impact assessment method, characterization factors, expressed as Human Toxicity Potentials (HTP), are calculated using USES-LCA, describing fate, exposure and effects of toxic substances for an infinite time horizon (CML, 2000). Bare (2006), however, aimed at coordinating the approach for conducting LCA and risk assessment using models consistent with the US Environmental Protection Agency's (US EPA's) handbooks, policies and guidelines. The author claims that, for certain impact categories, LCA can use many of the guidelines, methodologies and default parameters that have been developed for human health risk assessment, in conjunction with sensitivity and uncertainty analysis to determine the level of detail necessary for various applications. LCA can then identify "hot spots" that require the additional detail and level of certainty provided by ERA. Another interesting contribution regarding this matter has been proposed by Benetto et al. (2007). On the basis of mineral waste reuse scenarios, these authors deal with the problem of how to compare local scale impacts from ERA with local and global scale impacts arising from LCA. While ERA methods consider more realistic models, accounting for local specific conditions through an average or worst approach for a given time frame, in LCA the substances inventoried refer to the Functional Unit (FU) and encompass the whole life-cycle, using characterization models based on simplified relations or multimedia models referring to standard conditions. This would lead to consider that lifecycle toxic emissions are released instantaneously in a compartment of the multimedia model. Benetto et al. propose three different integration strategies: 1) Definition of new impact results, weighting the values for the local scale LCA impacts with a series of risk indexes (defined for ecotoxicity, acidification and eutrophication), whenever they are significant (i.e., higher than 1); 2) Two options are considered here: 2.1) use the cumulated emissions from ERA instead of LCI emissions and apply the standard LCA characterization models, and 2.2) Use a series of risk indexes instead of LCA impact results; 3) Definition of new impact categories based on ERA results, in addition to the existing LCA categories. Albeit, they recognize that these paths do not solve the problem of considering all the impact results simultaneously for decision making, and propose the use of multi-criteria methods as a possible solution to the problem.

1.4.4. Proposal for the integration of EF, LCA and ERA for the environmental assessment of production processes and products

Based on the strengths and weaknesses of the commented indicators (EF, LCA and ERA), a proposal for the integration of EF, LCA and ERA for the environmental evaluation of a production process is presented in Figure 1.5. The aspects for which it would be more appropriate to use one or another are signaled. Thus, the ecological footprint is more appropriate to examine the global environmental performance of a production process, providing a measure of its level of sustainability. However, other tools are preferable for more specific evaluations, like LCA to identify the operational units with major contribution to the environmental loads within the process, thus aiding the selection of Best Available Techniques (BATs).

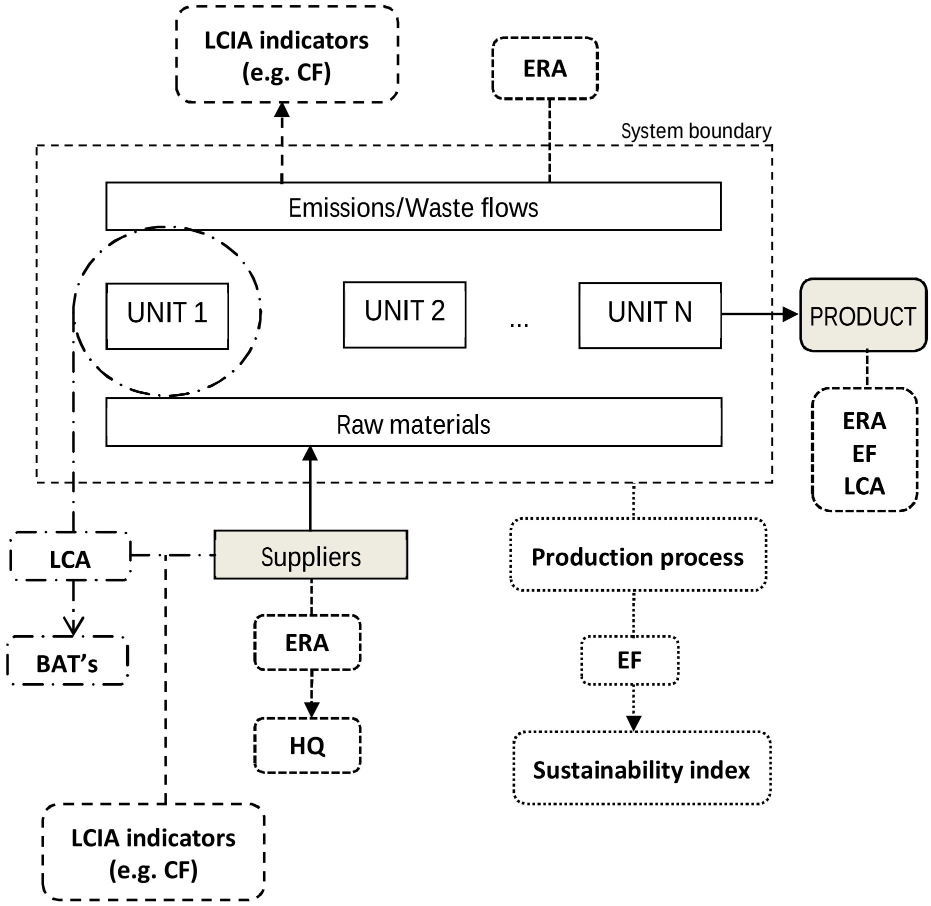

Figure 1.5. Scheme illustrating a proposal for the integration of EF, LCA, CF and ERA for the environmental evaluation of products and production processes.

The application of LCA would also be useful to assess the burdens linked to emissions released in the factory or the flows of wastes generated; hence, different life cycle assessment indicators can be derived (the case of the Carbon Footprint is highlighted in the figure). Given the relevance acquired by CF as environmental indicator, and given the transferring nature of this kind of burdens, it is interesting to encourage suppliers to declare their CF and hence to use it as a choice criterion. When evaluating the possible suppliers, it is also important to ensure that the risk associated to the raw materials provided do not imply a risk neither for the workers of the factory nor for the final consumers of the product; besides, the final disposal of the wastes generated in the process may pose a risk to the environment that should remain under safety levels. In this respect, ERA is the most suitable methodology to assess these aspects. Finally, the sustainability of the final product and its environmental impacts can be appraised using EF and LCA. All the information provided by these indicators at the different levels and stages should be considered together and interrelations taken into account in order to achieve the best option from an environmental point of view.

1.4.5. Summary of integration proposals from the literature

There exist several reviews of tools (pointing out their strengths and drawbacks) that have been used by industry, as well as proposals of indicators combinations made by different authors (Table 1.2). The use of several complementary indicators is advantageous and should be the preferred option to obtain different perspectives and a wider approach of the analysis carried out. Moreover, integration proposals can relate to a range of assessment contexts, ranging from the information sources themselves to the integration across environmental media (in regulation development, mainly). Other cases refer to vertical integration planning and management in large hierarchical institutions or integration of environmental concerns into governance (Scrase and Sheate, 2002). Sustainability itself implies the integration of environmental, economic, societal and institutional aspects.

Table 1.2. Reviews of tools and proposals of integration found in the literature.

Aim of the study	Tools	Reference
Modification of EF calculation	Ecological Footprint; Emergy Analysis.	Zhao et al., 2005
Review and characterization of tools	EMFA; Input-Output Analysis; Systems for economic and environmental accounts; Environmental Impact Assessment; Strategic Environmental Assessment; Environmental Management System; LCA; Cost-Benefit Analysis; EF; ERA.	Finnveden and Moberg, 2005
Evaluate the effectiveness of environmental indicators derived from three methodologies at farm level	Input-output accounting; Ecological Footprint Analysis; Life Cycle Assessment.	Thomassen and Boer, 2005
Comparison of EF and Ecoindicator 99	EF; LCA (end-point impact assessment method).	Huijbregts et al., 2007
Develop methodology of Integrated Environmental Assessment (IEA) of mineral waste recycling scenarios	Environmental Risk Assessment and Life Cycle Assessment.	Benetto et al. 2007
Environmental sustainability appraisal of regional systems, extracting complementary information	Emergy Evaluation; Ecological Footprint Analysis; Greenhouse Gas Inventory.	Marchettini et al., 2007
Mapping the flows of pollutants in the urban environment	Life Cycle Assessment; Substance Flow Analysis.	Azapagic et al., 2007
Construction of a decision support tool to analyze the performance of a plant	LCA, SFA, MCA, Technology Assessment, Sensitivity Analysis, Scenario Analysis, Cost-Effectiveness Analysis.	Neto, 2007
Evaluation of environmental sustainability at territorial level	EF; Greenhouse Gas Inventory; Extended Exergy Analysis; Emergy Evaluation; LCA; Remote Sensing.	Pulselli et al., 2008b
Environmental labeling of products	CF, EF, WF	Niccolucci et al., 2010

1.5. Methodologies of multi-criteria analysis

Composite indicators are considered as an innovative approach for evaluating sustainable development. However, uncertainty associated to imprecision of data, weighting schemes or aggregation methods must be taken into account (Singh et al., 2009). This kind of procedure is followed, for instance, in LCA, when apart from the compulsory characterization phase, normalization and weighting phases are also carried out. The single comparable index obtained can simplify the analysis and extraction of conclusions, but its subjective character derived from assumptions underlying in issues such as measurement of error in data, besides the choice of weights or aggregation system, cannot be neglected. A combination of uncertainty and sensitivity analysis can help to gauge the robustness of the composite indicator, and thus increase its transparency (Singh et al., 2009).

To facilitate IA, as those proposed in the previous section, a family of techniques referred to as Multi-Criteria Analysis (MCA) methods has been developed. MCA methods can be used to identify compromise solutions to complex policy and planning problems, as well as to avoid distortions and to manage all the information, criteria, uncertainties and importance of the criteria (Ladehlma et al., 2000; Greening and Bernow, 2004).

MCA can be defined as any structured approach to determine overall preferences among alternative options, where the options accomplish several objectives (EA-UK, 2003). MCA frameworks can vary from simple approaches requiring very little information to quite sophisticated methods based on mathematical programming techniques, requiring extensive information on each attribute and the preferences of the decision makers (Greening and Bernow, 2004). Sometimes, the variety of terms used to refer to MCA as well as the diversity of classifications of the available methods can result a bit confusing. A brief discussion on this matter is presented in subsection 1.5.1., and the classification adopted in this Thesis established (Figure 1.6).

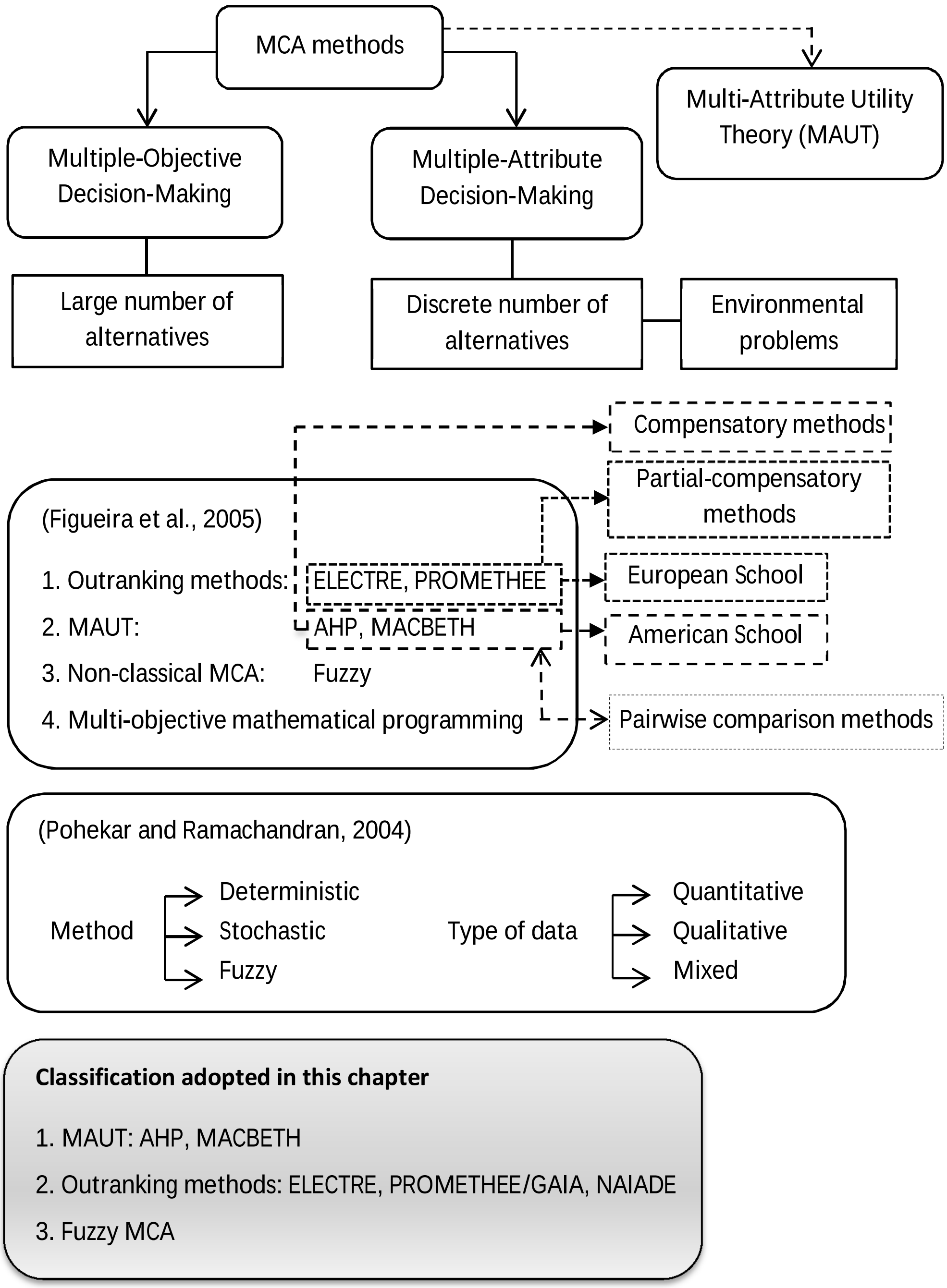

Figure 1.6. Scheme illustrating the diversity of classifications of MCA methods.

1.5.1. MCA methods classification

One of the most extended approaches first differentiates between Multiple-Objective Decision-Making (MODM) and Multiple-Attribute Decision-Making (MADM) methods (Pohekar and Ramachandran, 2004; Sadok et al., 2008). The former are used in problems with an infinite (continuous) or large number of alternatives; meanwhile, the latter are designed for situations with a limited (discrete) number of alternatives. Some authors refer to Multi-Attribute Utility Theory (MAUT) separately from multi-criteria decision support systems (Dyer et al., 1992; Korhonen et al. 1992), but it can also be found classified as a MCA method (Pohekar and Ramachandran, 2004; Figueira et al., 2005; Hajkowicz and Collins, 2007; Sadok et al., 2008), which will be the criterion followed in this Thesis. Environmental problems are usually described by a finite set of alternatives (Korhonen et al. 1992). Therefore, this section will only refer to MADM methods.

It is also frequent to hear about the American and the European Schools. The former is responsible for methods like MAUT and Analytical Hierarchy Process (AHP), while major contribution from the latter refer to the outranking methods ELECTRE and PROMETHEE.

The following list of MCA methods was suggested by the Institute for Environmental Studies (The Netherlands) in the methodology report of the Sustainability A-test project (Ridder, 2004): 1) Non-compensatory methods; 2) Multi-attribute utility theory; 3) Linear additive models; 4) The analytical hierarchy process; 5) Outranking methods; 6) Fuzzy MCA. The distinction between the compensatory and non-compensatory methods is a matter of the decision rule used. MAUT, AHP and weighted summation are examples of compensatory methods, while outranking methods such as PROMETHEE and NAIADE are partial-compensatory.

Pohekar and Ramachandran (2004) present a series of MCA methods in their work (AHP, ELECTRE, PROMETHEE, etc.) but they do not exactly group them into categories. Nonetheless, they contemplate another possible classification referring to deterministic, stochastic and fuzzy methods. Depending on the type of data employed, the MCA methods can also be classified as quantitative, qualitative or mixed.

The review by Figueira et al. (2005) establishes four main groups: outranking methods, MAUT, non-classical MCA approaches and multi-objective mathematical programming. Figueira et al. (2005), as well as Lahdelma et al. (2000) or Sadok et al. (2008), classify AHP under the MAUT heading; however, other authors present AHP separately (Pohekar and Ramachandran, 2004). In contrast, Hajkowcz and Collins (2007) define AHP and MACBETH as pairwise comparison methods. Actually, as such, they can be used for eliciting weights to be applied in other MCA methods (Lahdelma et al., 2000).

Hajkowcz and Collins (2007) contemplate the following MCA method categories: 1) Multi-criteria value functions; 2) Outranking approaches; 3) Distance to ideal point methods; 4) Pairwise comparisons; 5) Fuzzy set analysis; 6) Tailored methods.

Lahdelma et al. (2000) refer to MAUT and outranking methods as the two categories into which the main approaches can be classified based on the type of decision model they apply. A similar classification is proposed by Sadok et al. (2008), just adding an extra category for mixed or non-classical methods, which includes decision rule-based approaches, generally referred to as expert systems.

For simplicity and clarity, and in accordance to these latter proposals, the following categories have been considered in this review: MAUT (AHP, MACBETH); Outranking methods (ELECTRE, PROMETHEE/GAIA, NAIADE) and Fuzzy MCA. These were selected for their relevance in the environmental field. A brief description of each of these methods and a review of applications in decision making in the environmental field were conducted in subsections 1.5.2 to 1.5.4.

1.5.2. Multi-attribute utility theory

The methods classified within this category were AHP and MACBETH, presented in sections 1.5.2.1 and 1.5.2.2, respectively.

1.5.2.1. AHP

AHP, a leading methodology to solve decision problems by the prioritization of alternatives pioneered by Saaty (1980), has been used in almost all the applications related with decision-making (planning, selecting a best alternative, resource allocation, resolving conflict, optimization, etc.). The problem is structured in a hierarchy of different levels constituting goal, criteria, sub-criteria and alternatives (Vaidya and Kumar, 2006). Elements at given hierarchy level are compared in pairs to assess their relative preference with respect to each of the elements at the next higher level. The intensity of preference between two elements is established on the basis of the Saaty's scale of 1-9 (Pohekar and Ramachandran, 2004). Guiqin et al. (2009) recently applied the AHP for the selection of an appropriate solid waste landfill site based on environmental and economic factors. Huang and Ma (2004) built a comprehensive framework for the environmental evaluation of packaging materials combining the approach provided by LCA as quantitative method, AHP as qualitative method and cluster analysis to integrate the results of the former two. AHP has also been used to support a last aggregating step in LCA impact assessment, so that a single final score was obtained (Pineda-Henson et al., 2002; Hermann et al., 2007).

1.5.2.2. MACBETH

The main drawbacks of the AHP method are potential internal inconsistency and the questionable theoretical basis of the 1-9 scale. Alternative methods, such as MACBETH, have been developed to overcome some of these objections. MACBETH (Measuring Attractiveness by a Categorical Based Evaluation TecHnique), introduced by Bana e Costa and Vansnick (1994), is a multi-criteria decision analysis approach that requires only qualitative judgments about differences of value to help an individual or a group to quantify the relative attractiveness of options. MACBETH uses a simple question-answer protocol that involves only two options in each question, so that a pair-wise comparison is required. The difference of attractiveness is measured in terms of seven semantic categories ranging from null to extreme (Clivillé et al., 2007). This multi-criteria method is implemented in the software M-MACBETH (Bana e Costa et al., 2005).

Bana e Costa et al. (2004) used the MACBETH approach to evaluate flood control options for the catchment of Livramento creek in the peninsula of Setúbal, in Portugal. With that purpose, indicators from the environmental, social, and technical dimensions were considered to guide the evaluation and the comparison of the alternatives in order to select the most adequate option.

1.5.3. Outranking methods

The outranking family methods considered, according to the classification adopted in this work established in section 1.5.1, were ELECTRE, PROMETHEE/GAIA and NAIADE. They are described next.

1.5.3.1. ELECTRE

ELECTRE is a family of outranking multi-criteria decision analysis methods that originated in Europe in the mid-1960s (Roy, 1991). The acronym ELECTRE stands for ELimination Et Choix Traduisant la REalité (ELimination and Choice Expressing REality). Among the different ELECTRE versions, ELECTRE III is particularly suited to the environmental appraisal of complex engineering projects (Rogers and Bruen, 1998). The main improvement in ELECTRE III with regard to the previous methods is that outranking relations can be interpreted as fuzzy relations. ELECTRE III is a method used to rank problems using binary outranking relations (Rousat et al., 2009). The construction of these outranking relations is based on two major concepts as in other ELECTRE methods: 1) Concordance: Alternative *a* outranks alternative *b* if a sufficient majority of criteria are in favor of alternative *a*; 2) Non-Discordance: When the concordance condition holds, none of the criteria in the minority should be opposed too strongly to the outranking of *b* by *a*. Rousat et al. (2009) chose the ELECTRE III multi-criteria analysis method to aid the comparison of different strategies of demolition waste management combining eight sustainability criteria. Also in the waste management field, this technique has been applied for the adequate localization of an incinerator and a waste disposal plant (Franca Norese, 2006). Papadopoulus and Karagiannidis (2008), however, applied it for the selection of the best alternative for electricity supply of isolated systems, taking into account the technical, financial, environmental and social implications. The ability of these techniques to deal with a number of criteria at a time was checked. At the same time, the influence of using one or another in the final ranking obtained and, therefore, in the quality of the final decision, was explored and assessed.

1.5.3.2. PROMETHEE/GAIA

PROMETHEE (Preference Ranking Organization Method for Enrichment Evaluation) is a non-parametric outranking method for a finite set of alternatives and is based on positive (out -) and negative (in -) preference flows for each alternative in the valued outranking relation to rank the alternatives according to the selected preferences (weights). A positive flow expresses how much the specific alternative is dominating other alternatives, and a negative flow expresses how much that alternative is dominated by the others. The PROMETHEE method is one of the most recent MCDA methods that was initially developed by Brans in 1982 and further extended together with Vincke (Brans and Vincke, 1985). Meanwhile, GAIA (Geometrical Analysis for Interactive Aid) is a visualization method which complements the PROMETHEE ranking method. Based on the PROMETHEE II method, GAIA provides a visual guidance for the principal criteria that are used for ranking of the alternatives. The PROMETHEE methods have been applied in different areas, ranging from environmental management to manufacturing, logistics, energy management or social topics; actually, environmental management is considered as the most popular topic in PROMETHEE applications (Behzadian et al., 2010). PROMETHEE has been applied for waste management planning (Vego et al., 2008) or to appraise different renewable energy scenarios against a number of sustainability criteria, both at national and local level (Kowalski et al., 2009).

1.5.3.3. NAIADE

The NAIADE (Novel Approach to Imprecise Assessment and Decision Environments) can be classified as an outranking method using a pair-wise comparison technique for ranking scenarios with an emphasis on imprecise inputs and multi-stakeholder settings. It uses a criteria/scenarios matrix and allows for a range of values from precise, stochastic, or fuzzy numbers or linguistic expressions. It has been applied for the evaluation of energetic scenarios (Dinca et al., 2007) or to assess the sustainability of bioenergy systems (Buchholz et al., 2009). Regarding the uncertainties arising from data, models and practitioner's choices in impact assessment and interpretation phases in LCA studies, Benetto et al. (2008) applied a modified version of the NAIADE multi-criteria method to support the interpretation of LCA results including uncertainty evaluations.

1.5.4. Fuzzy MCA

The application of multi-criteria decision-making methods requires processing imprecise, uncertain, qualitative or vague data. Fuzzy Logic is one of the most common methodologies used to address uncertainty matters (Bellman and Zadeh, 1970). The use of fuzzy logic techniques (Zadeh, 1965) allows to obtain a quantitative approach using a qualitative representation; thus, it is able to simultaneously handle numerical data and linguistic knowledge (Carrasco et al., 2002). Phillis and Andriantiatsaholiniaina (2001) proposed the use of fuzzy logic to deal with the vaguely defined sustainability concept. Thus, they developed a model called Sustainability Assessment by Fuzzy Evaluation (SAFE) in which ecological and human inputs were treated individually and then combined with the aid of fuzzy logic to provide an overall measure. In a subsequent work they performed sensitivity analysis of the SAFE model to indentify the most important factors contributing to sustainable development (Adriantiatsaholiniaina et al., 2004). Some applications are also related to the combination of the fuzzy theory and ERA, as in the work by Li et al. (2007). These authors developed an integrated fuzzy-stochastic risk assessment (IFSRA) approach to systematically quantify both probabilistic and fuzzy uncertainties associated with site conditions, environmental guidelines, and health impact criteria for risk assessment of groundwater contamination. The contaminant of interest of the study was xylene and the risk derived from its ingestion was examined.

1.5.5. Joint use of MCA methods

Fuzzy Logic is also one of the methodologies more frequently combined with AHP; as an example of application in the environmental field, Ocampo-Duque et al. (2006) developed a comprehensive multi-attribute decision-aiding method based on the AHP to estimate the relative importance of water quality variables. In this work, AHP and fuzzy reasoning are combined to estimate a Fuzzy Water Quality (FWQ) index. Similarly, Sadiq and Hussain (2005) proposed the use of a fuzzy-based methodology and a three-stage hierarchical structure for estimating aggregative risk of various environmental activities, pollution sources and routes in a given process. The developed methodology was applied to a case study of offshore drilling waste for evaluating various discharge scenarios.

A number of favorable characteristics of the AHP method could also enhance the ranking MCA methodology PROMETHEE, namely at structuring the decision problem and at the determination of weights (Babic and Plazibat, 1998; Macharis et al., 2004).

Thus, the flexibility of AHP facilitates its integration with different complementary methodologies for a better consideration of the uncertainty and vagueness in the decision process; hence, this enables the user to extract benefits from all the combined methods, and consequently achieve the desired goal in a better way.

1.5.6. Summary of applications of MCA methods in the environmental field

All the applications previously related have been summarized in Table 1.3. As in the case of indicators selection, the wide variety of existing decision aid methods can pose a problem when choosing among them. For a real-life problem, different methods may provide different results with the same data. Besides, there is usually no means to objectively identify the best alternative or method (Lahdelma et al., 2000).

Consequently, it is advisable to bear in mind the fundamentals of each technique and the appropriateness of using one or another depending on the case-study and the aim pursued. Thus, applying an outranking method may provide a particular prioritization of alternatives; meanwhile, AHP or Fuzzy techniques may lead to classify alternatives in a different order. The expert knowledge is essential to succeed in the use of these methods.

Table 1.3. Application of MCA methodologies to integrate environmental indicators and aid decision-making.

MCA method	**Aim of the study**	**Reference**
AHP	Comprehensive framework for the environmental evaluation of packaging materials combining LCA, AHP and cluster analysis.	Huang and Ma, 2004
	Aggregate LCA impact categories into a single score. Application for the environmental evaluation for the pulp and paper industries.	Pineda-Henson et al., 2002; Hermann et al., 2007.
	Selection of appropriate solid waste landfill site based on environmental and economic factors.	Guiqin et al., 2009
MACBETH	Evaluate flood control options for the catchment of Livramento creek in the peninsula of Setúbal, in Portugal.	Bana e Costa et al., 2004
ELECTRE III	Decision-making on the localization of waste-treatment plants.	Franca Norese, 2006
	Selection of the best alternative for electricity supply of isolated systems.	Papadopoulus and Karagiannidis, 2008
	Evaluate strategies of demolition waste management.	Rousat et al., 2009
PROMETHEE + GAIA	Waste management planning.	Vego et al., 2008
PROMETHEE	Evaluate local and national renewable energy scenarios in Austria. Includes a review of MCA applied to energy issues.	Kowalski et al., 2009
NAIADE	Select the optimal energetic scenario from seven options based on natural gas, using LCA to assess the environmental aspects.	Dinca et al., 2007
	Support LCA results interpretation including uncertainty evaluation.	Benetto et al., 2008
	Sustainability evaluation of bioenergy systems.	Buchholz et al., 2009

Table 1.3 *(cont.)*. Application of MCA methodologies to integrate environmental indicators and aid decision-making.

MCA method	**Aim of the study**	**Reference**
Fuzzy Logic	SAFE model. Combine ecological and human inputs.	Phillis and Andriantiatsaholiniaina, 2001
	SAFE model + sensitivity analysis. Identify most important factors in sustainable development.	Andriantiatsaholiniaina et al., 2004
	An integrated fuzzy-stochastic modeling approach for risk assessment of groundwater contamination.	Li et al., 2007
AHP + Fuzzy Logic	Model for estimating aggregative risk of various environmental activities, pollution sources and routes in a given process.	Sadiq and Hussain, 2005
	Obtain a Fuzzy Water Quality index.	Ocampo-Duque et al., 2006

1.6. Conclusions

In this chapter, indicators of different nature have been reviewed under a process- and product- oriented approach, from those with a territorial dimension to the more generic material and energy flows, life-cycle or risk assessment indicators. The importance and usefulness of each of them have been highlighted, as well as the similarities among them and complementary characteristics.

The different kinds of available indicators can measure different potential environmental problems; consequently, their application can be more appropriate or of particular interest depending on the aim of the study or the characteristics of the case study. Thus, it is particularly important to be aware of the existing indicators and what they measure, in order to draw a map as complete as possible that would act as a reference for stakeholders.

Albeit, none of the analyzed environmental indicators can offer a comprehensive measure of the effects of natural resources degradation, consequence of anthropogenic activities. In some cases, the limited information provided by a single indicator can be assumed, while in other circumstances an integrative proposal needs to be considered.

Integrative frameworks enable the achievement of more comprehensive assessments. In this respect, MCA provides a family of flexible analytical tools that can effectively support decision making with regard to complex sustainability issues. They can be used to integrate the individual indicators figures into a global single score, or to outrank a series of operational options considering all the information at a time.

The application of MCA methods has successfully been explored in a number of fields like energy planning, resource (fisheries, forestry, water and land) management and waste management (Hajkowicz and Collins, 2007). However, as the method employed may affect the result obtained, it is important to take the time to select the most appropriate one for the purpose of the study, so that the choice can be justified.

Moreover, MCA methodologies imply the weighting of the criteria considered in the study. Therefore, they provide a means for giving relative importance to the different indicators of sustainability, which will depend on the conditions at the moment of the study, seeking to equilibrate the scales.

1.7. Perspectives

The integrative approach must not lead to discard research on the improvement of existing methodologies. Indicators like the EF with a recognized potential for measuring sustainability must be enhanced, and efforts to eliminate weaknesses carried out. Some of the shortcomings are detected when new fields of application are explored. Hence, the development of methodologies to incorporate aspects traditionally excluded from EF accounts, like emissions different from CO_2 or hazardous wastes, is needed. This would improve the application of this indicator in the environmental evaluation of products and processes. Both in the case of EF and LCA, future research should deal with the development of local and regional conversion factors, taking into account the implications of considering these instead of the global ones. Toxicity related categories in LCA could also be improved, thus enhancing its application as screening indicators before deciding to conduct an ERA. In all cases, the incorporation of more substances to databases is also required as new toxicology studies are carried out.

Finally, additional research is needed to acquire further knowledge and understanding of different types of uncertainty inherent in environmental decision-making, and how they affect the quality of decisions when MCA is applied (Ascough et al., 2008).

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Chapter 2

Materials and Methods

Chapter 2

Materials and Methods

Abstract

This chapter provides a general description of the materials and methods employed during the development of this thesis. The chapter has been divided into three main sections. The first one includes the methodologies regarding the environmental evaluation of products and processes, namely Ecological Footprint (EF), Life Cycle Assessment (LCA), Environmental Risk Assessment (ERA) and Energy and Material Flow Analysis (EMFA). The second part of the chapter deals with the multi-criteria analysis, describing those methodologies that were selected to be applied in the different chapters: Analytic Hierarchy Process (AHP), Fuzzy Logic techniques, ELECTRE and PROMETHEE/GAIA. The third section collects those statistical analysis tools that were applied to support, complement and give consistency to the different estimates conducted, namely sensitivity analysis and statistical correlations.

The materials mostly refer to software tools and database sources. A number of software tools were studied and applied during the development of the thesis, all of them linked to a specific tool from the three sections established in this chapter. Hence, SIMAPRO® was employed to conduct LCA studies, Umberto® for EMFA, Matlab® and specifically the Fuzzy Logic Toolbox for multi-criteria analysis, Decision Lab for PROMETHEE/GAIA, Crystal Ball® for sensitivity analysis and MS Excel® and SPSS® for general statistical analyses. The versions of the software packages were not indicated here since they varied along the period when the work was carried out. MS Excel® was also used to implement the simplified tools derived from the adaptation of EF, LCA and ERA to the production processes studied. Regarding databases, several sources were consulted to collect all the parameters necessary to conduct the analysis proposed. These were more specifically detailed in the chapters where they were used.

2.1. Environmental evaluation methodologies

The feasibility of a good environmental evaluation of a production process or product is conditioned by the availability of a set of methodologies that can properly account for the environmental loads associated, from which different indicators can be derived. In this first section of chapter 2, the methodologies employed in the thesis are described, indicating the software packages used in each case.

2.1.1. Ecological Footprint

The Ecological Footprint (EF) determines the space required to support an activity by means of the area needed to provide the resources consumed and to absorb the wastes generated (Rees and Wackernagel, 1996; Monfreda et al., 2004; Kitzes et al., 2007). Major land use types (Figure 2.1) in ecological footprint accounting are: cropland (crops for food, animal feed, fiber, oil, etc.); grazing land (raising animals for meat, hides, wool, milk, etc.); fishing grounds (harvesting fish and other marine products); forest area (harvesting timber products and fuelwood); and built-up land (infrastructure for housing, transportation, industrial production, etc.). Additionally, there is a component accounting for the carbon land, i.e., the area required to absorb carbon dioxide emissions derived from energy consumption. Many different ecosystem types have the capacity for long-term storage of CO_2, such as cropland or grassland. However, since most terrestrial carbon uptake in the biosphere occurs in forests, and to avoid overestimations, carbon uptake land is assumed to be forest land. For this reason, forest for timber and fuelwood is currently not separated from forest for carbon uptake (Ewing et al., 2010). There is also a 12% that must be reserved for biodiversity, as stated in the Brundtland Report (WCED, 1987).

Demand for resource production and waste assimilation are translated into global hectares by dividing the total amount of a resource consumed (or waste generated) by the global average yield of the land type that produces that resource or absorbs that waste. This area is multiplied by the appropriate equivalence factor to express the total demand in global hectares for each resource. Yields are calculated based on various international statistics, primarily those from the United Nations Food and Agriculture Organization (Kitzes et al., 2007).

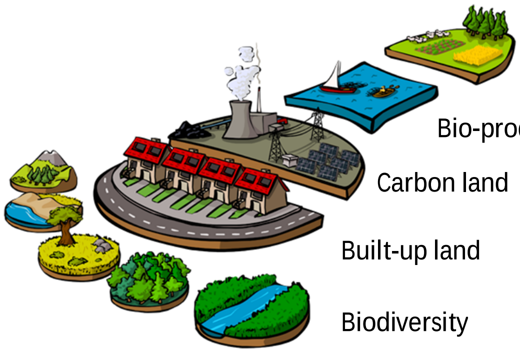


Figure 2.1. Scheme showing the different components of the EF.

Two EF approaches can be distinguished: compound and component (Chambers et al., 2000; Monfreda et al., 2004). While the compound method is based on national statistics of input-output flows (production, import, export), the component method is based on life-cycle data for each individual component involved in calculations. Given the system boundary defined in the evaluation of a production process or a product, the component method was selected in this thesis. Thus, individual EFs are calculated for each material and energy flow in the inventory data, and then they are aggregated to estimate the total EF of the process.

Besides, two perspectives, namely additive or mutually exclusive use of land, can be adopted. The former means that the same area can be used for different purposes at a time (e.g., the forest used to harvest timber can be the same that the one used for carbon assimilation) thus leading to lower EF values, while the later implies the opposite situation, considered as a more precautionary approach (Monfreda et al., 2004; Venetoulis and Talberth, 2008). The mutually exclusive approach was adopted in this thesis.

The EF is commonly expressed in units of global hectares (gha). A global hectare is a hectare that is normalized to have the world average productivity of all biologically productive land and water in a given year (Kitzes et al., 2007).

Mathematically, the steps required for the transformation of input-output flows (V_i) into a single score are expressed in equations [1-5], as adapted to be used for the evaluation of production processes in this thesis. Flows were classified into three main categories (k): energy consumption, resources consumption and waste generation. The sub-categories included in each case depend on the process studied and will be indicated in detail in the correspondent chapters.

It must be noticed that electricity is not a primary energy source that can be directly obtained from nature; consequently, it has to be broken down according to the power supplier company's rates (which may vary in the course of time) in order to distinguish the primary sources of energy. On the other hand, assessing the EF associated to the production of goods grown in land requires investigating its natural productivity, by which the flow must be divided to be translated into area units. However, when discussing about manufactured materials, the embodied energy (energy used during a product's entire life cycle in order to manufacture, transport, use and dispose of the product) must also be taken into account. This is attributed to fossil fuel consumption.

There are three different approaches to calculate the footprint of fossil fuel consumption (Rees and Wackernagel, 1996; Holmberg et al., 1999). Each of them has a sustainability basis and thus provide with similar results. The first one would be to account for the corresponding area needed for the sustainable production of bio-fuels, such as methane or ethanol, built on closed carbon cycles. A second method calculates the area needed to compensate only the biochemical energy from different combustion fossil sources, without taking into account that the biochemical energy of woods has not the same technical quality as fossil fuel or bio-fuels. Meanwhile, the third method is based on carbon dioxide sequestration, according to which the area is calculated by assessing the extension of newly planted forest required for sequestering the CO_2 released by the combustion of fossil fuel. When carrying out the calculation of the EF, it is important not to exaggerate the final outcome. For this reason, the third method, which leads to the smallest footprints for fossil fuel use, is the most frequently selected and was the one adopted in this thesis.

The basic calculation for each component, generally applied on a yearly basis, is then carried out as shown in equation [2.1].

$$A_{ik} = \sum_j \frac{V_i}{NP_i} F_j + \sum_j \frac{EV_i}{EP_i} F_j \qquad [2.1]$$

Where A_{ik} is the area required for the component *i* belonging to the main category *k* namely energy (*E*), resources (*R*) or waste (*W*); NP_i, EV_i and EP_i are the natural productivity, embodied energy and energy productivity (associated to fossil fuel) for element *i*; F_j is the equivalence factor for land type *j*. Equivalence factors translate a specific land type (i.e. cropland, pasture, forest, fishing ground) into a universal unit of biologically productive area, generally a global hectare (Kitzes et al., 2007). Thus, in contrast to other composite indicators that sum heterogeneous subcomponents using weights that are based on expert opinion, the EF assigns empirically based weighting coefficients to individual land types based on data on the relative productivity of these different area types (Kitzes and Wackernagel, 2009).

First EF estimates conducted during the thesis estimated the EF of wastes as a particular case of equation [2.1], where energy and material savings occur as a consequence of the incorporation of a certain rate of recycling, since the extraction and manufacture of new materials is avoided.

The area required to provide the resources and energy categories, and to absorb the wastes category, is then calculated following equation [2.2].

$$A_k = \sum_i A_{ik} \qquad [2.2]$$

Where A_{ik} are the single elements included in the main category *k*. Finally, the EF of the process due to the annual performance is expressed by equation [2.3].

$$EF = \sum_k A_k \qquad [2.3]$$

It is also interesting, especially to enable evolution tracking of the performance of the plant, as well as benchmarking against other factories, to express the EF as a relative indicator. In this case, the relative ecological footprint (EF_r) was constructed referring the total EF to the production rates (P_{yr}), as indicated in equation [2.4], being its units gha (or gm^2 depending on the magnitude of the figure) per item produced.

$$EF_r = \frac{EF}{P_{yr}} \qquad [2.4]$$

Finally, an additional concept must be considered: the Net Ecological Footprint (*NEF*). Until now, only those aspects referred to land consumption have been discussed. However, the opposite idea of Counter Footprint (*CF*) must be taken into account, since it represents the available hectares of land. Thus, the NEF can be calculated following equation [2.5].

$$NEF = EF - CF \qquad [2.5]$$

Consequently, a good way to diminish the net impact in the environment is to invest in natural capital protection (forest, pasture land, marine reserve, etc.) thus increasing the *CF* value.

The model for the estimate of the EF was implemented in a spreadsheet in MS Excel®. The factors required for transformations within the EF methodology have suffered from updates during the accomplishment of this thesis; consequently, the parameters used in each case are indicated in the respective chapter.

2.1.2. Life Cycle Assessment

Life Cycle Assessment (LCA) is a tool to assess the potential environmental impacts and resources used throughout a product's lifecycle (Figure 2.2), i.e., from raw material LCA acquisition, via production and use phases, to waste management (ISO, 2006a/b). LCA is a complex process that requires considerable time and data input; however, its comprehensive scope offers the advantage of avoiding problem-shifting, for example, from one phase of the life-cycle to another, from one region to another, or from one environmental problem to another (Hermann et al., 2007; Finnveden et al., 2009).

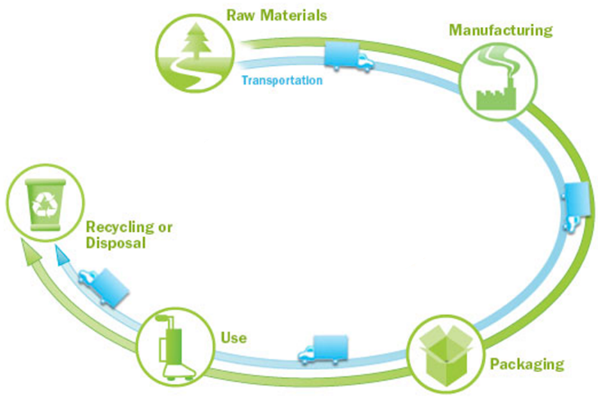


Figure 2.2. Scheme of the life cycle of a product.

The structure defined in the ISO 14040 series standards (ISO, 2006a/b) has been followed in the present study (Figure 2.3). Accordingly, the methodology comprises four main steps: 1) Goal and scope definition; 2) Inventory analysis; 3) Impact assessment; 4) Interpretation.

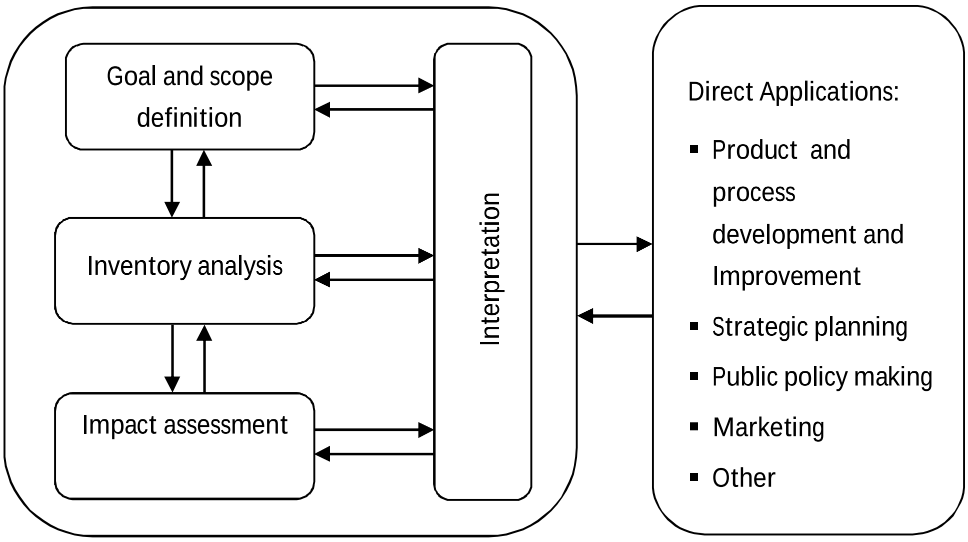


Figure 2.3. Main steps and applications of LCA (adapted from ISO, 2006a)

The goal and scope definition includes the reasons for carrying out the study, the system boundaries and the functional unit. The inventory analysis involves data collection and calculation of inputs and outputs of the product system in relation to the functional unit. Life Cycle Impact Assessment (LCIA) aims at evaluating the magnitude and significance of the potential environmental impacts of the studied system and consists of both mandatory and optional elements (Pennington et al., 2004; ISO, 2006a). Hence, according to the ISO standard on LCA, selection of impact categories, classification and characterization are mandatory steps in LCIA, while normalization and weighting are optional. Selection of impact categories and classification involves identification of the categories of environmental impacts which is of relevance to the study. During characterization, the impact of each emission is quantified according to the characterization model selected (mid-point or end-point) and expressed as an impact score in a unit common to all contributions within the impact category (e.g., kg CO_2-equivalents for greenhouse gases contributing to the impact category climate change) applying the concept of characterization factors which for each substance expresses its potential impact in terms of the common unit of the category indicator (equation [2.6]).

$$C_t = \sum_s C_{st} = \sum_s A_s \cdot W_{st} \qquad [2.6]$$

Where A_s is the amount of emission s released, W_{st} is the characterization factor for the emission s within the category t, C_{st} is the contribution of the emission s to the category t and, finally, C_t is the global value of the category t. C_t units depend on the category considered.

Normalization provides a measure of the relative contribution from a product system to the impact indicators identified by dividing the potential impact per functional unit by the impact score of a reference situation (equation [2.7]). Total yearly emissions for a reference year in a reference region are normally used to calculate normalization figures (Huijbregts et al., 2003).

$$C_{nt} = \frac{C_t}{N_t} \qquad [2.7]$$

Where N_t and C_{nt} represent the normalization factor and normalized value for the category t.

The weighting step implies the application of preferences and stakeholder values in a ranking, for grouping or quantitative weighting of the impact categories (Finnveden et al., 2009). Hence, this is quite a subjective phase within the LCIA (Finnveden and Moberg, 2005). Finally, during the interpretation stage, the results from the previous phases are evaluated in relation to the goal and scope in order to reach conclusions and recommendations (ISO, 2006a).

The mid-point impact evaluation method from the Dutch Institute of Environmental Sciences (CML) of the Leiden University was applied in this thesis (Guineé, 2001). As in the case of EF, the specific factors employed are indicated in the corresponding chapter. The software SIMAPRO (Pré Consultants, 2010) was applied to aid calculations at some stages of this thesis. For simplified tools, based on a gate-to-gate approach, the LCA methodology was implemented in a spreadsheet in MS Excel®.

2.1.3. Environmental Risk Assessment

Environmental Risk Assessment (ERA) is a standardized process for the estimation of the magnitude, probability and uncertainty of adverse effects on health derived from the exposure to substances present in the environment (US EPA, 2009; ORNL, 2009). Risk assessment comprises hazard identification, exposure assessment and risk characterization, from which two relevant indicators (Hazard Quotient *HQ* and the Cancer Risk factor *CR*) can be obtained as indicated in equations [2.8] and [2.9].

$$HQ = Dose/RfD \quad [2.8]$$

$$CR = Dose \cdot SF \quad [2.9]$$

Where *HQ* is the Hazard Quotient, *Dose* (mg kg^{-1} day^{-1}) is the exposure dose to the chemical, *CR* is the Cancer Risk factor, *RfD* (mg kg^{-1} day^{-1}) and *SF* (kg day mg^{-1}) are Reference Doses for non-carcinogenic effects and Slope Factors for carcinogenic effects, respectively. *HQ* and *CR* are dimensionless. The former expresses to what extent the maximum allowable doses are achieved, so that it must be less than 1 to ensure safety conditions; meanwhile, the latter is of probabilistic nature, so that the presence of a chemical with carcinogenic effects immediately means that there is a probability of someone suffering from cancer. A maximum value of 10^{-4} is considered as admissible, according to the US EPA methodology.

Reference doses and slope factors were extracted from the United States Environmental Protection Agency and the Risk Assessment Information System (RAIS) databases (US EPA, 2009; ORNL, 2009). There are different human exposure pathways that may be considered in a risk assessment scenario to estimate the dose of the chemical to which human beings may be exposed, namely inhalation, ingestion or dermal contact. The contribution from these different pathways must be added to assess the total risk, as indicated in equation [2.9].

$$HI = \sum_{p} HQ_p \quad [2.10]$$

Where *HI* is the total Hazard Index estimated for the risk scenario and HQ_p is the Hazard Quotient assessed for each pathway *p* considered in the study.

The exposure dose is a function of a set of parameters that depend on the pathway considered (i.e., rate of ingestion of a certain contaminated food or water), characteristics of the person being exposed (i.e., body weight), and the concentration of the chemical in the media. In other cases, risk assessments are conducted on the basis of migration rates. Hence, the specific equations and parameters used to appraise exposure doses will be detailed in the corresponding chapters. In all cases, the ERA methodology was implemented in a spreadsheet in MS Excel®.

2.1.4. Energy and Materials Flow Accounting

Energy and Material Flow Analysis (EMFA) comprises a whole family of tools to optimize the consumption of energy, raw material, water and the discharge of effluents by pursuing systematically internal flows of energy and mass in production processes (Torres et al., 2008). The methodology comprises different steps (Hendriks et al., 2000): 1) Definition of the targets of the study; 2) System description; 3) Data acquisition; 4) Modeling and scenario building; 5) Results and discussion.

Energy and material flows have been defined for the different production processes studied in this thesis. To fill in the gaps in initial inventories provided by the companies, information from Ecoinvent database (Frischknecht and Rebitzer, 2005) was employed, as well as correlations and emission factors.

To support the modeling and scenario building, the computer software Umberto® 5.5, developed by ifu - Institute for Environmental Informatics Hamburg GmbH and ifeu - Institute for Energy and Environmental Research Heidelberg Ltd. Umberto® offers versatility to model, calculate and visualize material and energy flow systems under particular specifications (Wohlgemuth et al., 2006). The main elements used to construct the models with Umberto® are transitions, places and arrows (Figure 2.4). Transitions are shown as squares and indicate the location of material or energy transformations. A place, represented by a circle, is a site where material and energy are stored or distributed. There are four different types of places: input and output, which determine the boundaries of the network; storage, to store materials; connection, for the distribution of flows; port, that link two network layers. Finally, arrows connect places and transitions and show the direction of flow (ifu and ifeu, 2005).

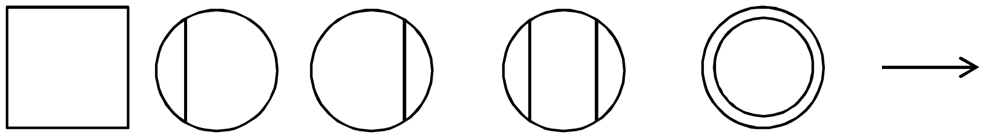


Figure 2.4. Representation of main elements of flow networks in Umberto®.

2.2. Multi-criteria analysis methodologies

All multi-criteria problems consist of a series of common elements, such as the goal of the study, the alternatives considered and the criteria on which the selection is based, and a similar general structure is followed during their resolution (Figure 2.5). The criteria must be estimated for each alternative considered; hence, a decision matrix similar to that in Table 2.1 can be obtained, where k represents the number of criteria considered and n the number of alternatives defined. The following notation is used in this thesis:

- $F = \{g_1, \ldots, g_j, \ldots, g_k\}$ is the set of criteria. J denotes the set of criteria indices.
- $A = \{a_1, \ldots, a_i, \ldots, a_n\}$ is the set of alternatives.
- $W = \{w_1, \ldots, w_j, \ldots, w_k\}$ is the weight vector modeling the preferences of the decision maker.
- $g_j(a_i)$ is the evaluation of criterion g_j for alternative a_i.

The particular methodologies of the multi-criteria analysis (MCA) techniques applied in this thesis, namely AHP, Fuzzy Logic techniques, ELECTRE III and PROMETHEE/GAIA, are described in the following sections, as well as the software tools employed.

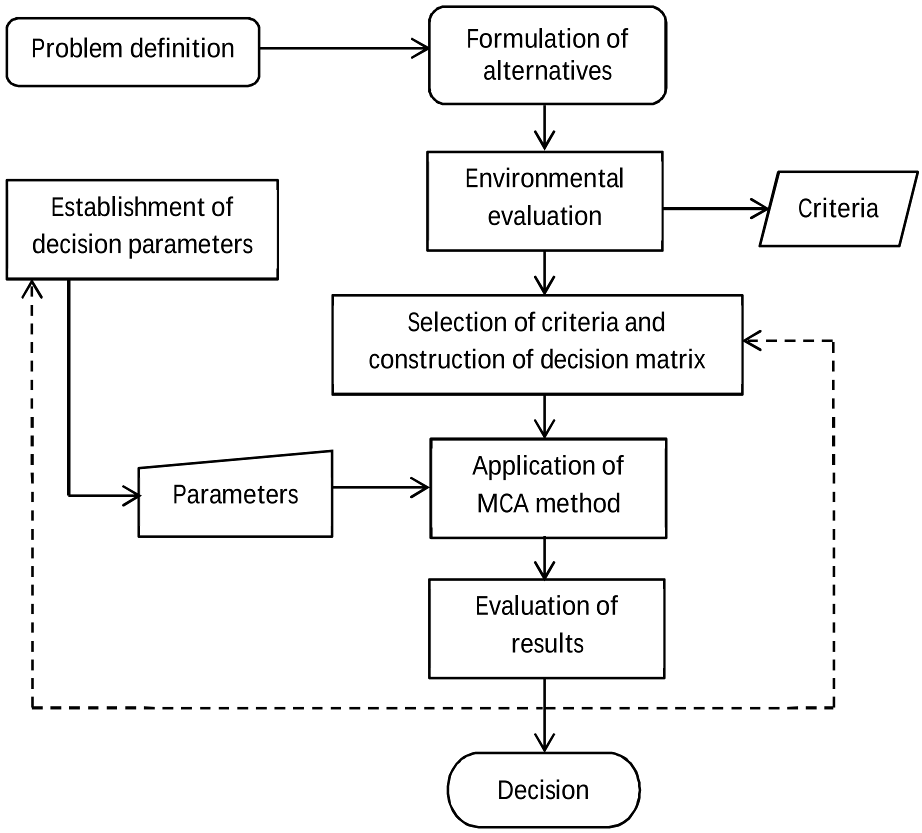


Figure 2.5. General scheme followed for MCA problems.

Table 2.1. Typical decision matrix employed in multi-criteria decision problems.

		Criteria					
		g_1	g_2	...	g_j	...	g_k
Alternatives	a_1	$g_1(a_1)$	$g_2(a_1)$	...	$g_j(a_1)$	...	$g_k(a_1)$
	a_2	$g_1(a_2)$	$g_2(a_2)$	...	$g_j(a_2)$	...	$g_k(a_2)$
	...	...	...	...	...	...	...
	a_i	$g_1(a_i)$	$g_2(a_i)$	...	$g_j(a_i)$	...	$g_k(a_i)$
	...	...	...	...	...	...	...
	a_n	$g_1(a_n)$	$g_2(a_n)$	...	$g_j(a_n)$	...	$g_k(a_n)$

2.2.1. AHP

In the Analytical Hierarchy Process (AHP), pioneered by Saaty (1980), the problem is structured in a hierarchy of different levels constituting goal, criteria, sub-criteria and alternatives (Figure 2.6).

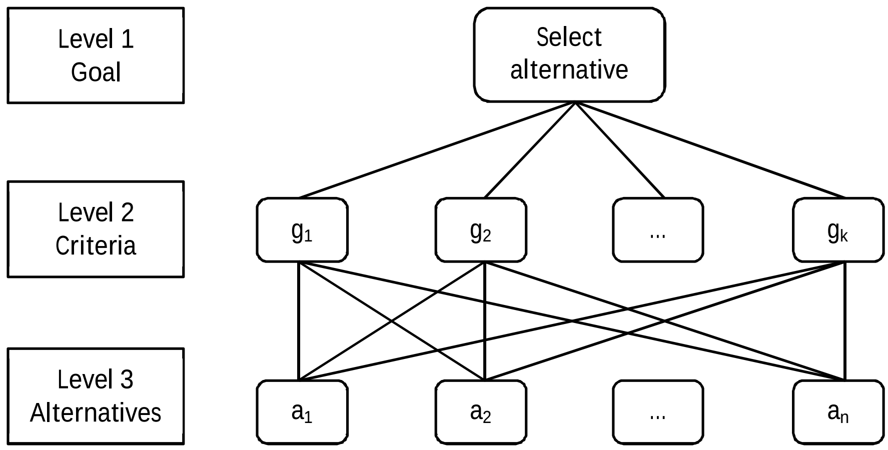


Figure 2.6. Hierarchical structure of AHP methodology.

Elements at given hierarchy level are compared in pairs to assess their relative preference with respect to each of the elements at the next higher level (Pohekar and Ramachandran, 2004; Vaidya and Kumar, 2006). This requires $m(m\text{-}1)/2$ comparisons, where m is the number of elements, with the considerations that diagonal elements are equal or 1 and the other elements will simply be the reciprocals of the earlier comparisons. Table 2.2 shows the comparison matrix at criteria level, where p indicates the intensity of preference. At the second hierarchical level, comparisons are carried out between alternatives for each criterion; hence, there will be as many matrices as criteria. The intensity of preference between two elements is established on the basis of the Saaty's scale of 1-9. The value 1 indicates equal importance, 3 moderately more, 5 strongly more, 7 very strongly and 9 indicates extremely more importance. The values of 2, 4, 6 and 8 are allotted to indicate compromise values of importance. Thus, the methodology calibrates both quantitative and qualitative performances.

Table 2.2. Pair wise comparison at criteria level in AHP process.

		Criteria					
		g_1	g_2	…	g_i	…	g_k
Criteria	g_1	1	p_{12}	…	p_{1i}	…	p_{1k}
	g_2	$1/p_{12}$	1	…	p_{2i}	…	p_{2k}
	…	…	…	1	…	…	…
	g_i	$1/p_{1i}$	$1/p_{2i}$	…	1	…	p_{ik}
	…	…	…	…	…	1	…
	g_k	$1/p_{1k}$	$1/p_{2k}$	…	$1/p_{ik}$	…	1

From here, weights for criteria and alternatives can be obtained following different methods, like the arithmetic mean, the eigenvalue approach proposed by Saaty (1980) or the geometric mean (Ishizaka and Lusti, 2006). The method developed by Saaty computes and aggregates the eigenvectors until the composite final vector of weight coefficients for alternatives is obtained. The entries of final weight coefficients vector reflect the relative importance (value) of each alternative with respect to the goal stated at the top of hierarchy.

One of the major advantages of AHP is that it calculates a consistency ratio (*CR*) as a relation between the decision maker's consistency, expressed in terms of a consistency index (*CI*), and a randomly generated index (*RI*), which is tabulated as a function of the judgment matrix dimension (Table 2.3). This ratio is important for the decision maker to assure that judgments were consistent and that the final decision is made well. The inconsistency index should be lower than 0.1 (Pineda-Henson, 2002).

$$CI = \frac{\lambda_{max} - t}{t - 1} \qquad [2.11]$$

$$CR = \frac{CI}{RI} \qquad [2.12]$$

Where λ_{max} is the maximum eigenvalue and t is the matrix dimension.

The AHP methodology developed by Saaty was implemented in Matlab® and Excel®.

Table 2.3. Randomly generated consistency index (*RI*) as a function of matrix dimension *t* (Saaty, 2005).

t	1	2	3	4	5	6	7	8	9	10
RI	0	0	0.52	0.89	1.11	1.25	1.35	1.40	1.45	1.49

2.2.2. Fuzzy Logic techniques

Fuzzy Logic (FL) techniques simplifies the process of taking decisions by simulating the way of reasoning of a human expert in environments characterized by uncertainty and imprecision. Thus, the use of FL techniques allows meshing a quantitative approach using a qualitative representation (Carrasco et al., 2002). In FL, unlike Boolean logic, an element can belong partially to several subsets with a degree between 0 and 1.

2.2.2.1. Characteristics

The most important features of FL are that it uses linguistic variables and that knowledge is represented by if-then linguistic rules.

A linguistic variable is defined by four items: 1) the name of the variable (the criteria $g_{j, j \in J}$, following the notation defined at the beginning of section 2.2); 2) its linguistic values (e.g., low, medium, high, etc.); 3) the membership functions of the linguistic values; 4) the physical domain or universe of discourse over which the variable takes its quantitative values (Phillis and Andriantiatsaholiniaina, 2001). A membership function is a curve that defines how each point in the input space is mapped to a membership value (or degree of membership) between 0 and 1. Triangular and trapezoidal functions were used in this thesis (chapter 7). These are the simplest membership functions and they are formed using straight lines. The triangular function is a collection of three points forming a triangle. The trapezoidal membership function has a flat top and it just consists of a truncated triangle curve. These straight line membership functions have the advantage of simplicity, but provide detail enough to describe the input variables considered.

The rules consist of an antecedent, in which several input variables are related by means of logical operators, and a consequent, where the same process occurs amongst the output ones (Table 2.4).

Table 2.4. Examples of typical if-then rules applied in FL.

IF	THEN
g_1 *is* Low	Output *is* Low
g_2 *is* High *and* g_3 *is* High	Output *is* Medium
…	…
g_j *is* Medium *or* g_k *is* High	Output *is* High

Each rule defines a decision node. Each decision node analyzes a condition and, using the result of it, controls the way of logical reasoning. Grouping several successive nodes, a decision tree can be generated (Figure 2.7).

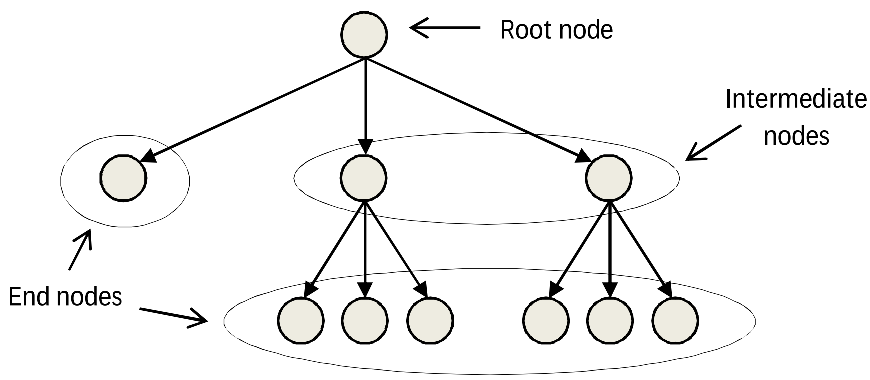


Figure 2.7. Typical branch of a decision tree.

The first premise in a decision tree is known as root node. End nodes are the end of each decision tree and they are usually used to indicate the conclusion obtained by the expert system. Finally, intermediate nodes are situated between the start and end nodes of a decision tree; in general, they do no achieve a conclusion and simply manage intermediate conditions (Carrasco et al., 2002).

2.2.2.2. Protocol

Once the membership functions and the rules are defined, the fuzzy inference process occurs in several steps (González et al., 2002; Mathworks, 2010):

1. Fuzzification. The first step is to identify for each value of the input variables, the degree of membership registered in each established label or category.
2. Apply fuzzy operation. To obtain a unique global degree of truth for the antecedent, the most accepted criteria are taking the lower degree of truth in the variables on the antecedent for *and* operator, the higher one for *or* and the gap to 1 in the case of *not*.
3. Implication or inferencing. From the global degree of truth of the antecedent, a membership function can be derived based on the membership function of the label of the output variable present in the consequent. For achieving this, several methods can be applied, the most accepted being PROD (which weights the membership function in the consequent by the value of the degree of truth of the antecedent) and MIN (which truncates the function of the activated label in the consequent according to the value of the degree of truth of the antecedent).
4. Aggregation. As several rules may affect the same output variable, it is necessary to aggregate the membership functions obtained in the inference of all the rules. The most common methods are SUM (which offers as the final membership function the sum of the ones obtained after inferencing all the rules), MAX (which offers a function that takes in each point of the output domain the maximum value of the ones obtained in each particular membership function) and PROBOR (which is very similar to SUM, but offers the sum of the values obtained in each output variable minus the end result of their multiplication).
5. Defuzzyfication. Finally, a method has to be applied for converting the membership function obtained in the previous step into a crisp value. Some common methods consist in taking the output value corresponding to the minimum, medium and maximum of the maximums of the membership function. The most frequently adopted method is the centroid, which offers as the output value the x coordinate of the center of gravity of the surface between the function and the x axis.

2.2.2.3. Software tools

The Fuzzy Logic Toolbox of Matlab® was used to apply FL techniques in this thesis. Two types of fuzzy inference systems can be implemented in the toolbox:

Mamdani-type and Sugeno-type. These two types of inference systems vary somewhat in the way outputs are determined. Mamdani-type inference expects the output membership functions to be fuzzy sets. On the other hand, Sugeno-type systems can be used to model any inference system in which the output membership functions are either linear or constant. The Mamdani-type (Mamdani and Assilian, 1975), the most commonly seen fuzzy methodology, was selected to be applied in this thesis because it is more intuitive and well suited to human input (Mathworks, 2010).

The default parameters set for the Mamdani method can be observed in Figure 2.8, although they can be changed. The centroid method is employed for defuzzification, which is considered to be the most objective one (González et al., 2002). Different membership functions can be selected for the input/output variables, defining the universe of discourse in each case. Besides, there is another option from the menu to introduce the rules.

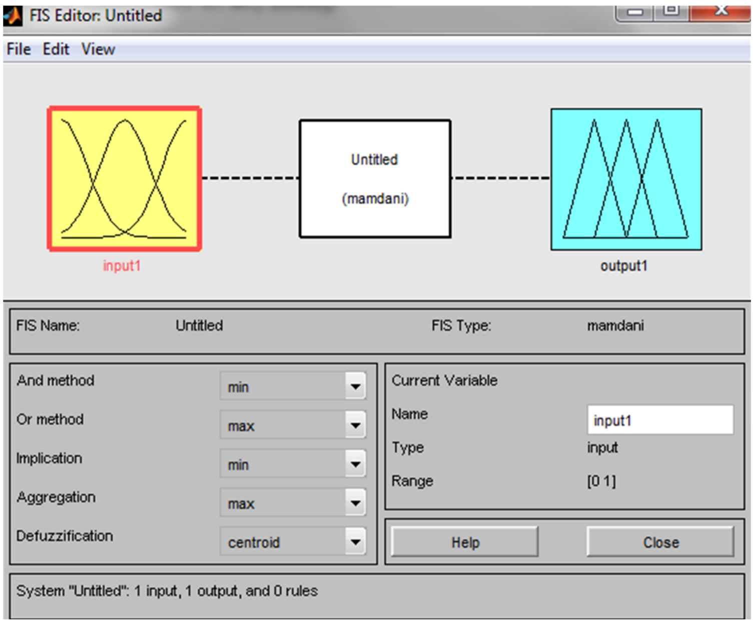


Figure 2.8. Screen from the Fuzzy Logic Toolbox of Matlab® 7.9.0 R2009b.

2.2.3. ELECTRE III

ELECTRE methods use binary outranking relations (S), whose meaning is "at least as good as", to rank alternatives (Figueira et al., 2005). Considering two alternatives a, $b \in A$, where A is the set of alternatives $\{a_1, a_2, \ldots, a_i, \ldots, a_n\}$, four situations may occur:

- aSb and not bSa, i.e., aPb (a is strictly preferred to b)
- bSa and not aSb, i.e., bPa (b is strictly preferred to a)
- aSb and bSa, i.e., aIb (a is indifferent to b)
- Not aSb and not bSa, i.e., aRb (a is incomparable to b)

The construction of an outranking relation is based on two major concepts (Figueira et al., 2005; Rousat et al., 2009):

1. Concordance. Alternative a outranks alternative b (aSb) if a sufficient majority of criteria are in favor of alternative a.
2. Non-discordance. When the concordance condition holds, none of the criteria in the minority should be opposed too strongly to the outranking of b by a.

The main difference between ELECTRE III and the other ELECTRE methods is that outranking relations can be interpreted as fuzzy relations by using pseudo-criteria. The assertion that a outranks b is characterized by a credibility index which permits knowing the true degree of this assertion (Figueira et al., 2005; Rousat et al., 2009).

2.2.3.1. Construction of an outranking relation

ELECTRE III is based on the definition of two matrices, the concordance and the discordance matrices, which determine if the statement aSb is acceptable (Papadopoulos and Karagiannidis, 2008). For a criterion j and a pair of alternatives (a,b), the concordance index (c_j) can be defined as follows:

$$c_j(a,b) = \begin{cases} 0 & d_j(a,b) \le q_j \\ \dfrac{d_j(a,b) - q_j}{p_j - q_j} & q_j < d_j(a,b) < p_j \\ 1 & d_j(a,b) \ge p_j \end{cases} \qquad [2.13]$$

$$d_j(a,b) = g_j(a) - g_j(b) \qquad [2.14]$$

Where q_j and p_j represent the indifference and the preference thresholds, respectively, which lead to a pseudo-criterion model on each criterion (Figure 2.9). If $d_j(a,b)$ is lower than q_j, there is no difference between a and b and then $c_j(a, b)$=0. Conversely, if $d_j(a,b)$ is higher than p_j, a is strictly preferred to b and then $c_j(a, b)$=1.

A global concordance index C_{ab} for each pair of alternatives (a,b) is computed with the concordance index $c_j(a,b)$ of each criterion j:

$$C_{ab} = \frac{\sum_{j=1}^{k} w_j \cdot c_j(a,b)}{\sum_{j=1}^{k} w_j} \quad [2.15]$$

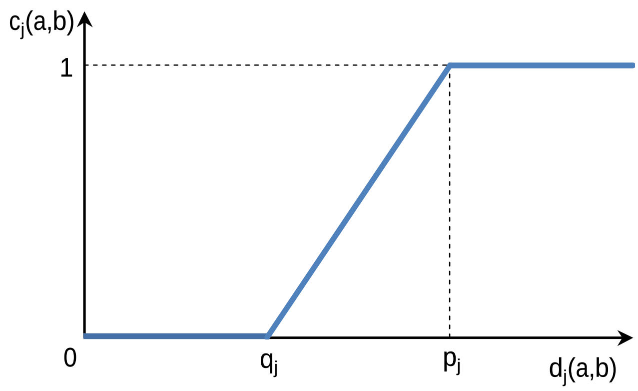


Figure 2.9. Pseudo-criterion used in ELECTRE III to determine concordance index.

A discordance index $dc_j(a,b)$ is also evaluated using a pseudo-criteria with a veto threshold v_j that represents the maximum difference $d_j(a,b)$ acceptable to not reject the assertion aSb (Figueira et al., 2005; Rousat et al., 2009). If $d_j(a,b)$ is lower than p_j, there is no discordance and $dc_j(a,b)$=0; conversely, if $d_j(a,b)$ is higher than v_j, then $dc_j(a,b)$=1. The discordance index is defined in equation [2.16].

$$dc_j(a,b) = \begin{cases} 0, & d_j(b,a) \leq p_j \\ \dfrac{d_j(b,a) - p_j}{v_j - p_j}, & p_j < d_j(b,a) < v_j \\ 1, & d_j(a,b) \geq v_j \end{cases} \quad [2.16]$$

$$d_j(b,a) = g_j(b) - g_j(a) \quad [2.17]$$

The concordance and discordance indices incorporate some senses of the membership degree as in the fuzzy systems. The construction of fuzzy relations in ELECTRE III requires the definition of a credibility index, $\rho(a,b)$, which characterizes the credibility of the assertion aSb (Figueira et al., 2005). It is defined in equation [2.18].

$$\rho(a,b) = C(a,b)\prod_{j\epsilon V}\frac{1-dc_j(a,b)}{1-C(a,b)} \quad V = \{j \in J, dc_j(a,b) > C_{ab} \qquad [2.18]$$

When a veto threshold is exceeded for at least one criterion, the index of credibility is null, i.e., the assertion "a outranks b" is rejected.

2.2.3.2. The exploitation procedure

The exploitation procedure consists of two phases. In the first phase, two complete pre-orders, descending (Z_1) and ascending (Z_2) need to be built. In the second phase, the partial pre-order Z is obtained as the intersection of the previous two complete pre-orders (Tervonen et al., 2005; Figueira et al., 2005).

The partial pre-order Z_1 is defined as a partition of the set A into L ordered classes, B_1, ... , B_l, ... , B_L, where B_1 is the head-class in Z_1. Each class B_h is composed of *ex æquo* elements according to Z_1. The complete pre-order Z_2 is determined in a similar way, where A is partitioned into E ordered classes, G_1, ..., G_e, ..., G_E, the latter being the head-class (Figueira et al., 2005).

The pre-orders are obtained as output from two distillation procedures: Z_1 from ascending (upward) distillation and Z_2 from descending (downward) distillation. In the descending distillation, the procedure designed to compute Z_1 starts (first distillation) by defining an initial set $D_0 = A$; it leads to the first final distilled B_1. After getting B_l, the procedure sets $D_0 = A\backslash(C_1 \cup ... \cup C_l)$. The actions in class C_l are, according to Z_1, preferable to those of class C_{l+1}; for this reason, distillations that lead to these classes are called descendants (top-down). The procedure leading to Z_2 is almost similar to descending distillation, but now the actions in class G_{e+1} are preferred to those in class G_e; these distillations are called ascendants (bottom-up). The complete algorithms for the distillation processes were not reproduced here since they were considered to fall out of the scope of this work.

The partial pre-order Z is computed as the intersection of Z_1 and Z_2. A complete pre-order is finally suggested taking into account the partial pre-orders and some additional considerations.

2.2.3.3. Software tools

ELECTREE III/IV software was developed between 1992 and 1994 by the Dauphine University of Paris and the Institute of Computing Science of the University of Poznan in Polonia. It implements the decision support models ELECTREE III and IV. However, this was not employed during the development of this thesis (chapter 8) but the ELECTREE III methodology was implemented in Matlab® and Excel®.

2.2.4. PROMETHEE/GAIA

PROMETHEE (Preference Ranking Organization Method for Enrichment Evaluation) is a non-parametric outranking method for a finite set of alternatives and is based on positive (out -) and negative (in -) preference flows for each alternative in the valued outranking relation to rank the alternatives according to the selected preferences (weights).

To obtain the outranking of alternatives, two previous steps have to be carried out: determination of weights and selection of preference function. The weights (w_j, j = 1, 2,..., k) represent the relative importance of the different criteria (the higher the weight, the more important the criterion). They are non-negative numbers independent from the measurement units of the criteria. Weights can be established so that $\sum_{j=1}^{k} w_j$ = 1, or they can be latter normalized dividing the numbers by their sum.

For each criterion, the preference function translates the difference between the evaluations obtained by two alternatives into a preference degree ranging from zero to one (Behzadian et al., 2010). The larger the deviation, the larger the preference (Brans and Mareschal, 2005). The preference function is defined as follows, where A is the set of alternatives $\{a_1, a_2, ..., a_i, ..., a_n\}$:

$$P_j(a,b) = F_j[d_j(a,b)] \qquad \forall\, a,b \in A \qquad [2.19]$$

$$0 \leq P_j(a,b) \leq 1 \qquad [2.20]$$

In case of a criterion to be maximized, this function is giving the preference of *a* over *b* for observed deviation between their evaluations on criterion $g_j(\cdot)$. The preferences equal 0 when the deviations are negative:

$$P_j(a,b) > 0 \Rightarrow P_j(b,a) = 0 \quad [2.21]$$

For criteria to be minimized, the preference function should be reversed or alternatively given by:

$$P_j(a,b) = F_j[-d_j(a,b)] \quad [2.22]$$

Six types of particular preference functions have been proposed (Table 2.5).

Table 2.5. Types of preference functions for PROMETHEE method.

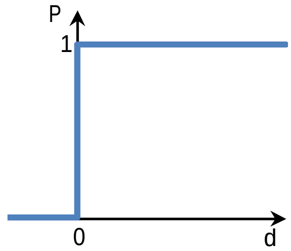
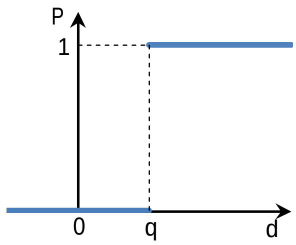
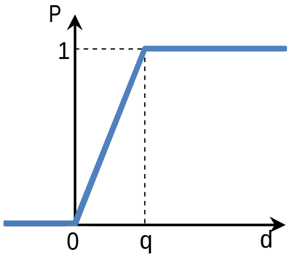
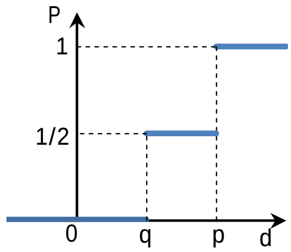
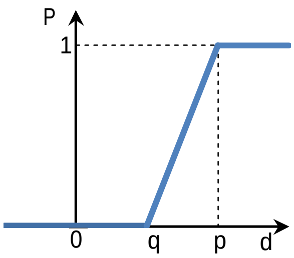
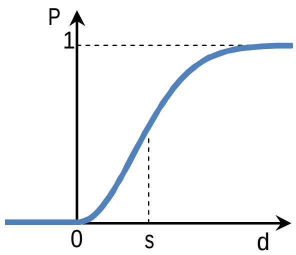
Type	**Function**	**Definition**	**Para-meters to fix**
Type 1 Usual criterion		$P(d) = \begin{cases} 0, & d \le 0 \\ 1, & d > 0 \end{cases}$	-
Type 2 U-shape criterion		$P(d) = \begin{cases} 0, & d \le q \\ 1, & d > q \end{cases}$	q

Table 2.5 (*cont.*). Types of preference functions for PROMETHEE method.

Type	Function	Definition	Para-meters to fix
Type 3 V-shape criterion		$P(d) = \begin{cases} 0, & d \leq 0 \\ \frac{d}{p}, & 0 < d \leq p \\ 1, & d > p \end{cases}$	*p*
Type 4 Level criterion		$P(d) = \begin{cases} 0, & d \leq 0 \\ \frac{1}{2}, & q < d \leq p \\ 1, & d > p \end{cases}$	*p, q*
Type 5 V-shape with indifference criterion		$P(d) = \begin{cases} 0, & d \leq 0 \\ \frac{d-q}{p-q}, & q < d \leq p \\ 1, & d > p \end{cases}$	*p, q*
Type 6 Gaussian criterion		$P(d) = \begin{cases} 0, & d \leq 0 \\ 1 - e^{-\frac{d^2}{2s^2}}, & d > 0 \end{cases}$	*s*

The significance of the parameters to be defined is: q is a threshold of indifference; p is a threshold of strict preference; s is an intermediate value between q and p. The q indifference threshold is the largest deviation which is considered as negligible by the decision maker, while the p preference threshold is the smallest deviation which is considered as sufficient to generate a full preference (Brans and Mareschal, 2005).

From here, the following aggregated preference indices can be defined:

$$\begin{cases} \pi(a,b) = \sum_{j=1}^{k} P_j(a,b)w_j \\ \pi(b,a) = \sum_{j=1}^{k} P_j(b,a)w_j \end{cases} \qquad [2.23]$$

$\pi(a,b)$ is expressing with which degree a is preferred to b over all the criteria and $\pi(b,a)$ how b is preferred to a.

Then, the outranking flows are calculated, taking into account that each alternative a is facing (n-1) other alternatives in A. The positive outranking flow, $\phi^+(a)$, expresses how an alternative a is outranking all the others. The higher $\phi^+(a)$, the better the alternative. Meanwhile, the negative outranking flow, $\phi^-(a)$, expresses how an alternative a is outranked by all the others. The lower $\phi^-(a)$, the better the alternative.

$$\Phi^+(a) = \frac{1}{n-1}\sum_{x\in A} \pi(a,x) \qquad [2.24]$$

$$\Phi^-(a) = \frac{1}{n-1}\sum_{x\in A} \pi(x,a) \qquad [2.25]$$

2.2.4.1. The PROMETHEE I Partial Ranking

The PROMETHEE I partial ranking (P^I, I^I, R^I) is obtained from the positive and the negative outranking flows. Both flows do not usually induce the same rankings. PROMETHEE I is their intersection.

$$\begin{cases} aP^I b \quad if \quad \begin{cases} \Phi^+(a) > \Phi^+(b) \ \ and \ \ \Phi^-(a) < \Phi^-(b), or \\ \Phi^+(a) = \Phi^+(b) \ \ and \ \ \Phi^-(a) < \Phi^-(b), or \\ \Phi^+(a) > \Phi^+(b) \ \ and \ \ \Phi^-(a) = \Phi^-(b) \end{cases} \\ aI^I b \quad if \quad \ \Phi^+(a) = \Phi^+(b) \ \ and \ \ \Phi^-(a) = \Phi^-(b) \\ aR^I b \quad if \quad \begin{cases} \Phi^+(a) > \Phi^+(b) \ \ and \ \ \Phi^-(a) > \Phi^-(b), or \\ \Phi^+(a) < \Phi^+(b) \ \ and \ \ \Phi^-(a) < \Phi^-(b) \end{cases} \end{cases} \qquad [2.26]$$

Where P^I, I^I and R^I stand for preference, indifference and incomparability, respectively.

2.2.4.2. The PROMETHEE II Complete Ranking

PROMETHEE II consists of the (P^{II}, I^{II}) complete ranking based on the net outranking flow ϕ(*a*), which is the balance between the positive and the negative outranking flows. The higher the net flow, the better the alternative. In this case all the alternatives are comparable.

$$\Phi(a) = \Phi^+(a) - \Phi^-(a) \qquad [2.27]$$

$$\begin{cases} aP^{II} b \quad if \quad \Phi(a) > \Phi(b) \\ aI^{II} b \quad if \quad \Phi(a) = \Phi(b) \end{cases} \qquad [2.28]$$

2.2.4.3. The GAIA plane

GAIA (Geometrical Analysis for Interactive Aid) is a descriptive complement to the PROMETHEE methods that makes use of the Principal Components Analysis (PCA) technique. It is based on the single criterion net flows defined by equation [2.25], where $\phi_j(a)$ is the single criterion net flow obtained when only criterion $g_j(\cdot)$ is considered. It expresses how an alternative *a* is outranking ($\phi_j(a) > 0$) or outranked ($\phi_j(a) < 0$). The matrix $M(n \times k)$ of the single criterion net flows of all the alternatives is not depending on the weights of the criteria, as it can be derived from equation [2.29].

$$\Phi(a) = \frac{1}{n-1} \sum_{x \in A} \left[P_j(a, x) - P_j(x, a) \right] \qquad [2.29]$$

$$\Phi(a) = \sum_{j=1}^{k} \Phi_j(a) w_j \qquad [2.30]$$

The set of the *n* alternatives can be represented as a cloud of *n* points in a *k*-dimensional space. To obtain a clear view of the relative position of the points with regard to the criteria, the information included in the *k*-dimensional space is projected on a plane. The GAIA *plane* is the plane for which as much information as possible is preserved after projection. According to the principal components analysis technique, it is defined by the two eigenvectors corresponding to the two largest eigenvalues of the covariance matrix *M'M* of the single criterion net flows (Brans and Mareschal, 2005).

The following elements are meaningful in the GAIA plane (Visual Decision Inc., 2009):

- Criteria are represented by axes. Axes oriented in similar directions correspond to criteria that are in general agreement. Longer axes correspond to criteria for which more important deviations have been observed.
- Actions are represented by shapes. The proximity between the shapes indicates actions with similar profiles. The actions that perform better on a given criterion will be located farther away in the direction of that criterion.
- The names of the criteria are represented by a separate axis. Its orientation emphasizes which criteria are predominant and which are possibly neglected.

2.2.4.4. Software tools

The software Decision Lab 2000 – Executive Edition 1.0 was used to aid the methodological calculations above described for PROMETHEE and GAIA (Visual Decision Inc., 2009).

2.3. Statistical analysis

This section deals with those statistical analysis tools that have been employed along the thesis to support, complement and give consistency to the different estimates conducted.

2.3.1. Sensitivity analysis

During the application of the environmental evaluation methodologies, sensitivity analyses based on Monte Carlo simulations were carried out to assess how changes in the input variables may affect the outputs to a specific model. This is a particularly useful tool for enterprises to predict behaviors based on operation decisions.

Besides, some of the methodologies above described depend on parameters that are not defined objectively: the weights in MCA, *p* and *q* for ELECTREE and PROMETHEE, etc. For this reason, a final step in the application of these methodologies is to conduct sensitivity and robustness analyses, in order to verify the influence that a variation on these parameters may have in the final ranking of alternatives.

The software Decision Lab internally incorporates this option. Meanwhile, the software Crystal Ball® 7.2.2 Edition (Decisioneering, 2005) was used in the cases when the methodology applied was implemented in a spreadsheet. Crystal Ball is a spreadsheet-based application suite for predictive modeling, forecasting, simulation and optimization. It runs under MS Excel®.

2.3.2. Statistical correlations

The application of this statistical technique was used in chapter 4. The software SPSS 17.0 was employed to seek the functions that better adjusted the data, based not only on the R^2 for the correlation, but also on the standard errors and p-values to assure significance of the model obtained.

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Chapter 3

EF application in the textile sector

Chapter 3

EF application in the textile sector

Abstract

The work developed in this chapter constituted a pioneering application of the Ecological Footprint (EF) to a production process. Until then, the EF had mostly been used as an indicator of environmental sustainability applied to individual lifestyles, regions, nations or even the world. Although its application to businesses and industry had been suggested, scarce initiatives existed in this field.

In the study presented here, a textile tailoring plant was analyzed. The overall purpose was to develop a tool useful for evaluating the environmental impact evolution due to the performance of the plant, as well as for comparing the environmental behavior of different tailoring processes. Therefore, the EF methodology was adapted to the case study and the selected data were those from the manufacturing process. Data were divided into three main categories: energy, resources and waste.

The principal contribution to the final EF (expressed in hectares of land) was the resources category, mainly due to the high value associated to the cloth. The consumed energy was the second contributor, while the waste category remained in third place. The final outcomes were divided by the production rates to obtain a comparable relative index, easy to be interpreted by the different stakeholders. This is of special importance for a company involved in Corporate Social Responsibility and thus meant to have a general communication strategy.

3.1. Introduction

The textile sector in Spain is composed of 6,100 companies with 206,000 workers. This figure means the 7% of the total industrial employment, situating this sector among those more outstanding in the Spanish industrial structure (Cityc, 2010). It is considered the sixth most important sector in the Spanish industry considering its economical results with a contribution to Gross Domestic Product (GDP) around 5% in the last years (Table 3.1.). In the particular case of the tailoring sub-sector the significance is even higher (INE, 2000).

Table 3.1. General data of the textile and tailoring sector in Spain. Data related to 2006 unless specified (Cityc, 2010).

Number of enterprises	6,100
Number of employees	206,000
Production (million €)	11,415
Exports (million €)	7,356

Contribution to GDP[(a)] 2001	1%
Contribution to GDP 2005	5%
Contribution to GDP 2006	4%

[(a)] Gross Domestic Product

In Galicia (NW Spain), the fashion industry has acquired especial importance in the last years due to the presence of several designers of national and international renown (Figure 3.1 and Table 3.2). This caused a strong development of this industry, which generated a great impact in the Galician economy and at the same time contributed to develop this source of employment.

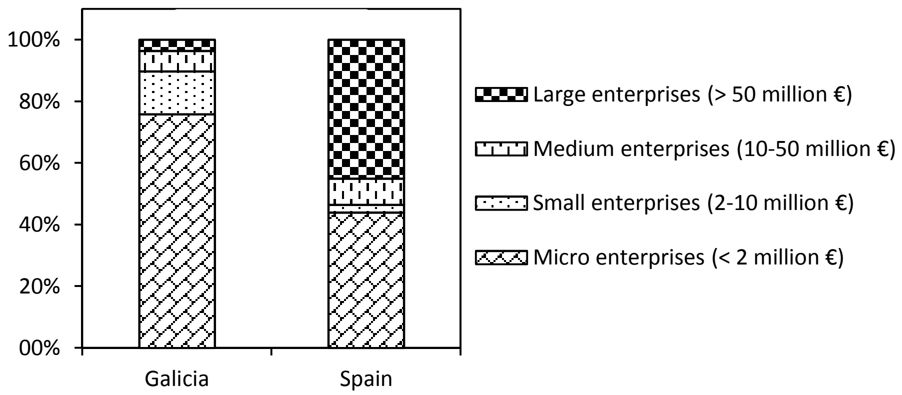

Figure 3.1. Fashion industry enterprises classification depending on the income. Data related to 2008 (Ardán, 2010).

Table 3.2. Textile sector (tailoring) enterprises with highest income (Ardán, 2010).

Enterprises in Galicia		**Income (€)** 2007	2008	**Change rate**
1	SOCIEDAD TEXTIL LONIA, S.A.	175.458.283		
2	ADOLFO DOMINGUEZ, S.A.	178.775.676		
3	INDIPUNT, S.L.	153.710.150	172.460.819	12.20%
4	STEAR, S.A.	169.357.327	128.914.759	-23.88%
5	ZINTURA, S.A.	104.515.363	76.709.447	-26.60%
6	DENLLO, S.A.	95.059.387	75.604.572	-20.47%
7	GLENCARE, S.A.	88.870.529	72.220.968	-18.73%
8	CONFECCIONES FIOS, S.A.	91.596.996	72.203.075	-21.17%
9	CARAMELO, S.A.	53.394.357	68.966.155	29.16%
10	BIMBA & LOLA, S.L.	39.988.791	48.561.524	21.44%
11	SAMLOR	54.723.249	47.123.413	-13.89%
12	CONFECCIONES GOA, S.A.	42.114.086	39.564.212	-6.05%
13	NIKOLE, S.A.	46.334.895	38.759.467	-16.35%
14	TRISKO, S.A.	57.426.509	37.664.545	-34.41%
15	CHOOLET, S.A.	39.257.893	28.540.873	-27.30%
16	SELMARK, S.L.U.	24.496.427	21.657.196	-11.59%
17	FLORENTINO COLECCION, S.L.	19.300.260	19.832.452	2.76%
18	UTERQUE, S.A.	1.855.719	18.650.201	905.01%
19	KARPI CONFECCION, S.L.	13.984.266	16.204.660	15.88%
20	CONFECCIONES ESQUIO, S.A.	14.195.460	14.345.245	1.06%

No. of enterprises of the sector: 218.

Enterprises in Spain				
1	HENNES & MAURITZ, S.L.	558,138,000		
2	INDUSTRIAS Y CONFECCIONES, S.A.	526,407,489	408,382,000	-22.42%
3	SOCIEDAD TEXTIL LONIA, S.A.	185,458,283		
4	ADOLFO DOMINGUEZ, S.A.	178,775,676		
5	BURBERRY (SPAIN), S.A.	253,066,409	175,608,329	-30.61%

No. of enterprises of the sector: 690.

There are near 400 textile enterprises in Galicia (IGE, 2004), distributed in the 4 provinces of this region as shown in Figure 3.2.

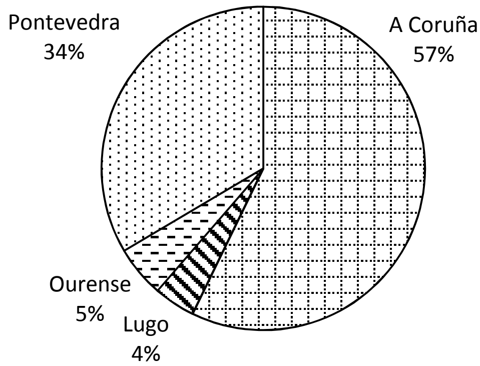

Figure 3.2. Distribution per province of textile enterprises in Galicia (IGE, 2004).

The factories are concentrated in few main locations. Arteixo is the most representative one, since it is where *Industrias de Diseño Textil, S.A.* (Inditex), the best example of this major development, has most of its factories (15 in total in Galicia). Nonetheless, there are many other factories in locations as A Coruña, Ferrol, Lalín, Ourense, Redondela, San Cibrao das Viñas, Santiago de Compostela and Vigo, where clothes for designers like *Carolina Herrera*, *Purificación García*, *Adolfo Domínguez*, *Roberto Verino* or *Florentino* are created (Figure 3.3).

In the last years, there has been an increase in the number of regulatory laws and voluntary and administrative instruments affecting different environmental management issues. Regarding the former, special attention is deserved by the Integrated Pollution Prevention and Control -IPPC- Law (Spanish Government, 2002), transposition of the IPPC 96/61 Directive that was recently actualized (European Commission, 2008). With respect to the latter, some examples are the ISO 14000 Standards, Eco-Management and Audit Scheme (EMAS), Eco-Label, Integrated Product Policy (IPP), Corporate Social Responsibility (CSR), etc., which are also important. This trend, together with the growing concern of the general public, has posed a change in management in all those companies willing to fulfill both the Administration requirements and society's demand of information. As a result, the evaluation of the environmental behavior of the textile sector in Spain and, particularly, in Galicia is of great interest. However, the lack of suitable evaluation tools makes necessary to develop adapted or simplified tools for being applied to a particular sector, as it is stated in the IPP (European Commission, 2003). Furthermore, those enterprises involved in a CSR strategy have the need for tracking their impact through indexes easy to be interpreted by the different stakeholders (Inditex, 2005; GRI, 2006).

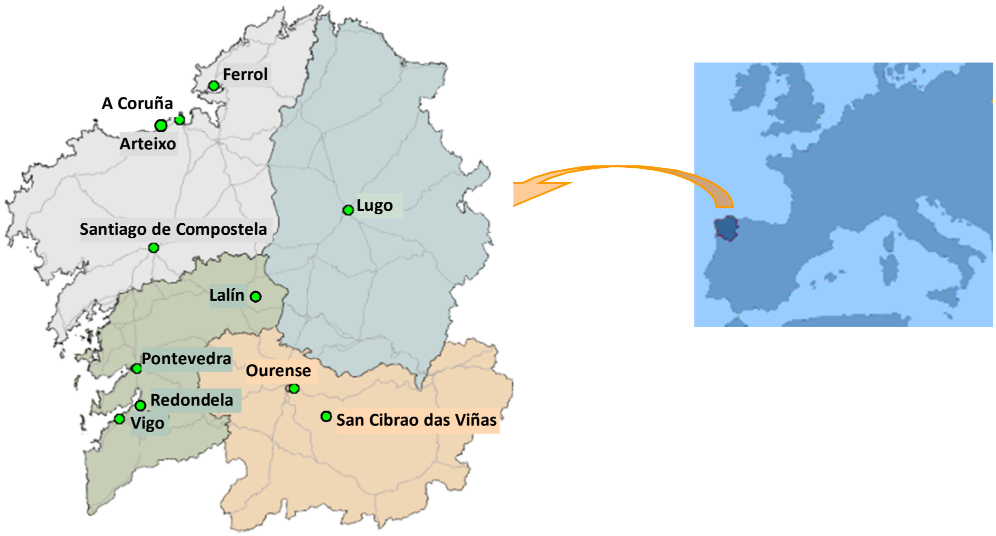

Figure 3.3. Geographical situation of the main locations where the textile sector enterprises are established in Galicia.

Many inventory data are required when applying environmental evaluation tools. This common first methodology stage is maybe the most laborious task (Dahllöf, 2004). Although the whole information is undoubtedly valuable at specific decision making level (Azapagic, 1999), it is also especially appealing the idea of summarizing all these values in only one index. In this sense, the Ecological Footprint (EF) fits all the characteristics desirable for this kind of indicator (Chambers and Lewis, 2001). Besides, it has the advantage of being a composite indicator that does not rely on the assignment of weights based on expert opinion; rather, the aggregation is carried out using empirical coefficients related to the productivity of the different area types considered.

Rees and Wackernagel (1996) defined the EF as the amount of land and water area a human population would hypothetically need to provide the resources required to support itself and to absorb its wastes. It has traditionally been applied to evaluate the environmental sustainability of individual lifestyles and at different geographical scales (towns, cities, regions, nations). The Global Footprint Network (GFN) publishes every year in the Living Planet Report a list of the calculated Ecological Footprints (GFN, 2006), as well as the biocapacity, of a large number of countries. Many other studies have been carried out to estimate the EF of regions, cities, towns, etc., throughout the world (Relea and Prat, 1998; Wackernagel, 1998; Barret and Scott, 2001; IHOBE, 2005).

By the time the work of this chapter was carried out, the application of the EF to different fields from those for which it was conceived was still scarce. Nonetheless, its application to enterprises had been suggested, taking into account that they are also goods and services consuming organizations which generate wastes. It was considered that this tool could also be used to scrutinize the ecological sustainability of processes and projects, rather than merely applying the analysis at various geographic or social scales (Wackernagel and Yount, 2000). However, no references in which the EF was applied with this purpose to an industrial production process were found in the literature by that moment. Examples falling out of the traditional scope were: the calculation of a hospital's EF (Germain, 2001); EF associated to a sports event audience (Collins and Flynn, 2004); EF of educational centers (Flint, 1999; Wood and Manfred, 2003). Albeit, there were other case studies which approximated most to the characteristics of a production process, like the Port of Gijón -NW Spain- (Doménech, 2004) or the dairy production (Beynon et al., 2002).

In this chapter the EF methodology was adapted to be used in the textile sector. Based on this concept, a tool for evaluating the sustainability of a dressmaking plant was developed. The product outputs of the plant were cotton jackets, for either men or women, already packed in a plastic bag. This tool was tested with data obtained during the period 2002-2005. Its application in the future will allow for comparing the environmental behavior of this plant with other similar ones (Albino and Kühtz, 2003).

3.2. Methodology

The equations that allow for the transformation of the original inventoried data into area units were described in Chapter 2. Based on the EF component method (Monfreda et al., 2004) and on a mutually exclusive use of land approach, a tool adapted to the particular case of a dressmaking factory was developed in MS Excel®. The particular input and output flows of the process were identified, as well as the necessary parameters to apply the methodology, and finally the mathematical equations were introduced to estimate the EF of the process.

3.2.1. Case study

A general scheme of the production process is shown in Figure 3.4 to manufacture the jackets, the cotton fabric (or any other used according to the current design) enters the factory where it will be cut and sewed according to a given pattern. The pieces of the jacket are first drawn on paper and then placed on the fabric, putting plastic over them to avoid undesired movements during the cutting process. Buttons, zips and other ornamental elements are added to the item of clothing. This part of the process is carried out externally, although the accessories are provided by the factory, but there are not records of energy consumption during this stage. Back at the plant, the jackets are ironed (steam supply is required for this part of the process). Finally, they are labeled and packed into bags to be stored and later distributed. The sources of energy are: electricity, wind-power, propane and gas-oil. The latter two are employed in cogeneration units, where the air emissions are generated and released. These facilities are common for a series of dressmaking factories belonging to the same company. The hazardous wastes are mainly generated in maintenance operations, also common for the different factories; meanwhile, the water is mainly consumed in the lavatories.

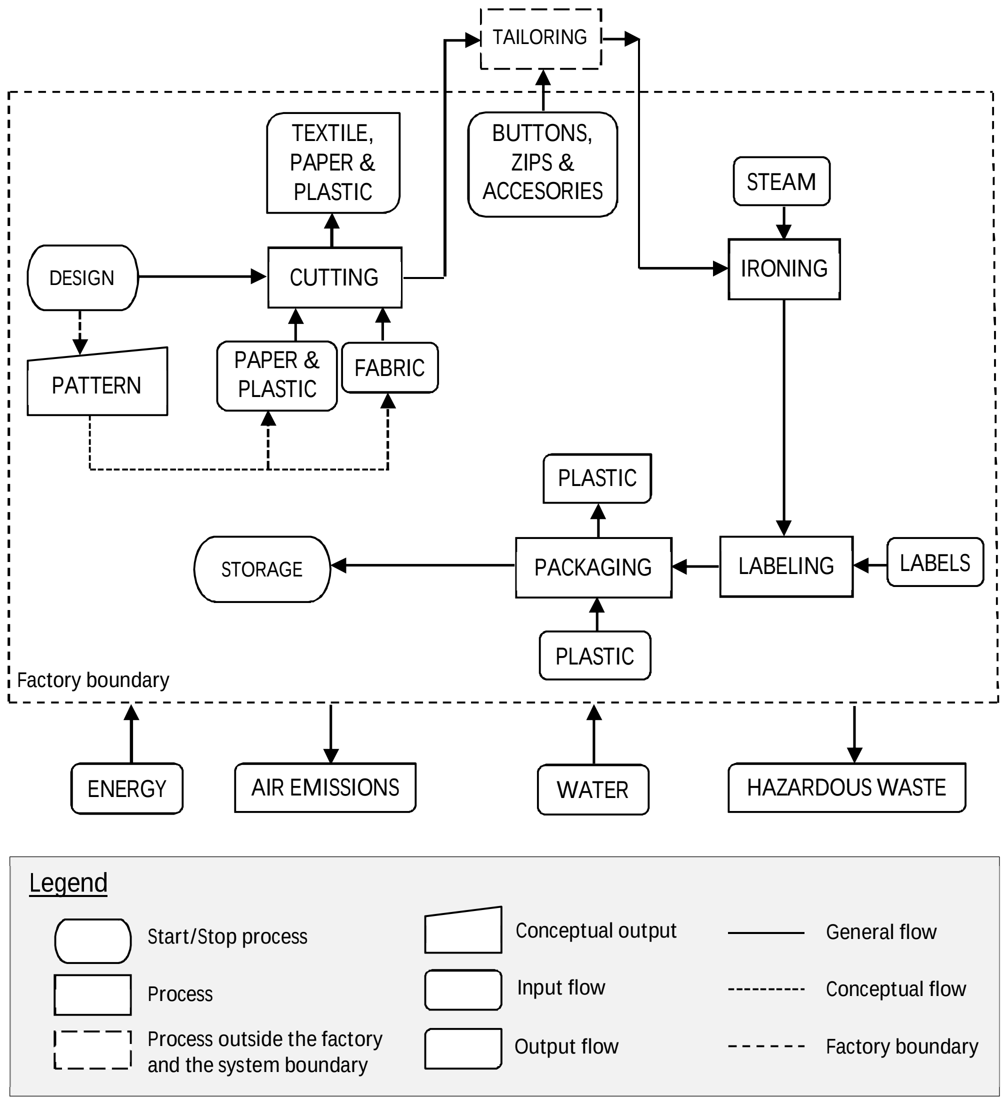

Figure 3.4. Dressmaking factory process flowsheet.

3.2.2. EF estimate

Data from the production process were classified into three main categories: energy, resources and waste, referring the first two ones to consumption, whilst the last one refers to generation. The components (V_i) included in each of them are those shown in Table 3.3. It must be noticed that a specific methodology to assess the footprint of nuclear energy does not exist. By the time this work was developed, it was appraised considering that an equivalent amount of fossil energy was employed; however, in 2007 the National Footprint Accounts (NFA) Committee concluded that this approach was not a scientifically justifiable method for calculating the footprint of nuclear electricity, and therefore the nuclear component was removed from NFA methodology (Ewing et al., 2008).

Table 3.3. Categories considered in the EF estimate.

Energy category	**Units**	**Resources category**	**Units**	**Waste category**	**Units**
Carbon	kWh	Plastic	t	Paper & cardboard	t
Liquid fuel	kWh	Paper & cardboard	t	Plastic	t
Gas fuel	kWh	Cotton textile	t	Textile	t
Nuclear	kWh	Synthetic stitch	t	Urban waste	t
Hydroelectric power	kWh	Wool stitch	t		
Wind power	kWh	Wood	t		
Solar energy	kWh	Metal	t		
Biomass	t	Water	m^3		

Assessing the EF associated to the production of goods grown in land requires investigating its natural productivity, by which the initial value must be divided in order to obtain the final area. However, when discussing about other materials, they must be converted into the equivalent energy consumed during their manufacture. In this case, the transformed value in energy units must be divided by the energy productivity of the land, i.e. the amount of energy that can be produced or assimilated by a hectare of land. The whole EF of the materials is attributed to fossil fuel. It must be noticed that in the considered case only the dressmaking process is studied within the textile chain. Accordingly, the material inputs to the plant are constituted by already manufactured products, while the output is made up of items of clothing ready to be sent to the shops.

To identify the primary sources of energy associated to the electricity consumption, the breakdown provided by the power supplier company was applied (Figure 3.5).

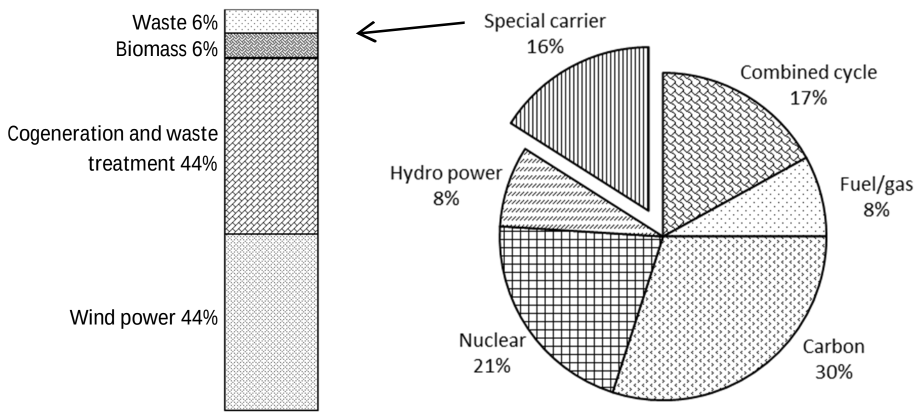

Figure 3.5. Electricity breakdown (Unión Fenosa, 2006).

All this information was introduced in a spreadsheet in MS Excel® where all the calculation steps took place, together with other parameters required like natural productivity (*NP_i*), embodied energy (*EV_i*) and energy productivity (*EP_i*). The input variables were organized by main categories, i.e. energy, resources and waste.

For the development of the tool only four of the general land types included in EF studies were considered: carbon land, arable land, pasture land and forest. The arable and pasture land are mainly associated to the materials employed in the manufacture of the jackets (i.e. cotton and wool), although other contributions may arise from primary energy (as a result of the electricity breakdown), like solar energy or biomass for arable land, and hydro power or wind power in the case of pasture land. Meanwhile, the area for fossil energy refers to the land required for the absorption of CO_2 emissions derived from the direct consumption of the different sources of energy indicated in table 1, as well as from the embodied energy of the manufactured materials employed (i.e. plastic and metal accessories) and the electricity breakdown. Finally, forest is necessary to provide resources such as paper or cardboard, or wood for certain accessories. No sea resources were consumed and the built-up land was excluded as not considered to be part of an indicator tracking for the environmental performance of the plant. The equivalence factors employed are indicated in Table 3.4.

In the case of the waste category, the recycling rate can be specified as an additional parameter, thus allowing for energy and material savings and, consequently, for a reduction in the total EF of the process.

For each row a total EF expressed in gha is computed, and then the results are summarized by main category and for the total EF of the process, expressing the percentages of contribution in each case. Finally, a relative indicator (EF_r) is estimated dividing the total EF by the number of items of clothing produced in the year considered (P_{year}). This allows for making comparisons between different years and also for benchmarking among different plants belonging to the same or other companies.

Table 3.4. Equivalence factors (F_j) used to normalize and homogenize the different kinds of land (Wackernagel et al., 2005)

Land category	**Equivalence factor**
Carbon land	1.4
Arable land	2.1
Pasture land	0.5
Forest	1.4

In this chapter, performing data of the analyzed tailoring plant were compiled from 2002 to 2005 and the EF was estimated for these years. Embodied energy values, natural and energy productivity indexes and equivalence land factors were extracted from different works (Rees and Wackernagel, 1996; Wackernagel, 1998; Wackernagel et al., 2005; Doménech, 2006).

3.3. Results and discussion

The results presented here show the suitability of a new approach for the application of EF to an enterprise (a dressmaking factory). The aim was to develop a tool for evaluating the environmental impact evolution due to the performance of the plant. Furthermore, a simple and wide understandable indicator for giving information of sustainability, useful for a comparative analysis in a CSR framework, was chosen.

3.3.1. Inventory

The inventoried data were those from the manufacturing process (Figure 3.4). Most of the information used came from Sustainability Reports and data directly inventoried in the plant. The raw materials were used in the tailoring and packaging of the jackets. The paper and plastic consumption for 2002 and 2003, as well as the waste generated, were estimated based on production rates in order to obtain complete series for the four years studied. The number of jackets produced rose in the last two years analyzed, with a consequent increase in energy requirements. Thus, in spite of introducing own renewable energy sources (wind power) the external electric energy supply has gone on increasing. The wind-power energy comes from a direct source of the plant, as the company has a wind turbine (1.5 MW of nominal power) in its productive center in Arteixo which supplies electricity to the manufacturing plants. The gas-oil and the propane were used in cogeneration units, in which air emissions were released. SO_2 emissions were estimated through gas-oil consumption and air emission factors (US EPA, 1984), considering 0.2% sulfur content (Spanish Government, 2005). Thus, these emissions showed an equal tendency to the gas-oil burnt up. Reduction in NO_x and CO emissions is more remarkable in 2004 than in 2003. In 2005 emissions increased again, as well as the gas-oil consumption did. CO_2 emissions show the same evolution that the electric energy consumption, which was the main energy source of the plant. Energy facilities are common for a series of dressmaking factories belonging to the same company; therefore, the specific consumption for the studied factory was allocated on the basis of production rates. Hazardous waste was mainly generated in maintenance works.

Despite there were some gaps, inventory data provided by the company were comprehensive enough to accomplish an approach of the tool (Table 3.5).

Table 3.5. Process inventory data.

			2002	2003	2004	2005
INPUT	Raw materials	Cotton fabric (kg)	643,402	651,881	798,199	919,504
		Stitch (kg)	-	-	15,800	35,500
		Lining (kg)	-	-	300,000	350,000
		Paper & cardboard (kg)	[a]5,867	[a]5,740	6,971	7,173
		Plastic (kg)	[a]32,153	[a]31,459	24,419	39,313
		Buttons (kg)	28,000	28,000	28,000	31,864
		Zips (kg)	13,500	8,100	6,300	7,164
		Labels (kg)	650	650	650	740
	Energy	Electricity (kWh)	236,193	210,660	322,059	386,621
		Wind power (kWh)	0	8,980	14,711	15,244
		Propane (kg)	0	96.3	123.9	133.9
		Gas-oil (m^3)	61.924	35.470	19.547	34.054
		Water (m^3)	777.5	160.9	110.3	124.6
OUTPUT		Production (number of items)	519,399	508,188	558,078	635,055
	Air emissions	SO_2 (kg)	[a]575	[a]330	[a]182	[a]316
		NO_x (kg)	18,194	3,542	3,554	6,086
		CO (kg)	11,529	11,502	3,652	4,623
		CO_2 (kg)	261,901	184,975	196,896	262,527
	Similar to urban waste	Textile (kg)	81,765	83,353	104,632	119,065
		Paper & cardboard (kg)	[a]5,867	[a]5,740	6,971	7,173
		Plastic (kg)	[a]605	[a]592	660	740
		Urban waste	-	-	-	-
	Hazardous waste	Paint (kg)	-	-	-	1.185
		Batteries (kg)	1.492	14.967	4.825	2.378
		Fluorescent lamp (kg)	11.114	5.443	13.669	6.817
		Computers waste (kg)	-	3.402	3.586	92.265
		Oil filters (kg)	60.719	11.566	7.706	4.756
		Mineral oil (kg)	104.430	115.658	100.823	-
		Contaminated containers kg)	0.746	1.565	4.594	3.171

[a]Estimated values

3.3.2. EF estimate

According to the methodology explained in Chapter 2 and using the developed tool, the EF was assessed with inventory data for the dressmaking plant (Table 3.5). In a first approach the EF was obtained considering the use of synthetic stitch material together with the cotton fabric for the manufacture of the jackets in 2004 and 2005, and without considering the recycling of waste (Figure 3.6).

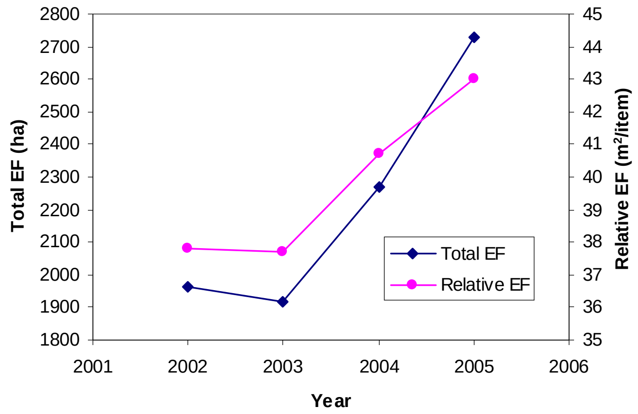

Figure 3.6. EF estimates considering cotton and synthetic stich in the jackets design and no recycling of waste.

An increasing tendency since 2003 was observed, both for the total and the relative ecological footprint (considering the number of items produced per year). The contribution of each category to the total EF was also determined, observing the high influence of the cotton textile (Table 3.6). The area required for its natural production was the main cause of the high values obtained.

As it was stated in the methodology section, the built-up area was not included in the spreadsheet since it did not influence the performance of the plant. Besides, the plant has an extension of 0.63 ha; therefore, the influence of this component would be negligible in the total EF.

Table 3.6. Contribution (%) of the considered categories to the final year 2005 EF estimate for synthetic stitch and cotton jackets and no considering recycling.

Energy category	**5.32**	**Resources category**	**91.33**	**Waste category**	**3.35**
Carbon	1.30	Plastic	12.30	Paper & cardboard	0.28
Liquid fuel	3.06	Paper & cardboard	0.52	Plastic	0.02
Gas fuel	0.63	Cotton textile	77.38	Textile	3.04
Nuclear	0.21	Synthetic stitch	1.12	Urban waste	-
Hydroelectric power	<0.01	Wool stitch	-		
Wind power	<0.01	Wood	-		
Solar energy	<0.01	Metal	-		
Biomass	0.12	Water	<0.01		

The EF values obtained were not very high in comparison with those found in other studies published by the time this work was carried out. For example, a total EF of 2,841 gha was estimated for the Lions Gate Hospital (Germain, 2001) and values near 6,500 gha were appraised for the Port of Gijón (Doménech, 2004). The balance of the footprint of the process would require the investment in ecosystems conservation, reforestation (not only considering fast-growing species but also respecting biodiversity and the equilibrium of the ecosystems), etc. However, there was no counter footprint contribution to calculate the NEF. Enterprises investments in natural capital would not only reduce the EF and supply the means for an ecological development, but it would also contribute to the fulfillment of the Kyoto Protocol. Furthermore, these actions would be accompanied by the creation of new local employments, thus including a social component in the EF estimate as sustainable development should combine the ecological and social matters (Huberg, 2000).

Sensitivity analyses were carried out for determining the way each category influenced the EF, by incorporating different materials into the composition of the jacket, varying the source of energy or introducing percentages of recycling for the waste. Furthermore, limitations of the EF found for the studied case, as well as the usefulness of this indicator, are discussed next.

3.3.3. Resources contribution to the EF

This category was the principal contributor to the Ecological Footprint. The type of the material used could change from one year to other, depending on fashion tendencies. Therefore, a simulation changing synthetic stitch by wool stitch in 2004 and 2005 was carried out to evaluate the influence of using different materials. A more noticeable increase was obtained in this case than the one observed when considering synthetic stitch. The EF values were 16.9% and 31.6% higher for 2004 and 2005, respectively. The increase in 2005 was almost twice higher than in 2004, following a close relationship with the increase in the amount of stitch material consumed in 2005 with respect to 2004 (2.2 times higher). This reflected the great influence the manufactured materials employed had on the EF value. This is also illustrated in Table 3.7, which reflects the contribution percentages for all categories for the 2005 EF estimate when considering wool stitch.

Table 3.7. Contribution (%) of the considered categories to the final year 2005 EF estimate for wool stitch and cotton jackets and no considering recycling.

Energy category	**4.04**	**Resources category**	**93.42**	**Waste category**	**2.54**
Carbon	0.99	Plastic	9.35	Paper & cardboard	0.22
Liquid fuel	2.32	Paper & cardboard	0.39	Plastic	0.02
Gas fuel	0.48	Cotton textile	58.78	Textile	2.31
Nuclear	0.16	Synthetic stitch	-	Urban waste	-
Hydroelectric power	<0.01	Wool stitch	24.89		
Wind power	<0.01	Wood	-		
Solar energy	<0.01	Metal	-		
Biomass	0.09	Water	<0.01		

In this case, the wool had a weight of 25.0% in the total EF, while the value for the cotton decreased from 77.4% to 58.8%. The wool mainly contributed to the required pasture land, while the cotton mostly influenced the needed arable land. Farms, regardless of their dairy or crop function, are intensive operations that impact the environment (Beynon et al., 2002). In addition, the materials here obtained are later treated in order to obtain the fabric ready to be tailored. Consequently, their EF is much higher than the corresponding to synthetic ones.

Other materials (metal and wood) were included yet not filled in the spreadsheet, since it was unknown what buttons or zips were composed of. These boxes were kept back for later studies when the inventory data would be more comprehensive. As an example, if all buttons in 2005 were considered to be made of metal, the EF would increase in 51 ha; if they were supposed to be made of wood the increment would be of 33 ha; finally, the lower augment occurred when they were considered to be plastic buttons. Anyway, this would not change the total EF in more than 2%.

3.3.4. Energy contribution to EF

In all cases, the resources main category represented more than 90% of the total EF. This poses a difficulty when assessing the influence of changes in either the amount or the sort of energy used, which can be overcome analyzing the energy main category separately. Thus, percentages of contribution within this group, which would not depend on the suppositions made in the materials used, were calculated (Figure 3.7). Though, it must not be forgotten that the input materials were already manufactured, and their contribution to the EF involved not only the land associated to the natural productivity (like for cotton or wool) but also the energy necessary for their subsequent elaboration.

Liquid fuels consumption contributed nearby a 57% to the total, mainly due to the consumption of gas-oil. Meanwhile, a low contribution of the so-called renewable energies was observed. As stated in the inventory section, only the wind-power energy came from a direct source of the plant. The hydroelectric power, the biomass and the nuclear categories came from the default breakdown of the electric energy into its primary sources, according to the power supplier company's rates (Unión Fenosa, 2006). Consequently, the choice of the electricity supplier company based on sustainability criteria would lead to select the one with highest renewable energies contribution, thus decreasing the EF of the tailoring plant. These different options could be possible in Spain, the second country in the world with highest installed wind power, despite the renewable energies only represented the 7% of the total primary energy consumed by the time this work was developed (IDAE, 2007).

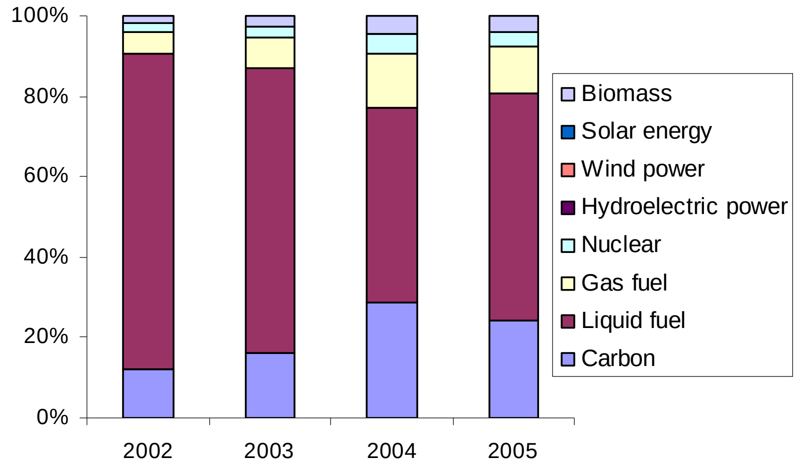

Figure 3.7. Analysis of the contributions to the EF within the energy category.

To assess how changes in energy sources (Δ Electricity, Δ Wind Power, Δ Propane, Δ Gas-oil) could affect the EF estimate, sensitivity analyses were carried out. The results obtained are shown in Table 3.8. It was observed that changes in electricity and gas-oil supply had the major effect in the energy component of the EF (energy EF in Table 3.8). However, the consequences were not very noticeable in the total EF, as it was shadowed by the high weight of resources consumption.

Table 3.8. EF sensitivity to a 10% increase in energy sources, considering no recycling and synthetic stitch (data related to 2005).

EF (gha)	**Initial**	**Δ Electricity**	**Δ Wind Power**	**Δ Propane**	**Δ Gas-oil**
Energy EF	145.1	152.0	145.1	145.2	161.1
Δ EF (%)	-	4.8	<0.01	0.07	11.0
Total EF	2,730	2,736	2,730	2,731	2,746
Δ EF (%)	-	0.22	<0.01	0.04	0.59

3.3.5. Waste contribution to EF

Based on the inventory data, four entries were defined within the waste main category (Table 3.3). The high amount of fabric used in the process and the waste generated were especially important, hence being the main contributor to the area associated to waste assimilation. There were not available data for urban waste, but the entry was included in the spreadsheet for future studies.

The cutting of the fabric according to the given pattern is an automatized process. The machine establishes how the different pieces that compose the jacket should be placed to minimize the discard of fabric (around 15%). Consequently, given that this is an already optimized process, diminishing the amount of fabric waste in the factory to lower levels is quite a difficult task. Nonetheless, a good alternative for reducing the waste impact on the environment is recycling. A high decrease in the waste contribution to the EF was observed when the similar to urban waste generated in the plant (textile, plastic, paper and cardboard) were considered to be recycled (Table 3.9). Since the weight of waste is very low in the total value of the EF, these results were not very noticeable in the overall estimate (a decrease of 2.0% when considering the recycling of the wastes altogether).

Table 3.9. Influence of waste recycling on EF considering synthetic stitch (data related to 2005).

Category	EF with recycling (gha)[a]	EF without recycling (gha)	Reduction percentage
Paper & cardboard	2.5	7.3	65.8%
Plastic	0.2	0.5	60.0%
Textile	22.4	83.5	83.3%

[a] 100% of the similar to urban waste generated in the plant is recycled.

3.3.6. EF usefulness as environmental indicator

In the previous sections, it was shown how the EF was sensitive to changes in the materials employed in the manufacturing of the jackets, as well as in the kind of energy sources introduced in the process. This means that this indicator is suitable to effectively assess the environmental performance of different competing management and manufacturing options that may be considered in an industrial production process.

The major benefit is that a great amount of handled information is synthesized and expressed in a way easier to communicate than that derived from the application of other methodologies used with similar purposes like Life Cycle Assessment (LCA). The EF would also allow people to relate the documented ecological demand to the biosphere's regenerative capacity (Wackernagel and Yount, 2000), since this indicator constitutes a good way for accounting the natural capital. Consequently, it could also be helpful in determining the ability of an industrial system to adapt to the local natural limiting factors (Kratena, 2008).

Another advantage of the EF in comparison to LCA is the absence of a requirement for an exhaustive data collection, as it is necessary for a complete LCA. Especially in the dressmaking process, where the input of the plant is not composed by raw materials but by manufactured ones (fabric, plastic, etc.), a simplified tool is demanded and therefore, the use of the EF could be much more appropriate. Besides, to analyze in depth the environmental impact of the process via LCA it would be necessary to start studying all the processes involved in the production of these input materials. This is a more time consuming task which would imply a higher effort, even supposing that all necessary data could be available (Curran, 2004).

All of the above mentioned does not mean that the EF is a more powerful instrument but that it is a more interesting one when the attention of the study is focused in a more general and less-in-depth analysis. A good measure of the sustainability associated to production changes can be obtained in a simplified and quicker way, so that environmental supporting information is available for decision making at process level (Wackernagel and Yount, 2000; Venetoulis and Talberth, 2007).

3.3.7. EF limitations

Nowadays, the analysis of the total environmental impact using the Ecological Footprint remains slightly incomplete since it does not take into account other emissions released by the combustion of fossil fuel but the carbon dioxide, or some other contaminants like hazardous waste, heavy metals or dyes (Moberg, 1999). The reason is that they do not have a closed cycle in biosphere. Thus, some of the inventoried data could not be included in the EF estimate.

From the inventory presented in Table 3.5, it can be observed that emissions other than CO_2 and hazardous wastes were not treated during the development of the tool or the discussion of results. As established in Chapter 2, the method selected for the estimate of the EF was the one based on the area required for the absorption of CO_2 emissions released during the combustion of fossil fuels. First EF assessments used an absorption factor of 1.8 t C/(ha·yr) (Rees and Wackernagel, 1996). Later studies, based on IPCC estimations, yielded to a factor of 1.42 t C/(ha·yr) (Doménech, 2006). Oliveros et al. (2004) confirm an absorption rate up to 25 t CO_2/(ha·yr) for *Eucalyptus*, the third most important species covering Galician forests (it is the dominant species in 174,210.40 ha and in 159,413.93 ha together with the *Pinus pinaster*), and the one with major presence in the surroundings of the factory (Galician Government, 2001). The most conservative rate was used in the spreadsheet. Thus, the CO_2 released in the cogeneration units were accounted by means of the different sources of energy employed.

Nonetheless, other emissions were recorded by the factory, namely CO, SO_2 and NO_x. The original EF methodology did not provide a means to include them; consequently, their environmental load could not be evaluated and was not included in the indicator. If emissions other than CO_2, like the acidifying ones, could be included in the tool, then it would not only be noticeably sensitive to changes in the material used but it would also be useful for evaluating the effects of changing the sources of energy (e.g. avoiding the use of gas-oil and thus reducing the acidifying emissions).

Hazardous waste was also generated in the plant. According to the idea previously exposed, it was not included in the EF estimate as it could never take part of a closed cycle under a sustainable development approach. However, it represented less than 0.25% of the total waste and it was properly managed and treated following legal constraints. Therefore, the damage to the environment was supposed to be under control and minimized.

3.4. Conclusions

The increasing development of the textile sector in Galicia has situated it among the most remarkable positions of the industry in this region. For this reason, it is important to develop a tool that allows for the measurement of its environmental impact. A tailoring plant, part of the productive textile chain, where cotton jackets were manufactured, was studied.

As part of a company that elaborates a CSR Report, simple sustainability indicators easy to understand are desirable to be used (GRI, 2006). In addition, only the impact due to the performance of the plant was analyzed. Thus, it was considered that the EF was the concept that better fit with this purpose, since the already manufactured inputs to the factory can be incorporated directly, without requiring the analysis of their own production processes.

The study has been carried out for the period 2002-2005. The results showed a continuous increase of the EF throughout the years. The overall EF value was strongly influenced by the resource category. The main contributors within this group were the cotton and the wool needed to manufacture the jackets. This means that changes in fashion tendencies will noticeably affect this category, depending on the materials incorporated to the design.

A small contribution to the total EF was obtained for the energy category. However, if the emissions released in the factory were included in the EF account, the influence of the sources of energy would be more noticeable and thus the EF would also be an interesting index for this category. Furthermore, the selection of an electricity supplier company with larger renewable energy contribution was pointed out as another way of reducing the EF.

It was shown that the EF is an environmental sustainability indicator that can be used in industrial processes (dressmaking plant). Some limitations were found as the EF does not include some of the environmental loads that can be found in the textile sector. As a recent methodology, the EF is continuously suffering from improvements that seek the consistency of the indicator. As an answer to the drawbacks detected during the work carried out in this chapter, some effective solutions were suggested in subsequent research, comprising methodological developments and the proposal of integration with other environmental evaluation tools. These advances are exposed in other chapters of this thesis.

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Chapter 4

Ecological footprint of wastes

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Abstract

The Ecological Footprint (EF) is a widely used indicator to assess the sustainability of people, regions or business activities. Although this metric has grown in interest and popularity over the years, it has also been the subject of criticism and controversy. The advantages of an aggregated indicator are often overshadowed by the shortcomings of its corresponding methodology. Hence, the application of the EF to a production process in Chapter 3 revealed that a complete measure of the environmental impact of the activity could not be provided. As a response, later research was focused on finding solutions to overcome the major critiques of the indicator, which prevented from considering the EF as an appropriate and useful indicator to be used at process or industrial level.

One of the flaws of the EF methodology more frequently treated in the literature is the fact that it does not account for toxic or hazardous pollutants and wastes, which cannot be part of a closed biological cycle. This poses a major problem when evaluating the environmental burdens of an industrial process, where these kinds of flows are expected to be generated.

The methodology developed in this chapter estimates the EF of toxic and hazardous wastes considering a closed cycle modeled through a plasma process; a phenomenon that naturally occurs in stars and volcanoes. Wastes from industry can be treated in a thermal plasma gasification process, and, by developing a methodology base on a model to describe this process, the EF of hazardous wastes was calculated. The developed methodology was tested with the case study shown in Chapter 3, both for hazardous and non-hazardous wastes generated in the tailoring factory. For the latter a value of 56.5 gha was obtained, a figure on the same order of magnitude as that obtained in Chapter 3 where the conventional ecological footprint methodology was applied.

4.1. Introduction

The need to report environmental behavior, for both legal and ethical reasons, has led to the proliferation of a wide variety of indicators in recent years. The Ecological Footprint (EF) is one of the most popular indicators among those that use territorial or natural resource units (namely, ghost acreage, environmental space, ecological rucksack, energy analysis and water footprint). The EF is a sustainability indicator that estimates the amount of bioproductive land required to produce resources and absorb wastes in a given system. In recent years, the EF has been applied in a variety of fields, i.e. policy-making, production processes, environmental evaluations and research projects (Niccolucci et al., 2008; Stowglehner and Narodoslawsky, 2008). The European Union (EU) is considering the use of this metric to measure the sustainability of natural resources (Best et al., 2008). However, shortcomings in the methodology behind the EF calculations have been reported, and there is a need for further improvement before the EF can provide a reliable global assessment (Kitzes et al., 2009a). Thus, despite its heightened popularity in recent years, the EF has been the subject of criticism and controversy (Ayres, 2000; Ferguson, 2001; Van Vuuren, 2001; Fiala, 2008; Kitzes et al., 2009b). As such, new and alternative EF methodologies continue to be studied by various authors (Stöglehner, 2003; Nguyen and Yamamoto, 2007; Stoeglehner and Narodoslawsky, 2008; Venetoulis and Talberth, 2008). These contributions, such as the one proposed here, will help to improve the appraisals obtained by the EF, although it will not perfect them. In the meanwhile, the indicator must be used cautiously; one must remain aware of the limitations implicit in the estimates, but also take advantage of its integrated nature.

A closed cycle, generated by the application of thermal plasma technology into the biological cycle, was proposed so that the requirements for EF conversion could be accomplished. Thus, a methodology to assess the EF of wastes, including both hazardous and non-hazardous wastes, was developed on this premise. Nonetheless, this does not mean that plasma treatment was considered as a panacea to deal with waste management problems; however, it was employed as a closed cycle model existing in nature for methodological purposes.

4.1.1. Closed biological cycles in EF calculations

Given that EF calculations only account for materials with an implicit biological productivity (for resources) or absorption rate (for wastes), there are many consumption inputs and pollutant outputs that are excluded from such estimations. This means that, for example, materials, such as plastics, that are neither created by biological processes nor absorbed by biological systems do not have an EF (Kitzes and Wackernagel, 2009). In some cases, however, there may be a specific assimilation rate, as for acidification emissions (Holmberg et al., 1999), but including such factors could result in an overestimation of the EF. Hence, when pollutants are considered to have an insignificant assimilation capacity in the biosphere, they are discarded from the EF calculations (Kitzes et al., 2007). Consequently, the EF should be considered as an indicator of minimum criteria (Kitzes et al., 2009b), i.e. if the calculated area (assuming and acknowledging underestimation) exceeds the available carrying capacity, then unsustainability is ensured. Otherwise, other factors that can degrade natural resources should be assessed.

Particular analyses must deal with global warming emissions. Generally, only CO_2 emissions are computed in EF estimations. For CO_2 emissions, a sufficiently accurate method is available for calculating the land area required to absorb them, but this is not the case for the other greenhouse gases (Best et al., 2008). Yet, the Carbon Footprint aggregates various global warming emissions and expresses them as carbon dioxide equivalents; for a full EF calculation, this data is translated into the area required to absorb these carbon emissions in units of global hectares (Kitzes and Wackernagel, 2009). This approach implies that CO_2 absorption rates are the same as the other greenhouse gases, such as CH_4 or N_2O. Calculating CH_4 emissions based on their global warming potential (GWP) would produce significantly different results than if a mass carbon transformation was considered. Hence, methane, which has a GWP factor that is 25 times that of CO_2, does not necessarily require 25 times the land area to sequester its carbon content (Walsh et al., 2009).

Eder and Narodoslawsky (1999) proposed an indicator called the Dissipation Area Index (DAI), which was designed as a tool for evaluating the flow of material from the technosphere to the biosphere. Given its strong relation to land occupation, the DAI is considered as a type of ecological footprint; in fact, it is an improvement on the EF methodology (Stoeglehner and Narodoslawsky, 2008).

The biosphere can only absorb limited output flows from the technosphere without suffering irreversible damage. Employing this assimilation capacity concept, previous reports calculated the dissipation area from the natural concentration of the substance in soil and the rate of soil replenishment. Thus, a number of dissipation areas are available for chemicals such as nitric oxides and lead in air and nitrates and copper in water (Eder and Narodoslawsky, 1999).

Kitzes and Wackernagel (2009) distinguish between the EF of a toxic material and the lifecycle EF stemming of the other biological materials extracted from the biosphere for the toxic material's production, and claim that these two concepts are often confused. Thus, from a lifecycle perspective, apart from the EF associated to the extraction of the original biological materials, other impacts like the carbon fossil emissions released during the production process or the physical area where the plant is built have to be considered. On the other hand, one can also consider the embodied energy, which is the energy used during a product's entire life cycle in order to manufacture, transport, use and dispose of the product (Kitzes et al., 2007). This concept is used in EF calculations to convert manufactured goods into their energy equivalents, using the best data available on the energy intensity of various goods. This lifecycle-based perspective is used for EF component approaches (Monfreda et al., 2004).

The development of new methods to incorporate traditionally excluded issues into EF appraisals is a result of the need to construct a better composite indicator. This means that while it is not plausible to assume the same EF levels for nuclear energy as for fossil fuels, it is also not accurate to exclude nuclear EFs in National Footprint Accounts (NFA), which could lead to the misinterpretation that nuclear powered countries necessarily have a higher ecological performance (Monfreda et al., 2004; Best et al., 2008; Ewing et al., 2008). A proposal to improve the EF of nuclear energy was made by Stoeglehner et al. (2005); from a life-cycle perspective, the authors considered the area associated with uranium mining, nuclear power plant accidents, nuclear transport and nuclear disposal. Even for an underestimation scenario, the footprint per energy was more than five times of that used in the EF estimations that did not consider the nuclear energy in the calculations.

Wastes from industry can be treated in a thermal plasma gasification process, a phenomenon that naturally occurs in stars and volcanoes as well. The methodology developed in the present work estimates the EF of toxic and hazardous wastes by closing their biosphere cycle and considering their transformation in a plasma process.

4.1.2. Thermal plasma technology fundamentals

Thermal plasma technology, which emerged in the Nineties, has received a great deal of interest for its ability to treat mixed forms of waste. It can be applied to solids, liquids or gases. Because high temperatures can be reached, it can be used for different applications, such as the destruction of organics or the vitrification of hazardous waste (Chu et al., 1998). Thus, plasma treatment is ideally suited for toxic wastes and complex waste streams that have recoverable energy content. The high temperature of the plasma arc greatly reduces the amount of undesirable by-products that are generated (Vaidyanathan et al., 2007).

A simultaneous dual reaction process takes place in a plasma reactor: the organic compounds are thermally decomposed into their constituent elements (syngas with more complete and advantageous conversion of carbon into gas than in incinerators), while the inorganic materials are melted and converted into a dense, inert and nonleachable vitrified slag, which does not require controlled disposal (Figure 4.1). Therefore, it can be viewed as a totally closed treatment system (Tendler et al., 2005).

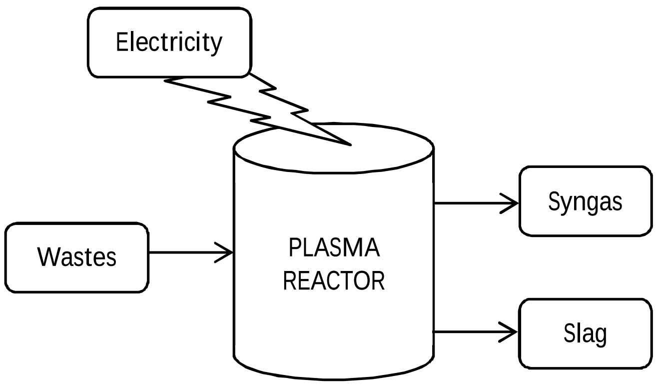

Figure 4.1. A simplified scheme of the plasma technology process, indicating the main input and output flows.

Syngas (mainly composed of CO and H_2) can be used to generate electrical power and produce valuable hydro-carbonic acids. Vitrification is the result of the interaction between the plasma and inorganic materials. Because the inert fraction is vitrified and harmful substances can barely leach from the lava, this product can be used for road construction or as a building material (Tendler et al, 2005). In addition, plasma can induce the thermal decomposition of toxic molecules into simpler ones that are benign (e.g., the CN molecule can be broken down into the elements C and N).

The application of this kind of technology to the treatment of hazardous wastes has been explored by different authors, and prior research includes studies on steel plant dust (Schoukens et al., 1993), nuclear waste (Lemont and Hugues, 2008), hazardous medical waste (Chu et al., 1998; Rutberg et al., 2002), tannery waste (Hetland et al., 2003) and organic wastes (Huang and Tang 2007).

4.2. Methodology

In this section, it is explained how the plasma technology was considered to close the cycle of wastes, as well as the calculations that were conducted to translate the input and output flows into EF values.

4.2.1. The application of plasma technology as a closed cycle model for calculating the EF of wastes.

It was assumed that, in a simplified approach, the thermal plasma process closes the waste cycle in the biosphere due to the fact that the combusted syngas returns to the biosphere via CO_2 absorption in forests and oceans, and the vitrified material returns to the production cycle as new input material. Hence, three main factors were computed in the estimates (Figure 4.2):

- The balance between the electricity consumed by the process (waste pre-treatment, plasma torching and syngas cleaning) and the energy generated in the combined cycle.
- Carbon emissions in the combined cycle, which are a result of the syngas combustion.

- The counter-footprint associated with the recovery of inorganic material in the slag, avoiding the extraction and manufacture of new raw materials.

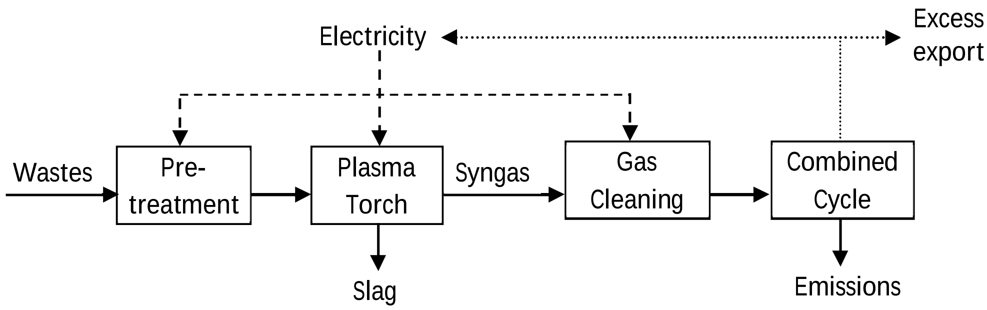

Figure 4.2. The operational units used in the thermal plasma process and their implications for the EF estimate.

In addition, the area occupied by the process itself was considered to be built-up land. However, due to the scarcity of data found in the literature on this topic, this term was not included, avoiding a major level of uncertainty in the calculations. The possible contribution of this factor is discussed later on.

4.2.2. Correlations

Using data from the literature about input and output flows in thermal plasma treatments of different kinds of wastes, correlation functions were established between the consumption and generation flows of the process and the carbon content percentage, which was selected as characteristic parameter of the waste. SPSS Statistics 17.0® software was used for this purpose, as well as for conducting a statistical analysis of the correlations obtained to explore their significance and reliability.

The electricity consumed by the plasma torch and the auxiliary units used in the process were directly found in the literature for different kind of wastes; then, the carbon content was estimated and correlations were established. The literature provided data on the electricity generated in the combined cycle, but in some cases, it was necessary to estimate the electricity generated by syngas heating considering the efficiency of conversion in the combined cycle power plant.

The net electricity balance can be expressed as follows:

$$E_N = E_P + E_A - E_c \qquad [4.1]$$

where E_N is the net external electricity demand (from a power supplier); E_P is the electricity consumed by the plasma torch; E_A is the electricity consumed by the auxiliary operational units; and E_C is the electricity generated in the combined cycle. All of these terms are expressed in relative units, that is, the energy per unit mass of waste treated. According to equation [4.1], if the electricity consumed by the plasma torch and the auxiliary units exceeds the electricity generation in the combined cycle, then a positive balance is obtained and external demand from a power supplier is required. Otherwise, for high energetic wastes, this balance may result in a negative net value, which indicates that a surplus of energy is generated and it can be exported -as contemplated in Figure 4.2- (Schaffner, 2000).

The carbon emissions from the combined cycle depend on the composition of the syngas obtained in the plasma process. Generally, the predominant components are CO and H_2, but in some cases, the syngas can contain small quantities of CH_4. Thus, to calculate the total carbon emissions, three components were taken into account:

- The CO_2 produced via CO oxidation
- The CO_2 produced via CH_4 oxidation
- The CO_2 initially present in the syngas

For the first two terms, a general oxidation factor of 0.995 for the gaseous fuels was used (European Commission, 2004). This means that the effect of the technology employed was also considered, apart from the carbon content in wastes.

Additionally, a correlation was obtained to express the slag production (per unit mass of waste treated) as a function of the carbon content of the wastes. This slag generated was directly reported in the literature for different kinds of wastes fed into the thermal plasma process.

4.2.3. EF estimation

Plasma technology allows for the recuperation of the energy contained in wastes. This is represented by a counter-footprint term in the electricity balance equation.

The amount of energy consumed by the conversion process should also be taken into account. Hence, the first term of the EF estimation considered the transformation of the net electricity balance into units of area. In this case, the average electricity breakdown for Spain was considered (Figure 4.3).

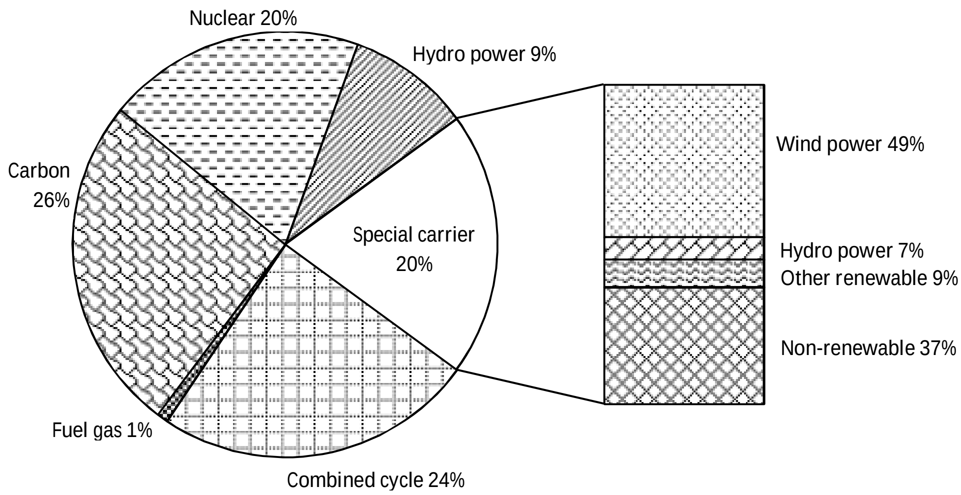

Figure 4.3. Coverage of electricity demand in Spain in year 2007 (SEN, 2007).

Even though the combustion of the syngas in the combined cycle implies external electricity demand savings, it also means that CO_2 emissions are released by the plant itself. The associated EF was measured by considering the area needed for the absorption of these emissions. An absorption rate of 1 t C/(ha·yr) was used according to the Living Planet Reports. For the vitrified slag, the counter-footprint was calculated on the basis of the energy (fossil) saved, because there was no need to manufacture new raw materials (i.e., inert construction materials). This assumption implies that all the slag generated will be reused, and that, consequently, no surplus slag will be stored. However, this may be over-estimation of the counter-footprint assigned to this material given that the market may not absorb all of it. The conversion of all of these terms into EF units was carried out using a spreadsheet (Microsoft Excel®), in a manner similar to the method employed in Chapter 3.

The final EF of the wastes was calculated using the following equation:

$$EF_{wastes} = EF_{electricity} + EF_{carbon\ emissions} - CF_{slag} \quad [4.2]$$

where EF_{wastes} is the total EF estimated for the wastes; $EF_{electricity}$ is the contribution from the net electricity balance, which may result in a positive or negative term; $EF_{carbon\ emissions}$ is the area required to absorb the CO_2 released in the combined cycle; and CF_{slag} is the counter-footprint associated with the slag production.

4.3. Results and discussion

First, correlations among the parameters used in the EF calculation were determined using data from the literature. Once these equations were obtained, the final model of the EF estimation of wastes was constructed using equation [4.2].

4.3.1. Correlations

The correlations based on data found in the literature are presented in this section, which discusses the electricity consumption by the plasma torch, electricity generated by the combined cycle, carbon emissions and slag production. It was not possible to establish a correlation for the electricity consumption in the auxiliary operations because of a lack of data, which will be explained later. The wastes considered in this chapter were characterized by their carbon content, as presented in Table 4.1. In fact, higher carbon content means that more energy will be provided by the combined-cycle power plant (Carabin and Gagnon, 2007).

Table 4.1. Data used for the estimation of carbon content in wastes.

Waste type	Description	Carbon content (%)	Source
Fly ash (1)	Fly ash from bark boiler	27.00	Carabin and Gagnon, 2007
Fly ash (2)	Fly ash from coal fired power boilers	5.00	Carabin and Gagnon, 2007
EAF dusts (1)	Electric-arc furnace dusts from metallurgical industry with additional carbon content of 75 kg per ton of dust.	6.98 [a]	Shoukens et al., 1993
EAF dusts (2)	Electric-arc furnace dusts from metallurgical industry with additional carbon content of 170 kg per ton of dust.	14.53 [a]	Shoukens et al., 1993
Alloy-steel dust	Alloy-steel dust from electric-arc furnace with additional anthracite (77.4% C) in a rate of 290 kg per ton of dust.	17.40 [a]	Shoukens et al., 1993
RDF (1)	Refuse Derived Fuel (common)	45.90	Baggio et al., 2008
RDF (2)	Refuse Derived Fuel (known composition)	36.10	MPM Tech., 2005
Wood (1)	Common wood	50.00	IPCC, 2006
TDF	Tire Derived Fuel	79.87	Figueroa et al., 1997
Rubber		79.87	Figueroa et al., 1997
Polypropylene		86.10	Baggio et al., 2008
Tannery waste	Considering the carbon content for leather	54.90	Bahillo et al., 2004
Carpet waste		*52.17*	*Weighted average*
Carpet fiber	2/3	59.10	Lemieux et al., 2004
Carpet fines	1/3	38.30	Lemieux et al., 2004

[a] Calculated for the final mixture fed to the plasma reactor.

Table 4.1. Data used for the estimation of carbon content in wastes (*cont.*).

Waste type	Description	Carbon content (%)	Source
USAF BEAR waste	United States Air Force Basic Expeditionary Airfield Resources Base waste	*52.25*	*Weighted average*
Non-durable paper	31.0%	46.00	IPCC, 2006
Cardboard	22.5%	46.00	IPCC, 2006
Non-durable plastics	14.5%	75.00	IPCC, 2006
Durable plastics	11.0%	75.00	IPCC, 2006
Rubber	2.0%	75.00	IPCC, 2006
Textiles	3.5%	50.00	IPCC, 2006
Glass	2.0%	0.00	IPCC, 2006
Metal	4.0%	0.00	IPCC, 2006
Wood (1)	3.0%	50.00	IPCC, 2006
Vinyl & Styrofoam	3.5%	75.00	IPCC, 2006
Food	3.0%	38.00	IPCC, 2006
Fly ash (3)	Fly ash with a carbon content of 1.7% treated in an oxidizing atmosphere	1.70	Haugsten and Gustavson, 2000
Fly ash (4)	Fly ash with a carbon content of 3.1% treated in an oxidizing atmosphere	3.10	Haugsten and Gustavson, 2000
Wood (2)	Wood with known composition	44.38	Zhao et al., 2001
Medical waste		51.10	Alvim Ferraz and Afonso, 2003

4.3.1.1. Electricity consumption by the plasma torch

To relate the electricity consumption to the carbon content, the data sets presented in Table 4.1 and Table 4.2 were employed. This data were then converted into homogeneous units, which are shown in Figure 4.4.

Table 4.2. Electricity consumed by the plasma gasification process.

Waste type	**Electricity consumption in plasma torch**		**Electricity consumption in auxiliary units**		**Source**	**Scale**[(a)]
	Value	Units	Value	Units		
EAF dusts (1)	816	kWh/t EAF dusts			Shoukens et al., 1993	T
EAF dusts (2)	1,130	kWh/t EAF dusts			Shoukens et al., 1993	T
Alloy-steel dust	1.08	MWh/t feed			Shoukens et al., 1993	P
Fly ash (1)	367	kWh/t fly ash			Carabin and Gagnon, 2007	P
Fly ash (2)	766	kWh/t fly ash			Carabin and Gagnon, 2007	P
RDF (1)	3.82	MJ/kg feed			Tendler et al., 2005	P
RDF (2)	530	kWh/t RDF			MPM Tech., 2005	I
Wood (1)	3.6	MJ/kg feed			Tendler et al., 2005	P
Wood (1)	325	kWh/t wood			MPM Tech., 2005	I
TDF	6.66	MJ/kg feed			Tendler et al., 2005	P
RDF (1)	4,930	kWh/t RDF	217.8	kWh/t waste	Moraga, 2006	I
RDF (1)	150.0	kWh/t waste	150.0	kWh/t waste	Gallego, 2008	I

[(a)]Scale of application of the study: P = Pilot; I = Industrial; T = Theoretical simulation.

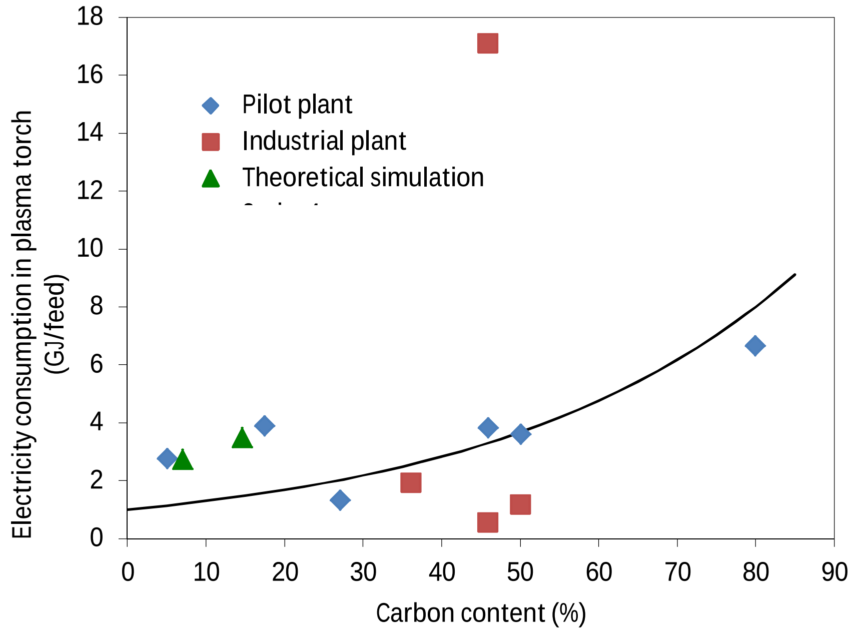

Figure 4.4. Electricity consumption in the plasma torch as a function of carbon content in wastes.

In Figure 4.4, there is a data point that clearly does not follow the general trend. This data point corresponds to a series of industrial source data (Moraga, 2006); actually, for the same waste type (Refuse Derived Fuel –RDF-), another industrial calculation (Gallego, 2008) indicated a much smaller energy consumption. The other industrial values are found at the bottom of the figure and displaced from the general data trend as well. However, for the pilot plant series, a more homogeneous trend was observed; thus, it was decided to construct the correlation only using this data, which showed a good correspondence with the values from a theoretical simulation. Given the variability of data points, different types of functions were tested to obtain the most suitable correlation to fit the experimental data. After conducting a statistical analysis (see subheading 4.3.1.6), an exponential correlation was preferred and equation [4.3] was obtained, which relates the electricity consumption per ton of waste treated to the carbon content of the wastes:

$$E_P = e^{0.026 \cdot C} \quad [4.3]$$

where E_P is the electricity consumed by the plasma torch in units of GJ/t feed and C is the carbon content of the wastes expressed as a percentage.

4.3.1.2. Electricity consumption by the auxiliary operations

It was particularly difficult to find data on the consumption of electricity by the auxiliary operations (i.e., waste pre-treatment, which occurs before the wastes enter the plasma reactor and syngas cleaning). In fact, direct values for the plasma process were only found available for two industrial scale applications; these values were 0.78 GJ/t feed (Moraga, 2006) and 0.54 GJ/t feed (Gallego, 2008).

Given the difference between these applications, the consistency of the data was verified using a comparison with an equivalent incineration process. The ancillary operations considered here include those required for waste pre-treatment processes, which were mainly size homogenization and water content conditioning as well as gas cleaning prior to combustion. Therefore, this part of the process was expected to be comparable for both of the thermal treatment processes. In the work by Grieco and Poggio (2009), a power requirement of 2.45 MW for ancillary units was reported for a waste flow of 4.28 kg/s, which yielded an electricity consumption of 0.57 GJ/t. Meanwhile, the electricity necessary for RDF sorting was 0.051 MJ/t waste, according to Arena et al. (2003). Adding these values, the total electricity consumption is 0.621 GJ/t feed, which is on the same order of magnitude as plasma processes. Thus, an average value of 0.65 GJ/t feed was used in the spreadsheet.

4.3.1.3. Electricity generated by the combined cycle

In most reports in the literature, the syngas heating value and flow rates were indicated, while in a few references the direct value of the electricity generation was provided. All of the data that was collected is summarized in Table 4.3. The electricity values were converted into syngas heating values using the efficiency of conversion in the combined cycle power plant to represent these values in a homogeneous manner. Some references provided specific efficiency values of 26% (Hetland et al., 2003), 34% (MPM Tech., 2005), 40% (Moraga, 2006) and 42% (Gallego, 2008). When a particular efficiency was not specified a value of 40% was assumed because this is an average value of efficiency for combined cycles, leading to lower EF values for wastes (the minimum criteria principle).

Given that the majority of the available data were expressed in terms of the syngas heating, the electricity values were converted so that all of the data could be presented in a homogeneous way (Figure 4.5a). An increase in the electricity generated as the percentage of carbon content in the wastes increased for the pilot plant data was noted; however, the industrial data did not show any clear behavior. For the electricity consumption by the plasma torch, the values from industrial installations did not follow any discernable trend. In particular, the data from Moraga (2006) were regarded as anomalous and were not used in the correlation. Moreover, for the case of tannery waste, there was uncertainty in the estimation of the carbon content as the value for leather was adopted; however, tannery waste consists of organic substances that are removed from hides and skins, which are composed of tissue and fat mixed with the chemicals used in the tanning process (Hetland et al., 2003).

Consequently, only the values from studies on a pilot scale were considered in this chapter (Figure 4.5b). However, there was a discordant value within the pilot series, which corresponds to carpet waste. The operational conditions of this study (Vaidyanathan et al., 2007) were quite particular, as wastes were introduced in aluminum cans and fed into the furnace in batches of three every 2 min. For this reason, it was decided to not include the two data points from this reference in Figure 4.5b.

The data points in Figure 4.5b show a very homogeneous tendency that could be adjusted either to a linear, a quadratic or an exponential correlation. Among these options, the most significant model was calculated based on an exponential function that allows for a residual electricity generation (from H_2) when the carbon content in wastes is zero. Thus, equation [3.4] was used to calculate the electricity generated by the combined cycle as a function of the carbon content of the wastes fed to the plasma treatment process:

$$E_C = e^{0.032 \cdot C} \quad [4.4]$$

where E_C is the electricity generated expressed in GJ/t feed and C is the carbon content in the wastes expressed as a percentage.

Table 4.3. Electricity generated in the combined cycle in the plasma gasification process for different wastes.

Waste type	Syngas heating value (units)	Electricity generated (units)	Source	Scale[(a)]
Fly ash (1)	1,785 (kWh/t fly ash)	-	Carabin and Gagnon, 2007	P
Fly ash (2)	766 (kWh/t fly ash)	-	Carabin and Gagnon, 2007	P
RDF (1)	5.88 (MJ/m^3 with gas yield 2.46 m^3/kg waste)	-	Tendler et al., 2005	I
RDF (2)	-	900 (kWh/t waste)	MPM Tech., 2005	P
Carpet waste	23.5–33.6 kW (waste feed rate 23.1 kg/h)	-	Vaidyanathan et al., 2007	P
Wood (1)	6.16 (MJ/m^3 with gas yield 2.48 m^3/kg)	-	Tendler et al., 2005	P
Wood (1)	-	930 (kWh/t waste)	MPM Tech., 2005	I
USAF BEAR waste	27.5-41 kW (waste feed rate 10.7 kg/h)	-	Vaidyanathan et al., 2007	P
Tannery waste	-	415 (kW from 560 kg/h waste feed)	Hetland et al., 203	I
Rubber	9 (MJ/Nm^3 with gas yield 3 Nm^3/kg rubber)	-	Huang and Tang, 2007	P/L
TDF	5.89 (MJ/m^3 with gas yield 5.03 m^3/kg waste)	-	Tendler et al., 2005	P
RDF (1)	4,106 (kWh/t waste)	2,328 (kWh/t waste)	Moraga, 2006	I
RDF (1)		1,150 (kWh/t waste)	Gallego, 2008	I

[(a)] Scale of application of the study: L = Laboratory; P = Pilot; I = Industrial.

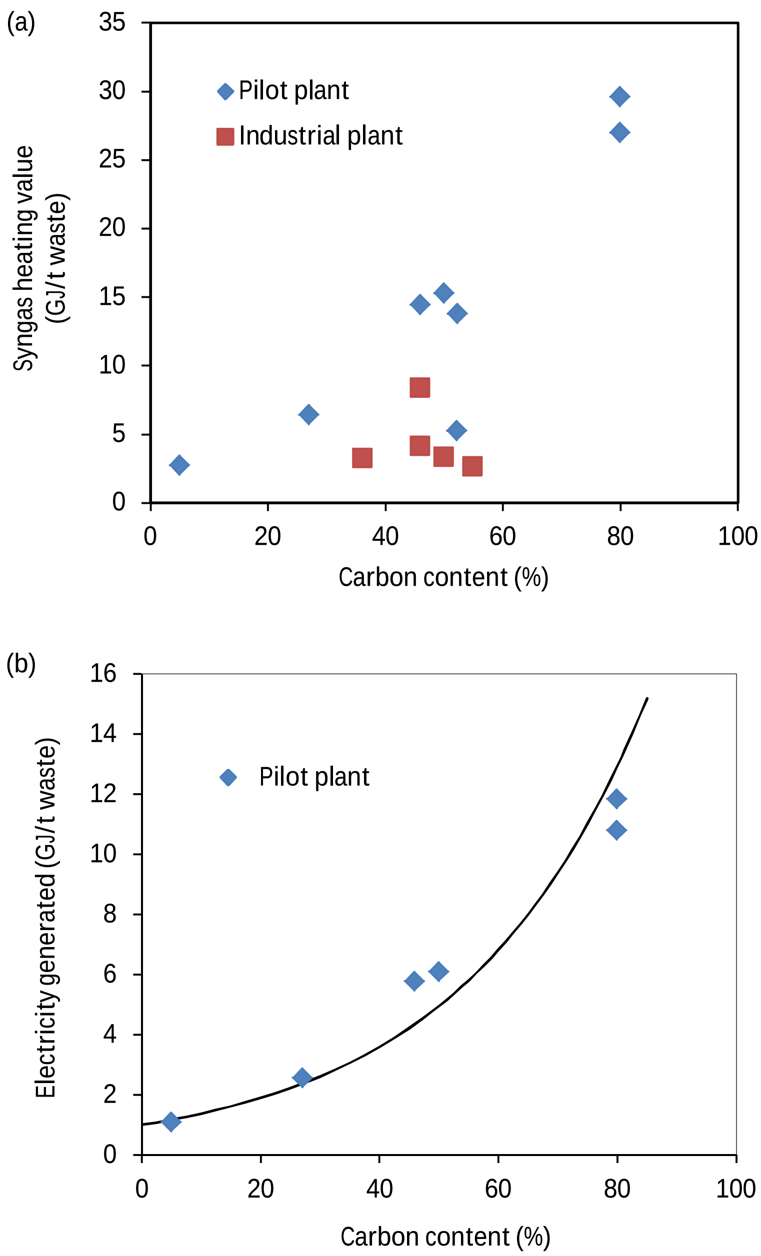

Figure 4.5. Electricity generation in the combined cycle as a function of carbon content in wastes. (a) All available data is expressed in terms of syngas heating values; (b) data from pilot plant studies expressed in terms of the electricity generated per ton of waste fed to the treatment system.

4.3.1.4. Carbon emissions in the combined cycle

The data used to estimate the carbon emissions released in the combined cycle is shown in Table 4.4. Given that 100% oxidation was not assumed, CO and CH_4 could have been present in the exhaust gas. Thus, these emissions were accounted for based on their carbon content and future transformation into CO_2 rather than considering their global warming potential factors. Nonetheless, their contribution in this particular case was practically negligible (< 0.3%).

The total emissions are shown in Figure 4.6, which allows for an examination of a relationship with the carbon content in wastes. As expected, an increasing linear relationship was observed for these parameters. The correlation, given below in equation [4.5], was calculated such that the origin condition was fulfilled. When this condition was not established, a constant was determined for the model, but it was statistically not significant (even for a confidence level of 90%). This is reasonable because if no carbon is present in the residues, then no CO_2 can be expected to form. Moreover, the data point from Wood (2) was discarded as it was an outlier (it corresponded to the only laboratory study considered for this calculation).

$$Carbon\ emissions = 0.035 \cdot C \qquad [4.5]$$

Here, the *Carbon emissions* are the CO_2 emissions released (expressed in tons per ton of waste treated) and *C* is the carbon content in the wastes expressed as a percentage.

Table 4.4. Data used for the estimation of CO_2 emissions released in the combined cycle.

Waste type	Waste input flow	Syngas flow	Syngas composition [a]			Source	t CO_2 / t waste
			CO	CO_2	CH_4		
Fly ash (1)	1,000 kg	1,843 kg	-	-	-	Carabin and Gagnon, 2007	0.99
Fly ash (2)	1,000 kg	341 kg	-	-	-	Carabin and Gagnon, 2007	0.18
Fly ash (3)	726 kg	38.7 kg C	-	-	-	Haugston and Gustavson, 2000	0.19
Fly ash (4)	810 kg	52.7 kg C	-	-	-	Haugston and Gustavson, 2000	0.24
EAF dusts (1)	1,075 kg/h	508 kg/h	33.1[b]	5.1[b]	-	Shoukens et al., 1993	0.27
EAF dusts (2)	1,170 kg/h	441 kg/h	88.2[b]	-	-	Shoukens et al., 1993	0.52
RDF (1)	1 kg	2.46 m^3	27.5[c]	3.2[c]	-	Tendler et al., 2005	1.48
RDF (2)	10,000 lb/h	11,067 lb/h	49.0[b]	30.6[b]	4.4[b]	Baggio et al., 2008	1.33
Wood (1)	1 kg	2.48 m^3	31.4[c]	3.5[c]	-	Tendler et al., 2005	1.70
Wood (2)	2.22 g/s	-	53.33[d]	2.95[d]	2.68[d]	Zhao et al., 2001	0.95
TDF	1 kg	5.03 m^3	24[c]	5.2[c]	-	Tendler et al., 2005	2.88

[a]Only those compounds implied in the estimation of CO_2 emissions are indicated.

[b]Percentage on weight basis. [c]Percentage on volume basis.

[d]Percentage on weight basis and referred to the waste input flow.

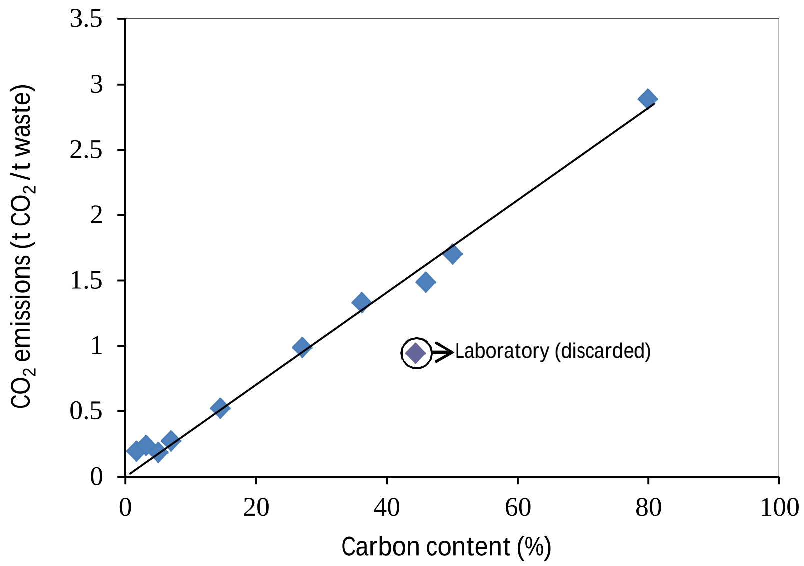

Figure 4.6. CO_2 released in the combined cycle as a function of the carbon content in wastes.

4.3.1.5. Slag production

To correlate slag production with the carbon content of the wastes, the used data corresponding to different processes reported in the literature are shown in Table 4.5. Unlike previously reported correlations, in this case, the quantity of slag obtained decreased as the carbon content increased (Figure 4.7). This was because the slag was mainly composed of the vitrified inorganic compounds present in the wastes treated in the plasma process. Furthermore, the trend was asymptotic with respect to the x axis. Therefore, the data were fitted to an exponential correlation by discarding the values from the study by Vaidyanathan et al. (2007), which are represented in dots in Figure 4.7 and did not follow the general tendencies of the other data. These two values were also excluded when calculating the correlation to estimate the electricity generated in the combined cycle (subheading 4.3.1.3), given the particular conditions of the mentioned study.

Table 4.5. Slag production in the plasma gasification process for different wastes.

Waste type	**Slag production**	**Source**
Fly ash (1)	730 kg slag/1,000kg ash	Carabin and Gagnon, 2007
Fly ash (2)	950 kg slag/1,000kg ash	Carabin and Gagnon, 2007
Fly ash (3)	439 kg slag/726 kg ash	Haugsten and Gustavson, 2000
Fly ash (4)	490 kg slag/810 kg ash	Haugsten and Gustavson, 2000
RDF (2)	600 (kg/h)/10,000 (kg/h)	Haugsten and Gustavson, 2000
Carpet waste	30.8–42.7 mass % of total input	Vaidyanathan et al., 2007
Medical waste	0.11 kg/kg waste	Rutberg et al., 2002
USAF BEAR waste	9.95–22.9 mass % of total input	Vaidyanathan et al., 2007
Polypropylene	2 % weight	Huang and Tang, 2007
RDF (1)	6,944.7 t/75,707.3 t feed	Moraga, 2006
RDF (1)	150 kg/t waste	Gallego, 2008

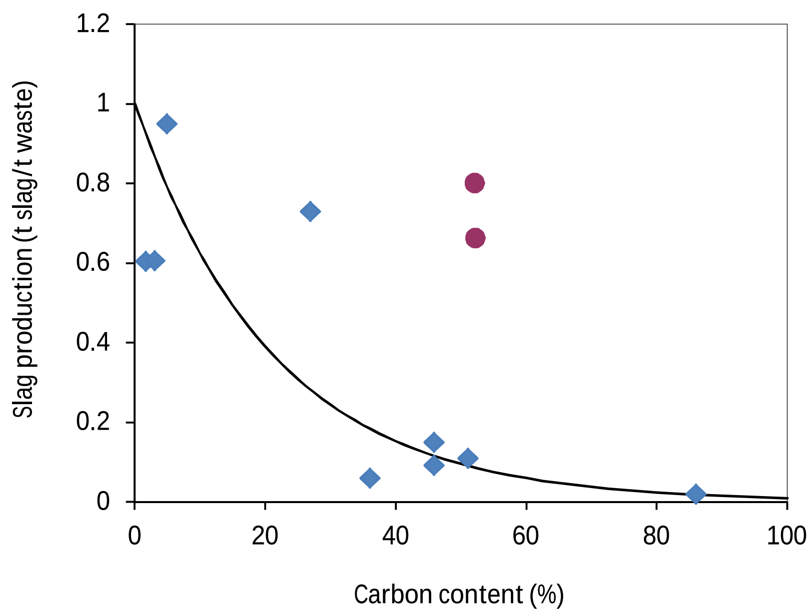

Figure 4.7. Slag production in the plasma reactor as a function of the carbon content of wastes.

To estimate the slag production as a function of the carbon content in wastes, the following equation was used:

$$Slag = e^{-0.047 \cdot C} \quad [4.6]$$

where *Slag* is the fraction of waste treated that is converted in vitrified material in tons of slag per ton of waste treated and *C* is the carbon content in wastes expressed as a percentage.

4.3.1.6. Statistical analysis of the correlations

The reliability and the significance of the correlations obtained were analyzed according to the statistical information provided by the SPSS 17.0 software. The information was summarized in Table 4.6, including the standard error and p-value for coefficients, standard error for the estimation and the R^2 value for the correlation.

Table 4.6. Summary of statistics for the correlations.

No. Eq.	Estimated parameter	Function type	Coefficient B			Standard error for the estimation	R^2
			Coefficient value	Standard error	*p*		
[4.3]	E_p	$y = e^{B \cdot x}$	0.026	0.006	0.005	0.606	0.816
[4.4]	E_c	$y = e^{B \cdot x}$	0.032	0.001	<0.001	0.186	0.991
[4.5]	Carbon emissions	$y = B \cdot x$	0.035	0.001	<0.001	0.086	0.996
[4.6]	Slag	$y = e^{B \cdot x}$	-0.047	0.005	<0.001	0.578	0.930

It can be seen that the R^2 values indicate that good correlations were obtained in all cases. Moreover, the significance of the coefficients in the equations was ensured by the low *p*-values obtained. Actually, this was a criterion taken into account during the selection of the function type for the correlations. In all cases, better significance was obtained when no constant was included in the model. Further reasoning to justify the correlations selected to model the electricity consumption in the plasma torch, and the electricity generated in the combined cycle is explained below (this is not done for carbon emissions and slag as the tendency of data in these cases was clear enough to avoid any doubt).

In the case of electricity consumption in the plasma torch, the major variability in data points pointed towards checking different likely function types. A quadratic correlation was discarded as no significance was obtained. Meanwhile, when a linear model without a constant was selected, a significant model with a good R^2 (0.869) was obtained; however, this would mean that no electricity is consumed when the carbon percentage is zero (the slag formation also requires energy). As a consequence, the exponential model was selected as the statistic parameters were also good. This model allowed the consideration of residual electricity consumption for null carbon content, and this kind of function fitted better to data points in Figure 4.4. Nonetheless, this is statistically the weaker of the functions calculated, and the availability of more data points would be desirable. The variability observed could be explained by the fact that the influence of water content was not considered, which can affect electricity consumption in the plasma torch (Tendler et al., 2005). In most of the experimental studies used in Figure 4.4 to obtain the correlation (Schoukens et al., 1993; Tendler et al., 2005; Carabin and Gagnon, 2007), additional vapor (in different quantities in each case) was supplied to the reactor, apart from the water embodied in the residues. The dependence of power consumption and plasma enthalpy on water content like that indicated by Tendler et al. (2005) was explored, but without obtaining significant results. For that reason, the influence of this parameter was excluded from the final model.

Regarding the electricity generated by the combined cycle, the linear and quadratic functions were significant only when no constant was included in the model. This meant that no electricity would be generated if the carbon percentage was zero; however, a certain amount of electricity can be generated in the combined cycle from H_2 and, therefore, this model would not properly represent the reality in the plasma process. Therefore, an exponential function was preferred that fit well with the data points in Figure 4.5b. In addition, this model allowed considering a residual electricity generation when the carbon content in wastes was zero.

Further analysis with respect to the behavior of the correlations in the boundaries of the study is provided in subheading 4.3.4.

4.3.2. Final model for the EF of wastes.

Once the correlations were developed, they were inputted into a spreadsheet (MS Excel®) together with the factors required to convert into units of area for the three main terms, i.e., net electricity balance, carbon emissions and slag. Thus, to estimate the EF of a given waste, it is necessary to account for the carbon content and the amount of waste generated. The used conversion factors for the calculations were equivalence factors (Kitzes et al., 2007), energy conversion factors (Doménech, 2006; Coto-Millán et al., 2008) and slag embodied energy (Doménech, 2006).

The tool was explored to assess the influence of carbon content in the EF of wastes for a specified amount of waste (1 t). Figure 4.8 conveys the results, including the contribution of the three main terms of the methodology. As the carbon content increases, more syngas is generated in the plasma torch and therefore a higher yield in electricity can be obtained. Hence, a positive contribution to the total EF occurs while the carbon content is lower than 24%; then, the electricity curve crosses the x-axis and the excess of electricity generated becomes a counter-footprint for the process. However, this contribution is not high enough to mitigate the EF associated to the CO_2 emissions generated during the combustion of the syngas, which linearly increase as the carbon content rises. On the other hand, the slag contribution to diminish the total EF reaches its maximum value when the carbon content is zero (at this point the whole waste is supposed to be inorganic material and is completely converted into slag), then it decreases exponentially. As a result, the total EF continuously augments as the carbon content does, but following a quite softened curve.

According to the estimates presented on the basis of 1 t of wastes generated, the EF of wastes could range from 1,292 to 5,988 gm^2. This means that the incorporation of this figure into the total EF appraisal of a process could mean a noticeable contribution.

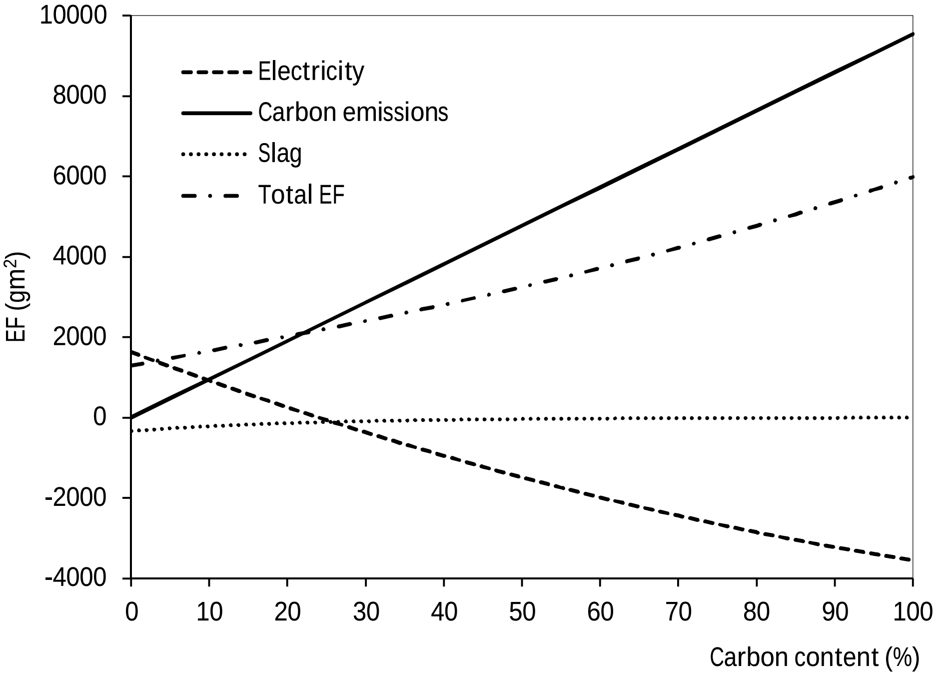

Figure 4.8. Influence of the carbon content of wastes in their EF using the thermal plasma based methodology.

4.3.3. Case study: textile process wastes

To test the proposed method with a real case, the wastes generated in the textile process analyzed in Chapter 3 were evaluated, for which the contribution of hazardous wastes to the total EF remained unevaluated due to the lack of an appropriate methodology. Non-hazardous wastes were also assessed using the approach proposed in this chapter and the results were compared to those obtained using conventional EF methodology. Thus, it was necessary to estimate the carbon content for the different kinds of wastes considered (Table 4.7).

The carbon content of batteries was associated with their casings, which, according to Ascent (2009), represents 6% of the total weight. The casings were assumed to be made of plastic and had a carbon content of 75% (MPM Tech., 2005). In the case of the oil filter, the carbon content of the waste was calculated considering the residual oil because the casing was usually made of metal.

According to the commercial enterprise SAIC Lubrication (2009), after filter compaction, 53% of the total weight recovered is oil. Considering that the carbon content in waste lubricant oil is 85.35% (Gómez-Rico et al., 2003), the carbon content of the oil filter was estimated to be 45.05%. In a similar way, if oil is considered to be absorbed by the mineral absorbent and assuming an absorption capacity of 0.7-0.9 kg liquid/kg absorbent (AB, 2009), then the percentage of carbon is 34.32%. Finally, for the paint, the percentage of carbon was calculated on the basis of the concentration of the VOCs (Hempel, 2009).

Table 4.7. Carbon content (*C*) of textile process wastes used for the EF estimate.

Waste type	**Comments**	***C* (%)**	**Source**
Non-hazardous wastes			
Textile		50	IPCC, 2006
Paper and cardboard		46	IPCC, 2006
Plastic		75	IPCC, 2006
Hazardous wastes			
Batteries	Associated to the casing (plastic)	4.5	IPCC, 2006; Ascent, 2009
Computers waste	Calculated as an average for motherboard, keyboard and casing.	47.64	Stewart and Lemieux, 2003
Fluorescent light		n.a.	
Oil filter		45.05	Gómez-Rico, 2003; SAIC, 2009
Used mineral absorbent	0.7 – 0.9 kg liquid/kg absorbent	34.32	Gómez-Rico, 2003; AB, 2009
Paint	455 g/l VOC´s and 0.9 kg/l specific weight	50.55	Hempel, 2009
Polluted containers	Considering plastic	75	IPCC, 2006

n.a. = not available

The proposed tool was applied to the flows of wastes generated during the year 2005 for the tailoring process described in Chapter 3, and the results are shown in Table 4.8 and indicate the contribution from each term in equation [4.2].

Table 4.8. Results of the application of the developed methodology to the textile process wastes and comparison with estimates in Chapter 3 for the year 2005.

Waste type	**EF previous estimate (gha)**		**EF new approach (gha)**			
	Recy-cling	No recycling	Elect.[(a)]	CO_2[(b)]	CF[(c)]	Total[(d)]
Non-hazardous wastes	*25.1*	*91.3*				*56.52*
Textile	22.4	83.5	-3.29	56.83	0.32	53.21
Paper and cardboard	2.5	7.3	-0.19	3.15	0.03	2.93
Plastic	0.2	0.5	-0.15	0.53	<0.01	0.38
Hazardous wastes						*$4.3 \cdot 10^{-2}$*
Batteries	-	-	$8.5 \cdot 10^{-5}$	$1.0 \cdot 10^{-4}$	$7.6 \cdot 10^{-5}$	$1.1 \cdot 10^{-4}$
Computers waste	-	-	$-2.9 \cdot 10^{-3}$	$4.2 \cdot 10^{-2}$	$3.9 \cdot 10^{-4}$	$3.9 \cdot 10^{-2}$
Fluorescent light	-	-	-	-	-	-
Oil filter	-	-	$-1.1 \cdot 10^{-4}$	$2.0 \cdot 10^{-3}$	$2.3 \cdot 10^{-5}$	$1.9 \cdot 10^{-3}$
Used mineral absorbent	-	-	0	0	0	0
Paint	-	-	$-5.0 \cdot 10^{-5}$	$5.7 \cdot 10^{-4}$	$4.3 \cdot 10^{-6}$	$5.2 \cdot 10^{-4}$
Polluted containers	-	-	$-6.4 \cdot 10^{-4}$	$2.3 \cdot 10^{-3}$	$3.7 \cdot 10^{-6}$	$1.6 \cdot 10^{-3}$

Terms in equation [4.2]: [(a)]EFelectricity; [(b)]EFcarbon_emissions; [(c)]CFslag; [(d)]EFwastes.

The results calculated here for non-hazardous wastes have the same order of magnitude as those previously reported. Hence, for the plasma-based methodology, a total contribution of 56.5 gha was estimated for these wastes, which was 91.3 gha (or 25.1 gha when 100% waste recycling was considered) using conventional EF methodology. Meanwhile, the EF calculated for the hazardous wastes was $4.3 \cdot 10^{-2}$ gha, which represented 0.08% of the total waste. This result means that negligible errors were assumed in Chapter 3 when considering the EF of this kind of waste, which was the situation here because very low quantities of hazardous wastes were generated (mainly during maintenance operations). Specifically, hazardous wastes represented only 0.25%.

However, other kinds of industries or activities that involve higher quantities of hazardous substances should produce wastes that are a major contribution to the total EF. Consequently, the availability of a methodology to assess their footprint and thus provide a more comprehensive and realistic measure of the total environmental impact of the process is essential.

Only two values were found in the literature that can be associated with the plasma treatment process for built-up land EF; these were 2 ha for a treatment capacity of 30,000 t/yr (Gallego, 2008) and 0.067 ha for a capacity of 1.1 million kg/yr (Sartwell, 2003). Both values yield a similar ratio for the area required per ton of waste treated ($6.7 \cdot 10^{-5}$ ha/t in the first case and $6.1 \cdot 10^{-5}$ ha/t in the second). Even after multiplying this result for the corresponding 2.21 equivalence factor for built land (Kitzes et al., 2007), the contribution of this land type compared with the total EF estimated for wastes could be considered negligible.

The influence of the consumption of electricity by auxiliary units was also determined to be negligible. Thus, the uncertainty due to the lack of data regarding this factor did not significantly influence the final results.

Furthermore, it was observed that, for the materials that were tested in the case study, the contribution of the counter-footprint calculated for the slag generated does not exceed a 1.4% contribution to the total EF, except for the particular case of the batteries (notice that the carbon content estimation in this case was not very accurate, as explained above). This means that the error assumed when considering that all the inert material generated in the plasma process could find a market application does not significantly alter the final results.

4.3.4. Limitations of the developed methodology

Even though plasma is a state of matter that, under appropriate conditions, can be induced for any type of waste, thermal plasma technology has only been developed for the treatment of hazardous wastes. As a result, the data available in the literature allows for evaluations of only certain kinds of wastes (Table 4.1). Thus, the usefulness of the developed methodology is more relevant for industrial activities where these wastes are generated, rather than for municipal policy makers.

The correlations obtained within this range were good, which shows that the carbon content acted as a characteristic parameter. However, inconsistent data points were observed at the boundaries of the study, especially for lower values of carbon percentage (i.e., lower organic matter content), as seen in subheading 4.3.1.6. Actually, the plasma process is particularly recommended for residues containing organic matter, thus allowing for energy recovery (Tendler et al., 2005). This means that not even a zero carbon-content waste could ever be treated by this technology. If this was the case, the evaluation of equation [4.6] would lead to calculate a generation of 1 t slag/t waste treated, i.e., because no organic matter is present the entire waste is converted into a slag. Therefore, from a cautious approach, the proposed methodology should presently be restricted to wastes within the studied range of carbon content. Moreover, the assumption of expressing the correlations as a function only of the carbon percentage could be considered very simple, given that different wastes with the same carbon content (Zolezzi et al., 2004) would lead to the same EF, regardless of their hazardousness. Thus, as for CH_4 emissions, we should ask whether EF accounts weigh the severity of impacts apart from assessing land requirements.

Another aspect that implies a source of error in the methodology is the likely over-estimation of the counter-footprint assigned to the slag. The methodology was constructed on the basis that all of the inert material produced could find an application in market, but whether this is feasible or not is difficult to know. A possible solution to mitigate the effect could be to consider a percentage of slag that could be reused as building material or for road construction, then accounting for it as a counter-footprint (energy savings as new materials production is avoided), while the remaining percentage should be assigned a footprint for its storage. Nonetheless, the uncertainty associated with the selection of this percentage would also be a source of error in the calculations.

As a final remark, the correlations for the estimation of electricity consumption by the plasma torch and electricity generated by the combined cycle were obtained on the basis of pilot plant data. This means that real processes at large scale may differ from the behavior predicted by the model. As more solid and reliable industrial plant data are available, a revision of these correlations should be considered.

4. Conclusions

A minimum criteria indicator, like the Ecological Footprint, may be sufficient for countries interested in knowing the pressure they exert on the environment that only consider whether or not they are exploiting more resources than are available. However, the situation is different at the corporate level, as more comprehensive analyses of all environmental burdens are required. To fulfill this aim, a new technique based on the application of different methodologies, where each methodology deals with different aspects, could be proposed. This would lead to an in-depth, but also laborious analysis. Indeed, handling more detailed information also means that it is more difficult to communicate the results; therefore, it would be useless for enterprises having corporate social responsibilities that need to report their behavior in a synthesized and easily understandable way. Thus, research must be carried out to improve integrated indicators, like the EF, that mostly fulfill the desired characteristics, i.e., an indicator that summarizes in one number a series of environmental impacts while possessing scientific rigor. This figure, expressed as a requirement of bioproductive area, can be interpreted by any stakeholder (policy makers, industry, scientific community or the general public) and compared to the available biocapacity to extract conclusions. Production activities would benefit from the availability of such an indicator to conduct more comprehensive analyses and express their environmental performance in corporate social responsibility reports.

In this chapter, a methodology for assessing the footprint of hazardous wastes (which is also suitable for non-hazardous wastes) was developed. The results were on the same order of magnitude of those previously reported using the standard EF methodology. Despite certain limitations, the usefulness of the proposed methodology relies on the availability of a method that accounts for the relative weight of hazardous wastes in the environmental evaluation of an activity, thereby allowing for a synthesized expression in terms of the ecological footprint in units of area. In addition, its application is quite simple and only requires knowing or being able to estimate the carbon content of the wastes considered. That being said, the consistency of the model will be improved as more data from studies on plasma technology become available in the future.

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Chapter 5

Methodological insights in EF accounting

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Abstract

The usefulness and drawbacks of the Ecological Footprint (EF) as sustainability indicator for products and production processes have been discussed in Chapters 3 and 4. In Chapter 4, a methodology to appraise the EF of wastes, including the hazardous ones, was presented. This methodology was focused on solid waste flows; however, there are other waste flows that can be stem from a production process, e.g., emissions to air and water. Chapter 5 deals with the former, since the EF is often criticized for not including emissions other than CO_2. In this respect, other greenhouse gases (GHG) were considered and the effect of incorporating acidifying emissions by considering a critical load was assessed. A ceramic industry (manufacture of bricks in baked clay) typical from Galicia was selected as study case to analyze the impact of emissions derived from the burning of fuel oil or natural gas during the drying and firing stages, as well as their influence in the EF figure.

Another controversial aspect is the use of global or local factors in EF assessments, leading to a discussion on the priority of assuring comparability among studies from all over the world or accuracy on estimates by adjusting to regional conditions. In this chapter, a specific CO_2 absorption rate was appraised for Galicia (NW Spain) using two different methodologies: a) Biomass Expansion Factors (BEF); b) Allometric equations. This rate was applied to the same case study to compare the results.

5.1. Introduction

The Ecological Footprint (EF) is an indicator mainly founded on the carrying capacity concept, which refers to the number of individuals who can be supported in a given area within natural resource limits, and without degrading the natural social, cultural and economic environment for present and future generations (Kratena, 2008; CCN, 2010). Thus, the EF determines the space required to support an activity by means of the area needed to provide the resources consumed and to absorb the wastes generated (Rees and Wackernagel, 1996; Monfreda et al., 2004). The advantages and usefulness of such indicator, as well as its main drawbacks and requirements for further research to overcome core critiques, have been discussed in Chapters 3 and 4.

One of the advantages that make the application of the EF to evaluate the sustainability of products and production processes very appealing is the fact that a great amount of information is synthesized and expressed in a way easy to communicate to the different stakeholders. As a consequence, applications in the field have been continuously increasing during the last years (Kratena, 2008; Limnios et al., 2009): e.g., aquaculture processes (Kautsky et al., 1997; Muir, 2005), water supplier company (Lenzen et al., 2003), mobile phones (Frey et al., 2006) or wine production (Nicolucci et al., 2008).

Generally, only CO_2 emissions are computed in EF assessments, founded on the availability of a sufficiently accurate method for calculating the land area required to absorb them, which is not the case for other greenhouse gases (Best et al., 2008). Other pollutants, such as acidifying emissions, are also excluded from EF estimates in spite of the existence of a specific assimilation rate for them (Holmberg et al., 1999). The reason is that they are considered to have an insignificant assimilation capacity in the biosphere and thus to result in an overestimation of the EF (Kitzes et al., 2007). Under this approach, pollutants or toxic flows that cannot be absorbed or broken down by biological processes are systematically excluded from EF estimates. This may pose the exclusion of many important input and output flows of a production process, such as the hazardous wastes analyzed in Chapter 4, thus leading to an underestimation of its real environmental impact. When applied to assess the environmental performance of a production process, a more comprehensive analysis of potential burdens is required; otherwise, the results reported could be misleading and useless when comparing two production processes from a sustainability point of view.

Another important issue in EF accounting is the quality and reliability of the factors employed. EF studies generally apply global factors. This means that global average values of productivity and biocapacity are considered and, therefore, studies carried out in different regions and countries can be compared since they are expressed in the same units (global hectares –gha-). Besides, this avoids the allocation of burdens to a certain area. However, some aspects handled in EF assessments are highly site specific, and using global factors can significantly alter the final results. Kitzes and Wackernagel (2009), for instance, suggest an analysis at regional or local scale for freshwater. Similarly, the capacity of a certain area to assimilate wastes would depend on soil characteristics, vegetation distribution, basal concentrations of contaminants or previous pollution episodes. This is of particular importance for environmental impacts with a local effect. In this regard, considering the local context could result a key issue for decision making at corporate level, although comparability among different production sites might be reduced.

This chapter aims to discuss and revise two methodological aspects that could improve EF assessments at corporate level: a) estimate of a carbon sequestration rate for Galicia (NW Spain) based on the specific species found in the forests of this region; b) incorporation of emissions other than CO_2. To test the influence of incorporating these issues in EF estimates, a typical industrial plant for the production of ceramic product placed in Galicia (NW Spain) was considered as case study.

5.2. Methodology

Next, the methodologies used to estimate the specific carbon absorption rate for Galicia, as well as the protocol proposed to integrate emissions other than CO_2 in EF appraisals, are presented. To observe the potential effects on EF estimates of the methodological aspects revised in this chapter, a typical industrial plant for the production of ceramic products was employed as example.

5.2.1. Case study

The proposed case study corresponds to a typical industrial plant for the production of ceramic products in Galicia (NW Spain). Generally the term ceramics (ceramic products) is used for defining inorganic materials, with possibly some organic content, made up of non-metallic compounds and hardened by a firing process. Specifically, the installation is dedicated to the manufacture of bricks in baked clay. The process for the production of bricks can be divided into four main stages including raw materials preparation, shaping, thermal treatment and post-processing. Bricks are made by stiff mud extrusion processes. After shaping, the ceramics are thermally treated: a) Drying stage; b) Firing stage. In this case study it is considered that bricks are made in a continuous process performed in a dryer and tunnel kiln. The system is heated mainly by natural gas or fuel oil burners, which make the temperature up to 120ºC over approximately a 24 h period for drying. Afterwards, bricks are heated up to a maturing temperature between 900 and 1200ºC. In this process, pollutants released are particulate matter, sulfur oxides (SO_x), nitrogen oxides (NO_x), volatile organic compounds, methane (CH_4), carbon monoxide (CO), carbon dioxide (CO_2), nitrous oxide (N_2O), fluoride, and heavy metals. According to the United States Environmental Protection Agency (US EPA, 1997) and the European Environmental Agency (EEA, 1996) the main relevant pollutants are emitted in the thermal process, i.e. SO_x, NO_x, CO, CH_4 and CO_2. The recycling of exhaust gases from the kiln to the pre-heat zone and to the dryer allows the improvement of heat recovery and decrease the concentration of pollutants (Barros et al., 2007). The use of natural gas for thermal treatment is growing, while the use of fuel oil is decreasing. Energy consumption is typically around 1,600-3,000 kJ/kg (JRC, 2007). A complete description of the process can be seen in Barros et al. (2007).

In this paper, the case study is focused on the evaluation of the EF based on the emissions produced in the thermal treatment stage, considering the use of two different types of fuel (natural gas and fuel oil), and also those stem from the raw materials. Operational data of the considered thermal process are given in Table 5.1.

Table 5.1. Operational data for the brick production process.

Concept	Value	Units
Bricks annual production	38,000	t/yr
Tunnel kiln capacity	4	m^3
Fuel energy consumption in thermal process units:		
Tunnel kiln	68,000	GJ
Dryer	9,000	GJ
C in clay	0.8	%
Clay/brick ratio	1.35	kg/kg

The estimation of pollutant emissions during drying and firing stages was carried out considering the emissions of the combustion processes using the following equations:

$$Emission\left(\frac{kg}{yr}\right) = Efac\left(\frac{kg}{t\ product}\right) \cdot P\left(\frac{t\ product}{yr}\right) \quad [5.1]$$

$$Emission\left(\frac{kg}{yr}\right) = Efac\left(\frac{kg}{GJ}\right) \cdot B\left(\frac{GJ}{yr}\right) \quad [5.2]$$

Where $Efac$, P and B are the emission factor, bricks annual production and annual energy consumption, respectively.

The CO_2 emissions derived from the raw material were calculated according to equation [5.3], considering the carbon content *($\%C_{clay}$)* in the raw material (clay), which is completely oxidized during the process. A ratio 1.35 kg clay/kg bricks was used.

$$Emission\left(\frac{kg\ CO_2}{yr}\right) = 1.35\left(\frac{kg_{clay}}{kg_{brick}}\right) \cdot \left(\frac{44}{12}\right) \cdot \left(\frac{\%C_{clay}}{100}\right) \cdot P\left(\frac{kg_{brick}}{yr}\right) \quad [5.3]$$

Emission factors, recorded from US EPA (1997), EEA (1996) and Government of Andalusia (2009), used in equations [5.1] and [5.2] for drying and firing processes in a tunnel kiln using natural gas and fuel oil, are given in Table 5.2.

Table 5.2. Emission factors (*Efac*) for pollutants produced in the thermal units (dryer and tunnel kiln) of the bricks production process using either fuel oil or natural gas.

	CO_2	SO_x	NO_x	CO	CH_4
Efac for a dryer using natural gas	56 (a),(c)	0.00411 (c)	0.100 (c)	0.017 (c)	0.01 (b),(d)
Efac for a dryer using fuel oil	76 (c)	0.498 (c)	0.159 (c)	0.015 (c)	-
Efac for a tunnel kiln using natural gas	56 (a),(c)	0.39 (b),(d)	0.090 (d)	0.030 (d)	0.0185 (b),(d)
Efac for a tunnel kiln using fuel oil	76 (c)	2 (c)	0.55 (c)	0.060 (d)	-

All emission factors (*Efac*) were taken from EEA (1996) except for (a)Government of Andalusia (2009) and (b)US EPA (1997). (c)Expressed in kg/GJ, estimated using equation [5.2]; (d)Expressed in kg/t product, estimated using equation [5.1].

5.2.2. Estimate of a specific carbon sequestration rate for Galicia

The specific carbon sequestration rate for Galicia (NW Spain) was assessed on the basis of the capacity of the forests to act as a carbon pool. With this purpose, the main species present in the Galician forests were taken into account: *Pinus pinaster*, *Eucalyptus globulus*, *Quercus pyrenaica*, *Quercus robur*, *Pinus radiata*, *Castanea sativa* and *Pinus sylvestris* (Galician Government, 2001).

Data from the second and third Spanish forest inventories –IFN2 and IFN3- (Spanish Government, 1998; 2002) were extracted to appraise the increase in biomass during the period of time elapsed between them (data for Galicia were collected in 1986 and 1997, respectively). The two methods generally used to convert field measurements of trees to aboveground biomass were applied: a) Biomass Expansion Factors (*BEF*); b) Allometric equations (Brown, 2002; Teobaldelli et al., 2009). The *BEF* expresses a ratio between the total aboveground biomass and the merchantable or bark volume (V_{cc}), the latter being more frequently available. Thus, the total wood or biomass (W_t) for a given species can easily be obtained following equation [5.4] and using data in Table 5.3.

$$W_t = V_{cc} \cdot BEF \qquad [5.4]$$

Table 5.3. Data for the estimation of biomass in the Galician forests.

Species	**IFN2[(a)] V_{CC} (m^3)**	**IFN3[(a)] V_{CC} (m^3)**	**BEF[(b)] (t m^{-3})**
Pinus pinaster	45,445,918	49,151,041	0.62
Eucalyptus globulus	15,620,749	34,800,921	0.81
Quercus pyrenaica	1,210,592	3,573,121	1.11
Quercus robur	10,117,114	16,922,380	0.84
Pinus radiata	4,679,509	7,571,425	0.44
Castanea sativa	3,736,416	5,639,445	0.75
Pinus sylvestris	1,749,453	3,756,839	0.62

[(a)]Galician Government, 2001; [(b)]For Spain. Solla-Gullón et al., 2006.

On the other hand, allometric equations predict the biomass of tree components (e.g., stem bark, branches, leaves, etc.) on the basis of characteristic parameters of the trees, such as the diameter at breast height (*d*), although regressions fairly improve when the height (*h*) of the tree is also considered (Brown, 2002). The forest inventories classify trees into diameter classes (Table A5.3), thus facilitating the application of these equations. The total biomass for each species is obtained according to equation [5.5].

$$W_t = \sum_i \left(\sum_j W_{i,j} \right) \cdot N_i \qquad [5.5]$$

Where $W_{i,j}$ is the partial contribution of each tree component *j* in the diameter class *i* (estimated using the corresponding allometric equation) and N_i is the number of trees belonging to the diameter class *i*.

This is a more complex but also considered as a more accurate method. For many of the species considered in this chapter (e.g., *Pinus pinaster*, *Pinus radiate*, *Eucalyptus globulus* and *Quercus robur*), specific allometric equations have been developed using experimental data from Galician forests and were available in the literature (Merino et al., 2005; Balboa-Murias et al., 2006a/b; Solla-Gullón et al., 2006). To complete the study, allometric equations for *Quercus pyrenaica*, *Pinus sylvestris* and *Castanea sativa* were extracted from studies carried out in Spain but as close as possible to Galicia (Santos Regina, 2000; Montero et al., 2004). The equations used in this chapter were collected in Annex A5 together with the parameters required for their application.

To estimate the carbon stored in the biomass determined applying either of the above explained methods, data on carbon content in wood was compiled (Annex B5). For some species (Tables B5.1 to B5.3), the references employed provided values differentiated by tree component (C_j), while in other cases (Table B5.4) only an average value (C_{av}) for the whole tree was available. This meant that a slightly different protocol had to be followed depending on the species considered and on the method used to assess the biomass:

- BEF method. The stem wood is the major contributor to the total biomass of a tree. Thus, for the species in Tables B5.1 to B5.3, the carbon content for the stem wood was considered as representative of the whole tree (C_{av}). Using these values and those from Table B5.4, expressed in g g^{-1}, the total carbon stock (C_t) for a given species was assessed following equation [5.6]:

  $$C_t = W_t \cdot C_{av} \quad [5.6]$$

- Allometric equations. For the species in Table B5.4, first equation [5.5] was applied and then the total biomass was multiplied by the average carbon content as indicated in equation [5.6]. Meanwhile, for those species in Tables B5.1 to B5.3, the carbon stock was appraised separately for each tree component and then the results were aggregated, as indicated in equation [5.7].

  $$C_t = \sum_i \left(\sum_j W_{i,j} \cdot C_j \right) \cdot N_i \quad [5.7]$$

The results obtained for each individual species were summed up to obtain the total carbon stored in the two years of reference (C_{IFN2} and C_{IFN3}, expressed in t). Finally, the carbon sequestration rate (r_C, t C/(ha·yr)) for Galicia was assessed using equation [5.8]:

$$r_C = \frac{C_{IFN3} - C_{IFN2}}{S \cdot \Delta t} \quad [5.8]$$

$$r_{CO_2} = r_C \cdot 44/12 \quad [5.9]$$

The total wooded area in Galicia for the two years of reference was: 1,045,376 ha for IFN2 and 1,405,452 ha for IFN3 (Galician Government, 2001). An average value of 1,225,414 ha (*S*) was used to assess the final carbon absorption rate. The time elapsed between the two forest inventories was 11 years (Δt). The conversion into CO_2 is easily achieved by the mass relation 44/12 (equation [5.9]), so that r_{CO2} is expressed in t CO_2/(ha·yr).

5.2.3. Incorporation of emissions other than CO_2

In this section, the incorporation of global warming and acidifying emissions into EF estimates is discussed.

5.2.3.1. Global warming emissions

Apart from CO_2, the other gases released during the drying and firing stages of the ceramics production considered as greenhouse gases (GHG) were CO and CH_4. As discussed in Chapter 4, the former has a much lower equivalent warming effect than CH_4, according to IPCC reports (Forster et al., 2007; IPCC, 2007). The higher or lower contribution to global warming is measured in terms of the characterization factor used to translate the emissions of a certain substance into its CO_2 equivalent (as it is done in Carbon Footprint assessments). These CO_2 equivalent emissions could then be converted into area requirements by means of assimilation rates; however, it would mean weighting the severity of the impact which may not be directly applicable to bio-productivity (e.g., a larger area would be assigned for the sequestration of CH_4 than for CO_2). As a consequence, the incorporation of GHG other than CO_2 into EF assessments is controversial and highlights the difficulty of a land-based indicator to measure all kinds of anthropogenic impacts (Walsh et al., 2009). Moreover, this does not necessary imply that CH_4 requires a much larger area to be sequestered. To this respect, controversy exists around including weighting factors (i.e., the characterization factors) which may not be directly applicable to bio-productivity (Walsh et al., 2009). An intense debate is taking place on this matter (Kitzes et al., 2009), but so far a widely accepted position does not exist. Therefore, a precautionary approach was adopted in this chapter (as in Chapter 4), and the EF of CO and CH_4 was assessed by means of their natural oxidization to CO_2 in the atmosphere.

The area (A_{GHG}) required for the absorption of annual estimated CO_2 emissions is then assessed using equation [5.10].

$$A_{GHG} = \frac{C\ emissions}{r_C} = \frac{CO_2\ emissions}{r_{CO_2}} \qquad [5.10]$$

The global carbon sequestration rate of 1 t C/(ha·yr) used in Living Planet Reports was applied, as well as the specific one obtained for Galicia to make comparisons.

5.2.3.2. Acidifying emissions

Acidifying emissions relevant in the case study were sulfur and nitrogen oxides, which were translated into area units (A_{acid}) by means of a critical load (equation [5.11]).

$$A_{acid} = \frac{H^+\ equivalents}{critical\ load} \qquad [5.11]$$

A critical load for acid deposition is the highest deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on ecosystem structure and function (Nilsson and Grennfelt, 1988). A general threshold for Europe, $20 \cdot 10^{-3}$ eq $H^+/(m^2 \cdot yr)$ (Holmberg et al., 1999), was considered. Although the critical load is a human defined concept, in contrast to the CO_2 absorption factor, it responds to the EF philosophy of closed cycles in nature. Other authors have accomplished the introduction of other wastes flows in EF accounts, as nitrogen and phosphorus, calculating the areas needed for total denitrification (lakes, wetlands) and total phosphorus retention (lakes, agricultural land), considering areas of different local yields or world-average yields as common units (Folke et al., 1997; Wackernagel et al., 1999).

5.3. Results and discussion

5.3.1. Inventory for the ceramic process

As a result of the application of emission factors and equations in section 5.2.1, the inventory for the case study process was obtained (Table 5.4). Emissions released in the dryer and tunnel kiln, as well as those stem from the raw material, are presented for the cases when natural gas and fuel oil are employed. CO_2 emissions are largely the most important ones in both cases, followed by SO_x and NO_x. As expected, the use of fuel oil implies the release of higher amounts of emissions than when natural gas is employed. Thus, acidifying emissions (SO_x and

NO_x) become particularly relevant in the former case, posing an additional damage to the environment.

Table 5.4. Inventory of emissions for the case study.

Pollutant	Emissions (kg/yr)					
	Natural gas			Fuel oil		
	Tunnel kiln	Dryer	Raw material	Tunnel kiln	Dryer	Raw material
CO_2	3,808,000	504,000	1,505	5,168,000	684,000	1,505
SO_x	14,820	37	-	76,000	4,482	-
NO_x	3,420	900	-	20,900	1,431	-
CO	1,140	153	-	2,280	135	-
CH_4	703	380	-	-	-	-

5.3.2. Specific carbon sequestration rate for Galicia

In this work, a regional specific carbon sequestration rate for Galicia was estimated and applied to assess the EF. First, BEF and allometric equations were applied to appraise biomass (equations [5.4] and [5.5]) based on data from the second and third national forest inventories; then, these values were transformed into carbon stocks (equations [5.6] and [5.7]).

Different allometric equations were available in the literature for some of the species (Table A5.1). For *Pinus pinaster*, for instance, the same authors proposed different equations. Both groups of equations indicated in Table A5.1 were tested. Equations [A5.7-A5.12] yielded values around 5% higher than equations [A5.1-A5.6]. The results obtained from the former were selected since equations were extracted from an only reference, thus ensuring a major consistency among them. Moreover, they approximated most to results from the BEF method, reducing the difference to levels around 3%. In the case of *Eucalyptus globulus*, equations [A5.21-A5.24] from Merino et al. (2005) introduced an additional parameter (basal area). As an approximation, an average for data in Table A5.2 was used. The results were tested with those from equation [A5.25] extracted from Jiménez et al. (2007), yielding quite insignificant differences (3.5% for IFN2 and 0.4% for IFN3). Results from equations [A5.21-A5.24] were finally employed.

Apart from equations for different tree components, regressions for total aboveground biomass were available for *Quercus pyrenaica* (equation [A5.34]), *Pinus sylvestris* (equation [A5.39]) and *Castanea sativa* (equation [A5.46]). The aboveground biomass estimated using these equations was compared to the value resulting from the aggregation of individual tree component regressions. In all cases, the second method yielded the higher biomass values and was the preferred one (more accuracy is expected from a differentiated analysis by tree components). The difference was more significant for *Quercus pyrenaica* and *Castanea sativa* (the higher the diameter class the higher the difference), being quite negligible for *Pinus sylvestris*.

It must also be remarked that for *Quercus robur*, *Quercus pyrenaica* and *Pinus sylvestris*, equations ([A5.33], [A5.38], [A5.45]) to estimate biomass in roots were also available, apart from aboveground tree components. The average contribution of roots represented 21%, 16% and 22%, respectively.

Figure 5.1 conveys the results for carbon stocks indicating the contribution by species of tree. BEF and allometric equations methods yielded an increase in carbon stored by all the species considered; however, when the latter were used, a major quantity of carbon stocks was determined, except for *Pinus pinaster* and *Quercus robur*. The most significant difference between both methods was observed for *Castanea Sativa* (304% in IFN2 and 229% in IFN3), for which specific allometric equations for Galicia were not available. In average, allometric equations provided a carbon stock 18% and 17% higher for IFN2 and IFN3, respectively. Total carbon stocks were presented in Figure 5.2.

As a result, the r_{CO2} obtained was different depending on the biomass estimation method employed: a) 3.83 t CO_2/(ha·yr) for BEF and b) 4.33 t CO_2/(ha·yr) for allometric equations. The former is in quite good agreement with the world-average one (3.67 t CO_2/(ha·yr)) used in Living Planet Reports, but the latter was 18% higher, which is quite noticeable. Allometric equations can be considered as a more accurate method, as the different parts of the tree are assessed independently for each species. Besides, for the majority of the species, site specific correlations for the area were available (Merino et al., 2005; Balboa-Murias et al., 2006a/b; Solla-Gullón et al., 2006). Thus, the absorption rate estimated on the basis of allometric equations could be considered as more reliable.

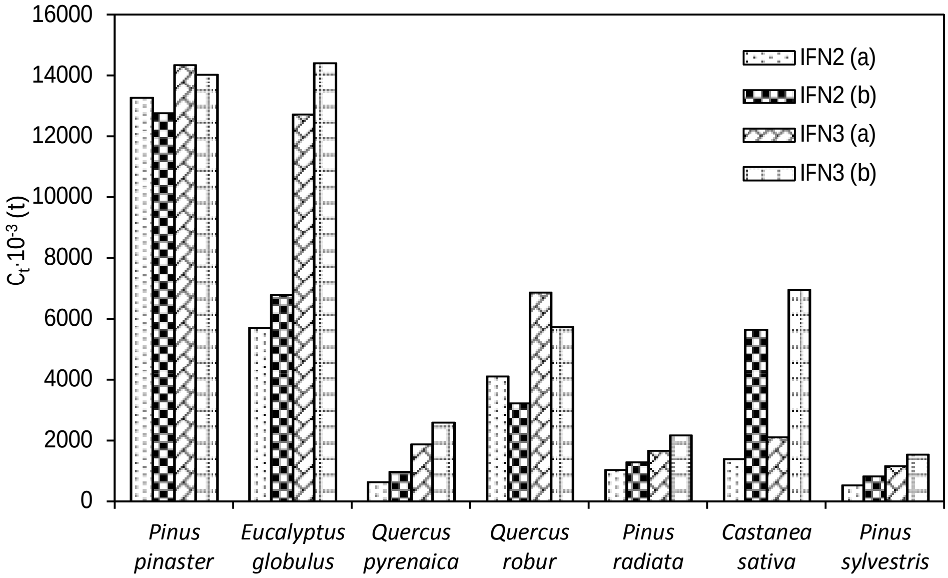

Figure 5.1. Total carbon stocks by species appraised using data from the second and third national forest inventories (IFN2 and IFN3) and applying (a) the BEF method and (b) allometric equations for biomass estimation.

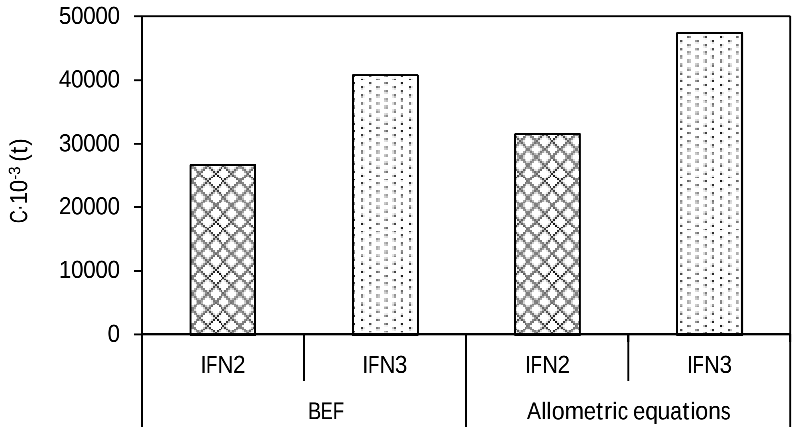

Figure 5.2. Total carbon stocks appraised for the Galician forests.

Considering the results from the allometric equations, Figure 5.3 shows the differences in carbon storage between IFN2 and IFN3 by species, thus representing their contribution to the total assimilation capacity for the Galician forests. As it can be observed, *Eucalyptus globulus* was clearly the major contributor (the increment in number of trees was very significant as it can be inferred from Table A5.3), followed by *Quercus robur* and *Quercus pyrenaica*.

Although carbon storage in tree biomass reaches high values, assessment of carbon budgets in forests should also take into account the litter layer and soil, as these are major storage compartments. The potential of soils as long-term carbon sinks is, however, much less well understood than that of tree carbon, even though mineral soil is the compartment of the system that stores most stable carbon (Liski et al., 2002; Balboa-Murias et al., 2006b). Further, the biomass in roots could not be assessed for most of the species studied due to the lack of available allometric equations. Therefore, the carbon stock capacity of Galician forests is still being underestimated, and could be much superior to the world-average; as a consequence, employing global instead of regional factors could lead to an overestimation of the EF of activities developed in Galicia.

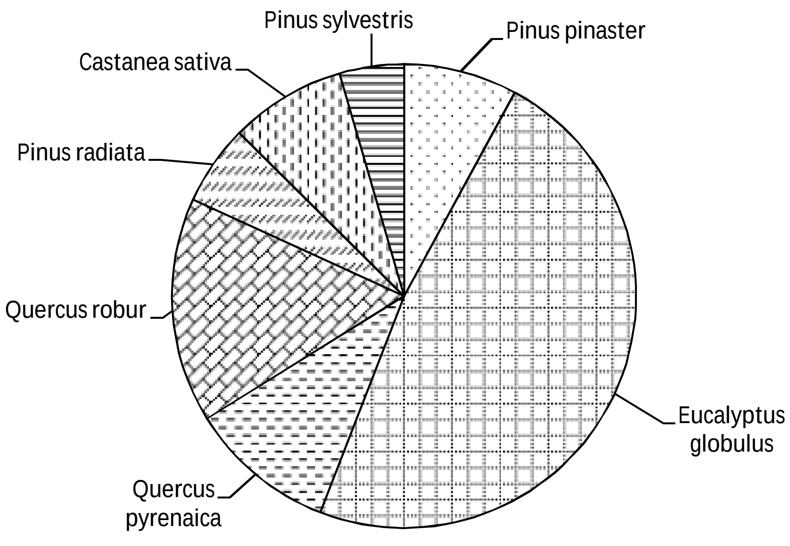

Figure 5.3. Differences in carbon storage between IFN2 and IFN3.

The effects of using the global or regional value (from allometric equations) in EF estimates for the case study were explored in the following section.

5.3.3. EF of emissions

The results for the EF estimates associated to the emissions in the ceramic process are shown in Figure 5.4. As it can be observed, the tunnel kiln is the process stage that poses a major impact to the environment in terms of emissions released. Besides, when fuel oil is used the EF is largely higher than for natural gas. In both cases, the major contribution comes from the acidifying emissions, mostly from SO_x. Nonetheless, the influence is much more significant when fuel oil is employed, with a 90% contribution of total acidifying emissions to the total EF, in contrast to the 69% for the natural gas case (Figure 5.5). For GHG emissions, the contribution from CO and CH_4 seems to be negligible, since the amount of emissions released is much lower than for CO_2.

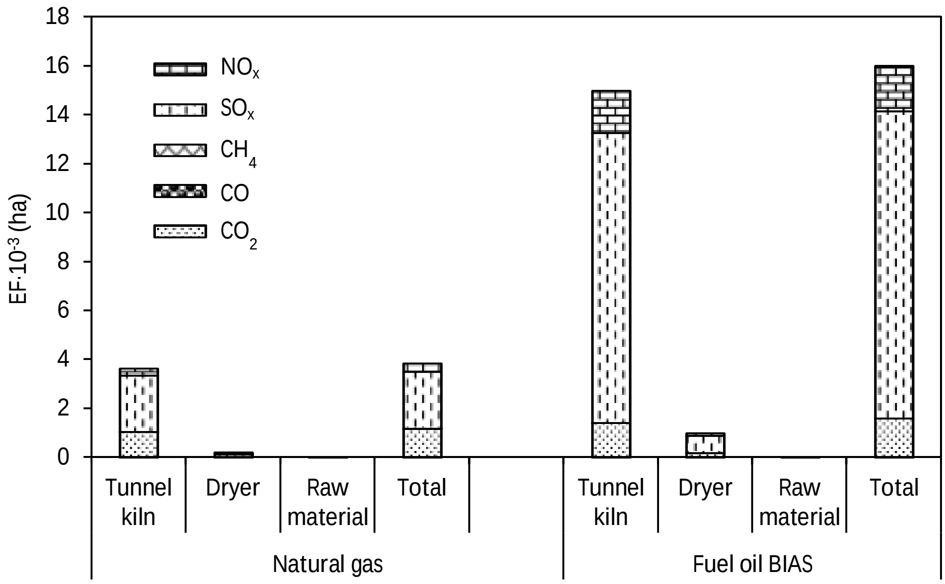

Figure 5.4. EF of emissions at the different process stages considered, indicating the pollutant and the source of energy employed.

As mentioned above, some of the main drawbacks of applying the EF at process level in the production of bricks arise from the quantification of the environmental impact due to acidifying emissions (SO_x and NO_x). These were the most relevant ones, although those stem from the raw material were not included as they usually are subjected to variability in raw material composition. A number of Best Available Techniques (BATs) were identified to reduce the concentration of these pollutants in bricks production process (JRC, 2007; Barros et al, 2007). These BATs consist of a combination of primary measures/techniques. The use of low sulfur raw materials (low sulfur clay) and additives (body additives as sand) or low sulfur fuels, such as natural gas or LPG (liquefied petroleum gas), leads to significantly reduce SO_x emissions. Meanwhile, minimization of nitrogen compounds in raw materials and additives can reduce NO_x emissions. Furthermore, these NO_x emissions can be minimized by the operation of special low-NO_x burners in the tunnel kiln. These burners are useful for reducing the flame temperature and thus reducing thermal and fuel derived NO_x. The application of these techniques implies the reduction of acidifying emissions, while CO_2 emissions keep more or less the same (except for the case of using natural gas). As a consequence, while only accounting for CO_2 emissions and discarding acidifying emissions, the EF would not be sensitive to the application of BATs, thus being useless as environmental indicator in decision-making at process level. Therefore, the integration with other environmental evaluation methodologies would be required to accomplish a more complete and reliable analysis.

The results are not expressed in global hectares as it is a common practice in EF appraisals (Monfreda et al., 2004). The purpose of the chapter was to assess the influence of using global or regional CO_2 sequestration rates. Therefore, the results were expressed in terms of the area needed to absorb these emissions. The subsequent conversion into gha was avoided since it would be the same in both cases and, consequently, it would not affect the comparison between the two cases analyzed. Besides, applying an equivalence factor for acidifying emissions would mean allocating the absorption of these emissions to a certain type of land, which cannot accurately be done. Further, a mutually exclusive approach (Venetoulis and Talberth, 2008) has been applied; this means that the area absorbing CO_2 emissions cannot be used to absorb SO_x emissions, for instance. If an additive approach was assumed, then the total EF could be reduced to the value determined for the major contributor (acidifying emissions in this case).

Figure 5.5 conveys the differences in EF appraisals when the carbon absorption rate estimated for Galicia (using allometric equations) is employed instead of the global one. The problem derived from CO_2 emissions has a global effect, although it is not possible to distinguish where emissions will be absorbed (in the surroundings or far away from where they are being released). The initial allocation of allowable emissions is established by the different national governments and it intends to take into account initiatives to capture CO_2 emissions, like the storage in biomass, thus giving relevance to regional capacities to assimilate emissions (European Commission, 2009). Therefore, deriving regional factors can usefully assist decision making in the local context. In this case, since a major carbon stock capacity was determined for the Galician forests, the EF due to GHG emissions decreased in the same order of magnitude (18%). However, the effect is not very significant in the total EF figure because of the major influence of acidifying emissions. Thus, a more noticeable variation would be observed if a specific critical load for Galicia was available and applied.

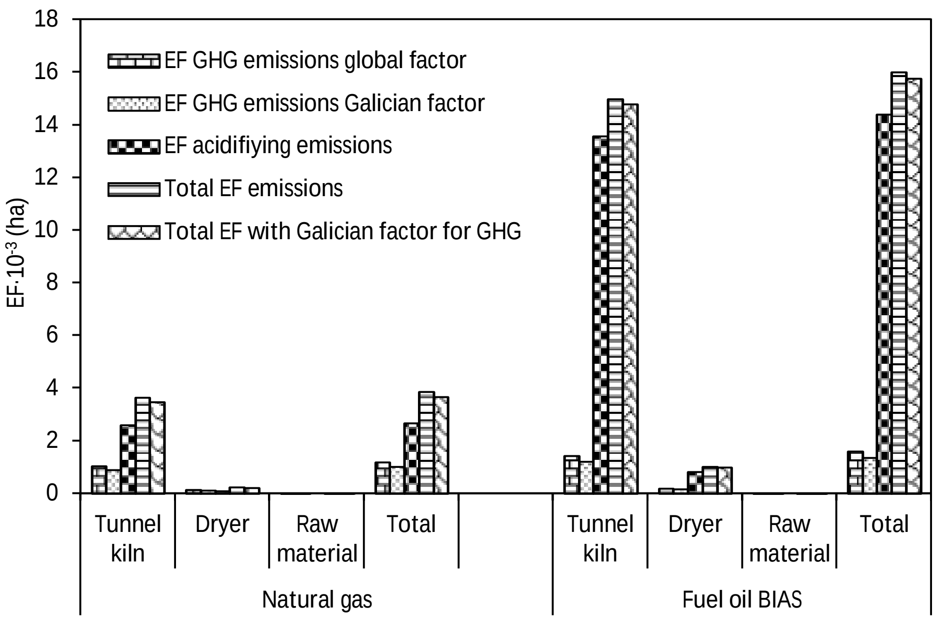

Figure 5.5. EF for the global process and comparison of results for GHG emissions applying global and site specific carbon absorption rate.

5.4. Conclusions

The availability of composite indicators like the EF is very interesting as it allows reporting the environmental performance of a company or industry in an easily understandable way. However, this simplicity poses a limitation since it implies the exclusion of some important aspects of the environmental behavior of the process. This is the case for residues, acidification and GHG emissions other than CO_2, analyzed in Chapters 4 and 5 of the thesis.

The discussion on whether using local emission sequestration potential instead of a global average one is methodologically feasible or not becomes relevant due to the important implications of the source/sink emission balances related to the emission trade off in a national context. Standardization is desirable, but the use of local factors should be discussed to approximate as much as possible to reality. From this work it could be concluded that, for activities developed in Galicia, EF assessments carried out using the global average 1 t C/(ha·yr) rate could be overestimated, since a 18% higher carbon stock capacity was appraised for the Galician forests (rate that could increase by incorporating other carbon sinks not evaluated in this chapter). Regarding the particular case of global warming gases, constraints on emissions are established at national and then at regional level, thus making it more appealing for enterprises the availability of site specific factors (European Commission, 2009).

The influence in EF of acidifying emissions became apparent in the application to the ceramic process. Although the methodological tendency in EF accounts is to exclude those substances with such a low assimilation in nature that lead to very high area requirements, it also becomes clear that by doing so the capacity of the indicator remains incomplete.

As a final conclusion, the refining and enhancement of the EF methodology (wastes, local factors, etc.) makes it more attractive for the evaluation of production processes, thus progressing towards the definition of the desired robust metric tracking for the whole environmental impact.

Annex A5. Allometric equations and data for biomass estimation

Notation:

- *W* is the dry weight of the different biomass components of the tree (kg)
- *d* is the breast height diameter (cm)
- *h* is the total height (m)
- *ba* is the basal area (m^2 ha^{-1})

Table A5.1. Allometric equations for biomass components.

Pools	Biomass equation	
Pinus pinaster (Balboa-Murias et al., 2006a; Solla-Gullón et al., 2006)		
Stem wood	$W = 0.3882 + 0.0115 \cdot d^2 \cdot h$	[A5.1]
Stem bark	$W = 2.54 + 0.002 \cdot d^2 \cdot h$	[A5.2]
Thick branches	$W = 3.2019 - 0.0148 \cdot d^2 - 0.4228 \cdot h + 0.0028 \cdot d^2 \cdot h$	[A5.3]
Thin branches	$W = 0.0978 \cdot d^{2.2881} \cdot h^{-0.9648}$	[A5.4]
Twigs	$W = 0.0019 \cdot d^{2.1537}$	[A5.5]
Needles	$W = 0.00082 \cdot d^{2.8845}$	[A5.6]
Pinus pinaster (Merino et al., 2005)		
Stem wood	$W = 0.0026 \cdot d^{1.734} \cdot h^{1.851}$	[A5.7]
Stem bark	$W = 0.0079 \cdot d^{2.098} \cdot h^{0.446}$	[A5.8]
Thick branches	$W = -9.707 + 0.0025 \cdot d^2 \cdot h$	[A5.9]
Thin branches	$W = 0.0036 \cdot d^{2.656}$	[A5.10]
Twigs	$W = 0.0022 \cdot d^{2.116}$	[A5.11]
Needles	$W = 0.0050 \cdot d^{2.383}$	[A5.12]
Pinus radiata (Balboa-Murias et al., 2006a)		
Stem wood	$W = 0.0123 \cdot d^{1.6042} \cdot h^{1.4131}$	[A5.13]
Stem bark	$W = 0.0036 \cdot d^{2.6564}$	[A5.14]
Thick branches	$W = 1.937699 + 0.001065 \cdot d^2 \cdot h$	[A5.15]
Thin branches	$W = 0.0363 \cdot d^{2.6091} \cdot h^{-0.9417}$	[A5.16]
Twigs	$W = 0.0078 \cdot d^{1.9606}$	[A5.17]
Needles	$W = 0.0423 \cdot d^{1.7141}$	[A5.18]

Table A5.1 (*cont.*). Allometric equations for biomass components.

Pools	Biomass equation	
Eucalyptus globulus (Merino et al., 2005)		
Stem wood	$W = 0.0062 \cdot d^{2.35} \cdot h^{1.001}$	[A5.19]
Stem bark	$W = 0.0093 \cdot d^{2.460}$	[A5.20]
Thick branches	$W = 0.0076 \cdot d^{3.39} \cdot ba^{-0.83}$	[A5.21]
Thin branches	$W = 0.0264 \cdot d^{2.63} \cdot ba^{-0.81}$	[A5.22]
Twigs	$W = 0.0451 \cdot d^{3.08} \cdot ba^{-1.59}$	[A5.23]
Leaves	$W = 0.0042 \cdot d^{3.69} \cdot ba^{-1.22}$	[A5.24]
Eucalyptus globulus (Jiménez et al., 2007)		
Total aboveground	$W = (15.262 \cdot d^{1.521} \cdot h^{1.664})/1000$	[A5.25]
(Includes stem wood, branches and leaves)		
Quercus robur (Balboa-Murias et al., 2006b)		
Stem wood	$W = -5.714 + 0.018 \cdot d^2 \cdot h$	[A5.26]
Stem bark	$W = -1.500 + 0.032 \cdot d^2 + 0.001 \cdot d^2 \cdot h$	[A5.27]
Thick branches (> 7 cm)	$W = 3.427 \cdot 10^{-9} \cdot (d^2 \cdot h)^{2.310}$	[A5.28]
Thick branches (2-7 cm)	$W = 4.268 + 0.003 \cdot d^2 \cdot h$	[A5.29]
Thin branches (2-0.5 cm)	$W = 0.039 \cdot d^{1.784}$	[A5.30]
Twigs (< 0.5 cm)	$W = 1.379 + 0.00024 \cdot d^2 \cdot h$	[A5.31]
Leaves	$W = 0.020 \cdot (d^2 \cdot h)^{0.737}$	[A5.32]
Roots	$W = 0.0851 \cdot d^{2.151}$	[A5.33]
Quercus pyrenaica (Montero et al., 2004)		
Total aboveground	$Ln(W) = -2.59695 + 2.53456 \cdot Ln(d)$	[A5.34]
Stem wood and thick branches (> 7 cm)	$Ln(W) = -4.2211 + 2.95974 \cdot Ln(d)$	[A5.35]
Thick branches (2-7 cm)	$Ln(W) = -1.97519 + 1.77301 \cdot Ln(d)$	[A5.36]
Thin branches (< 2 cm)	$Ln(W) = -4.85139 + 2.38766 \cdot Ln(d)$	[A5.37]
Roots	$Ln(W) = -2.4543 + 2.13346 \cdot Ln(d)$	[A5.38]

Table A5.1 (*cont.*). Allometric equations for biomass components.

Pools	Biomass equation	
Pinus sylvestris (Montero et al., 2004)		
Total aboveground	$Ln(W) = -2.50275 + 2.41194 \cdot Ln(d)$	[A5.39]
Stem wood	$Ln(W) = -3.80519 + 2.70808 \cdot Ln(d)$	[A5.40]
Thick branches (> 7 cm)	$Ln(W) = -15.0469 + 4.80367 \cdot Ln(d)$	[A5.41]
Thick branches (2-7 cm)	$Ln(W) = -4.07857 + 2.1408 \cdot Ln(d)$	[A5.42]
Thin branches (< 2 cm)	$Ln(W) = -2.08375 + 1.51001 \cdot Ln(d)$	[A5.43]
Needles	$Ln(W) = -2.36531 + 1.5099 \cdot Ln(d)$	[A5.44]
Roots	$Ln(W) = -4.56044 + 2.62841 \cdot Ln(d)$	[A5.45]
Castanea sativa (Santa Regina, 2000)		
Total aboveground	$W = 0.066 \cdot d^{2.647}$	[A5.46]
Stem wood	$W = 0.079 \cdot d^{2.541}$	[A5.47]
Branches	$W = 4.67 \cdot 10^{-4} \cdot d^{3.675}$	[A5.48]
Leaves	$W = 5.44 \cdot 10^{-5} \cdot d^{3.943}$	[A5.49]

Table A5.2. Basal area (m^2 ha^{-1}) of *Eucalyptus globulus* plantations for selected stands in Galician forests (Merino et al., 2005).

Species	Stand						Average
	1	2	3	4	5	6	
Eucalyptus globulus	34.0	29.5	39.8	49.6	30.7	22.3	34.3

Table A5.3. Diameter (*d*), breast height (*h*) and number of trees by diameter class in IFN2 and IFN3 (Galician Government, 2001).

Species	*d* (cm)	*h* (m)	No. trees (thousands)		*d* (cm)	*h* (m)	No. trees (thousands)	
			IFN2	IFN3			IFN2	IFN3
Pinus pinaster	10	7.54	55,157	64,687	45	21.78	1,831	3,261
	15	10.52	44,282	38,186	50	22.92	843	1,641
	20	13.12	38,100	27,942	55	23.84	335	796
	25	15.49	26,479	20,276	60	24.68	148	350
	30	17.38	17,557	15,790	65	24.94	56	123
	35	19.16	9,608	10,891	≥ 70	26.58	43	112
	40	20.51	4,835	6,510				
Pinus radiata	10	8.23	13,421	15,312	45	22.44	166	319
	15	12.25	7,759	9,730	50	23.45	75	164
	20	15.18	4,008	6,164	55	24.72	30	77
	25	17.53	2,022	3,919	60	26.00	13	39
	30	19.67	1,206	2,309	65	25.67	6	10
	35	21.04	693	1,423	≥ 70	24.79	6	12
	40	22.22	374	664				
Euclayptus globulus	10	12.81	32,409	92,686	45	31.69	574	847
	15	17.33	15,907	44,128	50	33.31	354	631
	20	21.08	9,100	22,310	55	34.03	223	400
	25	24.31	4,603	10,893	60	35.32	119	227
	30	26.90	2,758	5,716	65	35.12	58	138
	35	28.63	1,791	2,958	≥ 70	37.57	82	298
	40	29.76	1,064	1,607				

Table A5.3 (*cont.*). Diameter (d), breast height (h) and number of trees by diameter class in IFN2 and IFN3 (Galician Government, 2001).

Species	***d* (cm)**	***h* (m)**	**No. trees (thousands)**		***d* (cm)**	***h* (m)**	**No. trees (thousands)**	
			IFN2	**IFN3**			**IFN2**	**IFN3**
Quercus robur	10	7.51	14,086	23,410	45	16.41	390	727
	15	10.04	5,457	9,366	50	16.90	256	523
	20	12.07	3,224	5,178	55	17.28	145	332
	25	13.66	2,002	3,255	60	17.58	65	154
	30	14.70	1,584	2,392	65	16.72	33	95
	35	15.46	1,174	1,822	≥ 70	16.70	35	115
	40	15.90	791	1,315				
Quercus pyrenaica	10	7.41	8,740	22,294	45	15.81	84	182
	15	10.31	2,083	8,030	50	16.36	43	149
	20	12.19	877	2,973	55	16.85	17	66
	25	13.58	463	1,194	60	15.14	16	38
	30	15.15	385	835	65	16.34	5	12
	35	15.90	247	639	≥ 70	15.52	16	25
	40	16.24	154	361				
Castanea sativa	10	8.45	4,722	7,318	45	14.49	212	272
	15	11.21	2,698	3,468	50	14.49	154	209
	20	13.21	1,489	2,305	55	14.14	156	196
	25	14.39	833	1,356	60	13.45	113	124
	30	15.02	454	958	65	13.22	119	97
	35	14.55	385	593	≥ 70	14.32	863	1,001
	40	14.57	276	405				
Pinus sylvestris	10	6.84	14,056	10,215	30	14.85	178	1,052
	15	9.26	8,515	9,371	35	15.93	43	274
	20	11.53	3,246	6,676	40	16.61	13	51
	25	13.55	765	3,055	45	17.79	2	12

Annex B5. Carbon content in different species of trees.

Table B5.1. Mean (and standard deviation) carbon concentrations (g g^{-1}) in different components of *Pinus radiate* and *Pinus pinaster* (Balboa-Murias et al., 2006a).

Species	Stem wood	Stem bark	Thick branches	Thin branches	Twigs	Needles
Pinus radiata	0.504 (0.029)	0.541 (0.026)	0.513 (0.036)	0.525 (0.039)	0.532 (0.033)	0.527 (0.029)
Pinus pinaster	0.471 (0.027)	0.508 (0.029)	0.479 (0.024)	0.505 (0.030)	0.497 (0.029)	0.497 (0.023)

Table B5.2. Mean (and standard deviation) carbon concentrations (mg g^{-1}) in different components of *Quercus robur* (Balboa-Murias et al., 2006b).

Species	Stem wood	Stem bark	Branches			Twigs	Leaves	Roots
			> 7 cm	2-7 cm	< 2cm			
Quercus robur	484.4 (37.0)	512.0 (12.8)	490.9 (8.2)	484.0 (20.7)	502.7 (1.1)	506.8 (5.0)	503.8 (25.3)	486.4 (39.0)

Table B5.3. Mean (and standard deviation) carbon concentrations (g g^{-1}) in different components of *Eucalyptus globulus* (Solla-Gullón et al., 2006).

Species	Stem wood	Stem bark	Thick branches	Thin branches	Twigs	Needles
Eucalyptus globulus	0.452 (0.003)	0.425 (0.031)	0.455 (0.009)	0.451 (0.007)	0.464 (0.008)	0.520 (0.010)

Table B5.4. Average carbon concentrations (g g^{-1}) in *Quercus pyrenaica, Pinus sylvestris* and *Castanea sativa*.

Species	C (g g^{-1})	Reference
Quercus pyrenaica	0.475	Montero et al., 2004
Pinus sylvestris	0.500	Montero et al., 2004
Castanea sativa	0.500	Generic value for wood (Lamlom and Savidge, 2003)

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Chapter 6

Integration of EMFA, EF and LCA for environmental evaluation in the textile sector

Chapter 6

Integration of EMFA, EF and LCA for environmental evaluation in the textile sector

Abstract

There is a growing demand of instruments to measure the environmental impact of industrial processes caused by the generalized tendency of introducing environmental issues in management policies. This trend was mainly originated by the appearance of new regulatory laws and voluntary and administrative instruments, as well as the growing concern of the general public. In this respect, the application of the Ecological Footprint (EF) to assess the environmental performance of a production process was presented in Chapter 3. Although the usefulness of the indicator was proved, certain drawbacks were detected.

To extend and make the evaluation more comprehensive, the same study case was evaluated again in this chapter incorporating the new methodological proposals from Chapters 4 and 5. Further, apart from the EF, other environmental evaluation methodologies, namely Energy and Materials Flow Analysis (EMFA) and Life Cycle Assessment (LCA), were applied to provide a complementary perspective. The dressmaking process was modeled using Umberto® 5.5 and an analysis was conducted on the basis of material and energy flows. The updated results for the EF were presented and air emissions initially excluded from the EF assessment (Chapter 3) were evaluated under two approaches: 1) using characterized categories from LCA as complementary environmental indicators; 2) incorporating emissions into the EF by means of absorption factors including or not weighting factors, as discussed in Chapter 5.

Finally, sensitivity analyses based on Monte Carlo simulations were carried out to assess the influence of variability in the input variables.

6.1. Introduction

The entrance into force in June 2007 of the REACH (Registration, Evaluation, Authorization and Restriction of Chemicals) Regulation (European Commission, 2006) is only one of the latest examples of the regulatory pressure that industries are receiving to improve their environmental performance. The REACH Regulation or the Integrated Pollution Prevention and Control Law (Spanish Government, 2002; Barros et al., 2007) imply compliance with legal requirements. However, the improvement of the environmental performance is not only a matter of compliance but also an advantageous way of competing in market. Thus, companies have explored new management strategies based on voluntary administrative instruments (ISO 14000, EMAS, Eco-Label, Corporate Social Responsibility, etc.) which are also welcome by the general public (Hertwich et al., 1997).

Even though first attempts to prevent environmental pollution were focused on solving problems at different stages independently, there is a growing awareness that effective strategies must be based on integrated analyses and in life-cycle thinking. The Integrated Policy Product –IPP- (European Commission, 2003a) is a recent example of how policy makers are recognizing the requirement for broader based strategies (Curran, 2004; Gottberg et al., 2006). This new approach shows up the need to develop new tools to assess the improvements carried out by the enterprises in order to achieve the stated objectives.

Companies are incorporating environmental aspects as additional criteria in the design and operation of their industrial processes, not only due to the increasing legislation pressure, but also to the economic benefits linked to reduction of materials and energy (Santos and Gonçalves, 2009). Thus, interest has been focused on the evaluation of the environmental impact of productive processes employing different methodologies (Stefanis et al., 1995; Pun et al., 2003; Jiang et al., 2008).

Different tools and methodologies are currently available to deal with this task. In this sense, Life Cycle Assessment (LCA) is an internationally standardized methodology (ISO, 2006) that widely fulfills the desired characteristics (Finnveden and Moberg, 2005; Gustaffson and Börjesson, 2007). A number of life-cycle assessments were carried out on textile products in the EDIPTEX project (Laursen et al., 2007) or in the framework of the COSTAction 628. The latter was established to produce industrial environmental data of textiles in Europe and to suggest tools for comparisons of present technologies and practices with cleaner applications (Nieminen et al., 2007). In contrast, the Ecological Footprint (EF) is a more recent indicator and, consequently, its application in the textile sector is scarcer, although there are some examples like the estimation of the area required for the production of natural fibers (Hornborg, 2006). Energy and Material Flow Analysis (EMFA) is an assessment methodology especially valuable to identify, trace and allocate energy and material flows throughout a production system (Lou et al., 2004). Although not directly related to the environmental perspective, De Toni and Meneghetti (2000) modeled a knitwear network for production planning in the textile-apparel industry.

Several authors have addressed the comparison of EF and LCA as environmental evaluation tools, as well as their integration (Thomassen and Boer, 2005; Frey et al., 2006; Wood et al., 2006; Huijbregts et al., 2007; Pulselli et al., 2008), or have also proposed the integration of other different methodologies to create more comprehensive environmental evaluation frameworks (Robèrt et al., 2002; Pun et al., 2003; Pollard et al., 2004; Azapagic et al., 2007). Thomassen and Boer (2005) stated that providing a good insight into the environmental impact of a dairy production system required, besides input–output accounting indicators, LCA indicators. In their opinion, EF analysis was not effective for land and fossil energy use, because of its limited relevance and low quality, whereas LCA resource-based indicators were effective because of their high relevance, good quality and availability of data. Huijbregts et al. (2007) accomplished a comparison between the EF results and those obtained with the Ecoindicator 99 (EI), a commonly used life cycle impact assessment method. They studied 2,630 products and services and found that, for the majority of the products, there was an EF-EI ratio of around 30 m^2-eq·yr/ecopoint. Despite the interesting results, the EF methodology is not developed enough to be used as a stand-alone indicator and this ratio can vary substantially in processes in which high amounts of gases are released, metals employed or hazardous waste generated. Thus, a complementary use of EF and a gate-to-gate LCA is suggested in this chapter. Frey et al. (2006) successfully used a life-cycle approach to calculate the environmental burdens for producing and using a mobile phone, transforming them into the instantaneous rate of resource consumption and, therefore, including sustainability implications to the analysis developed by combined EF-LCA methodology. Pollutants flows from source (products, processes and human activities) to receptors in an urban environment were mapped by an integrated LCA methodology in the work by Azapagic et al. (2007), including substance flow analysis, fate and transport modeling and geographical information system.

Many industrial sectors are affected by environmental restrictions. In this case, a dressmaking factory, in which cotton jackets are manufactured, was studied. The textile sector is an important worldwide consumer of natural resources (raw materials and energy). In this respect, the International Cotton Advisory Committee has pointed out that the world cotton trade was expected to rebound by nine per cent to 8.9 million tons in 2007-08. Particularly, the organic cotton production climbed a 53% from 2005 to 2007 (Ecotextile, 2007). Besides, the increase in energy and raw material prices worldwide leads the textile industry to search new ways of optimizing its sustainable management (ICAC, 2007). Depending on the regional scenario, all stakeholders from the whole textile chain should take actions together during the whole life-cycle of the products to promote the market for greener products (Sondergard et al., 2004; Moore and Ausly, 2004; Manring and Moore, 2006). However, given that high legal restrictions are applied in Europe as boundary conditions to the supplier chain, the analysis of this chapter was focused on the tailoring stage within the textile chain. Furthermore, considering the increasing demand for environmental performance evaluation of industry and the subsequent need for sector-specific environmental performance indicators (Ren, 2000), it is of high interest the development and availability of tools to measure the biocapacity requirements and the environmental impact of the textile sector, as well as the degree of sustainability of the involved processes (Nieminen et al., 2007).

As explained in Chapter 3, the focus of the study was subjected to the interests of the company providing data. Hence, as part of their communication strategy, an EF analysis was conducted first due to its nature of aggregate indicator, easy to be interpreted by all the stakeholders. In this chapter, to extend and make the evaluation more comprehensive, the new methodological proposals from Chapters 4 and 5 were incorporated. Further, Energy and Materials Flow Analysis (EMFA) and Life Cycle Assessment (LCA) were applied to provide a complementary approach. The integration of these methodologies aimed to overcome the lacks of each of them and to take advantage of their complementary use.

6.2. Materials and methods

The following sections refer to the production process studied, the environmental evaluation methodologies applied and the strategy followed for their integration.

6.2.1. Description of the production process

The production process studied in this chapter is the same that was presented in Chapter 3. However, due to problems in data collection, changes in the inventory occurred with respect to the sources of energy employed in the factory. Thus, the updated inventory used during the development of this chapter is presented in Table 6.1.

The interest of the company was to develop a simplified tool to specifically evaluate the environmental impact of the tailoring process, so that improvements carried out throughout the years in the environmental performance could be properly assessed and identified, at the same time that comparisons among different plants of the company could be established. The process selected for the development of the tool corresponded to a factory located in Arteixo (NW Spain), where cotton jackets are manufactured and was described in Chapter 3 (section 3.2.1). An item of clothing (jacket) was selected as functional unit to facilitate comparisons among production years or factories.

The boundaries established corresponded to a gate-to-gate analysis, focused on burdens derived from the tailoring production process. However, it must be noticed that, under the EF approach, impacts beyond the inventory data boundaries were included in the form of embodied energy, natural productivity of raw materials and electricity breakdown into primary sources of energy.

Table 6.1. Process inventory data ([a]estimated values).

			2002	2003	2004	2005
INPUT	Raw materials	Cotton fabric (kg)	643,402	651,881	798,199	919,504
		Stitch (kg)	-	-	15,800	35,500
		Lining (kg)	-	-	300,000	350,000
		Paper & cardboard (kg)	[a]5,867	[a]5,740	6,971	7,173
		Plastic (kg)	[a]32,153	[a]31,459	24,419	39,313
		Buttons (kg)	28,000	28,000	28,000	31,864
		Zips (kg)	13,500	8,100	6,300	7,164
		Labels (kg)	650	650	650	740
	Energy	Electricity (kWh)	236,193	210,660	322,059	386,621
		Wind power (kWh)	0	8,980	14,711	15,244
		Propane (kg)	0	96.3	123.9	133.9
		Gasoil (m^3)	61.924	35.470	19.547	34.054
		Natural gas (kWh)	-	485,411	1,045,137	1,105,012
		Water (m^3)	777.5	160.9	110.3	124.6
OUTPUT		Production (number of items)	519,399	508,188	558,078	635,055
	Air emissions	SO_2 (kg)	[a]575	[a]330	[a]182	[a]316
		NO_x (kg)	18,194	3,542	3,554	6,086
		CO (kg)	11,529	11,502	3,652	4,623
		CO_2 (kg)	261,901	184,975	196,896	262,527
	Similar to urban waste	Textile (kg)	81,765	83,353	104,632	119,065
		Paper & cardboard (kg)	[a]5,867	[a]5,740	6,971	7,173
		Plastic (kg)	[a]605	[a]592	660	740
		Urban waste	-	-	-	-
	Hazardous waste	Paint (kg)	-	-	-	1.185
		Batteries (kg)	1.492	14.967	4.825	2.378
		Fluorescent lamp (kg)	11.114	5.443	13.669	6.817
		Computers waste (kg)	-	3.402	3.586	92.265
		Oil filters (kg)	60.719	11.566	7.706	4.756
		Mineral oil (kg)	104.430	115.658	100.823	-
		Contaminated containers kg)	0.746	1.565	4.594	3.171

6.2.2. Environmental evaluation methodologies

The protocols followed for the application of the environmental evaluation methodologies selected in this chapter, as well as the materials employed, are described below.

6.2.2.1. Energy and Material Flow Accounting

The production process of the dressmaking factory was modeled using the software Umberto® 5.5 (ifu and ifeu, 2005). With this purpose, a project was created and the materials (in Umberto both substances and forms of energy are referred to as materials) involved in the production process were defined and classified into the categories established in the inventory collected in Table 6.1. Then, different scenarios were created for the years studied (2002-2005).

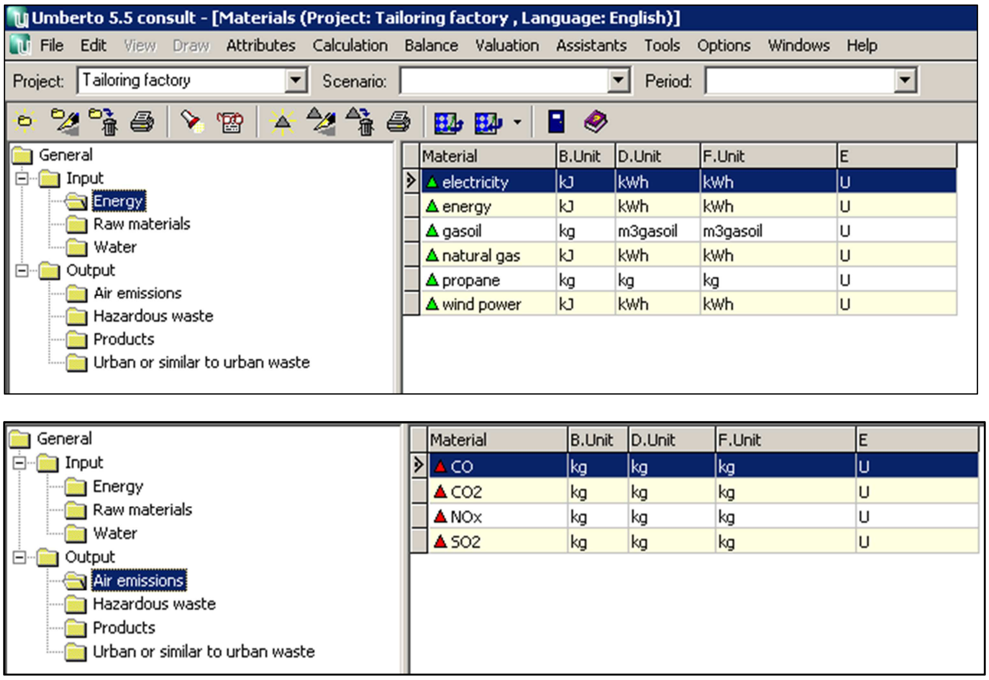


Figure 6.1. Definition of materials and groups of materials for the project, showing the specification for energy and air emissions.

All input materials and products were defined as “good” materials (green triangle), while air emissions and wastes were considered as “bad” materials (red triangle), as it can be observed in Figure 6.1.

The basic units are those into which all values are converted when a calculation is conducted (kg for mass-bound substances and kJ for forms of energy). To facilitate the comparability with the inventory data, entry units were defined using a conversion formula and established as display units. In the inventory (Table 6.1) most of the substances were expressed in kg except for water and gasoil, for which volume display units (m^3 water and m^3 gasoil, respectively) were defined using the density as conversion factor. Similarly, kWh was selected as display unit for electricity, wind power and natural gas.

A main network showing the input and output flows of the process was constructed, while the energy supply and the different stages of the dressmaking manufacture were modeled in subnets to make the visualization of the process more intuitive. All places connected to the subnet transition must remain the same for both levels. These places are the interface between the upper network level and the subnet level and are called port places (they are marked with a colored square in the symbol for the place). Two methods were employed for the specification of transitions: 1) input/output relations, i.e. introducing coefficients that represent the interrelations of the flow quantities; 2) defining the relations between input and output flows by a set of mathematical functions.

Apart from the sources of energy consumed in the factory, an additional material named energy was defined (Figure 6.1) to serve as common flow in the energy conversion subnet once proper conversions were conducted (all sources of energy were finally consumed in the form of electricity). Besides, to be consistent with the functional unit selected, the number of products was selected as reference flow.

6.2.2.2. Ecological Footprint

The same protocol described in Chapter 3 was followed to assess the EF of the tailoring process, but considering the updated inventory. Additionally, the results obtained in Chapter 4 (section 4.3.3) for the hazardous wastes applying the developed methodology were included.

6.2.2.3. Life Cycle Assessment

The structure defined in the ISO 14040 series standards (ISO, 2006), described in Chapter 2, was followed in this chapter. In a first approach, professional LCA software, SIMAPRO 7.0 (Pré Consultants, 2010), was used; hence, when all inputs/outputs in Table 6.1 were introduced, SIMAPRO appraised the studied case taking into account all the burdens associated to the processes included in its databases. This meant, for example, including the fabric or the plastic manufacturing processes. Consequently, the system boundaries were distorted and impacts not directly derived from the dressmaking plan were also evaluated, thus falling outside of the scope established in this chapter.

As a consequence, instead of a cradle to grave assessment, a complementary approach to the gate to gate analysis conducted with the EF was preferred. Thus, only air emissions released in the cogeneration units (that were excluded from the EF) were introduced in the software (CO_2 indirect emissions from the electricity breakdown and embodied energy in materials were already appraised with the EF). The purpose of applying professional software first was to collect information regarding the most adequate categories to reflect the impact of emissions and how to classify them. The normalized profile was used to determine the most relevant impact categories. The impact assessment method from the Dutch Institute of Environmental Sciences was used (CML, 2000).

After the previous analysis using SIMAPRO, for the simplified tool implemented in MS Excel® emissions were classified into two categories: Global Warming Potential (GWP; CO_2 and CO) and Acidification Potential (AP; SO_2 and NO_x), from which two environmental impact indicators additional to the EF were derived, expressed in CO_2 equivalent and SO_2 equivalent, respectively. The characterization factors employed were collected in Table 6.2.

Table 6.2. Characterization factors used (CML, 2000)

Pollutant	Global Warming Potential 100 years (GWP100)	Acidification Potential (AP)
CO_2	1.00	
CO	1.53	
NO_x		0.50
SO_x		1.20

6.2.3. Combination and integration of methodologies

The complementary characteristics of the environmental evaluation methodologies used in this chapter were widely discussed in Chapter 1 (section 1.4). Here, the specific protocol followed for the combination of EFMA, EF and LCA to achieve a comprehensive assessment of the dressmaking factory is detailed.

Although the research conducted in this chapter mainly aimed to achieve an aggregated indicator to measure the whole environmental impact of the factory, it was also important to carry out a differentiated analysis of material and energy flows to find out the major contributors to this impact and, therefore, the areas that required a prior intervention.

Regarding the integration between EF and LCA, two approaches were proposed: 1) combination of results from EF (evaluation of materials, energy consumption and wastes generated) and characterized categories from LCA for air emissions released in the cogeneration units, as explained in section 6.2.2; 2) incorporating emissions into the EF by means of absorption factors including (i.e. taking the characterized categories as basis) or not (i.e. using the original emission flows) weighting factors, as discussed in Chapter 5. Figure 6.2 shows a scheme representing these options. For the integration, global instead of site-specific absorption factors for global warming (IPCC) and acidifying emissions (European level) were selected in this chapter to assure comparability with estimates from section 6.2.2.2. Further, the transformation into global hectares was required to homogenize units. In this respect, emissions were assumed to be absorbed in forest areas. The integration was proposed on the basis of a precautionary approach, i.e. considering mutually exclusive lands (Venetoulis and Talberth, 2008) given the lack of knowledge about how a certain kind of land could either absorb CO_2 and acidifying emissions at the same time, which falls out of the scope of the study. It must be remarked that, for option 1, the analysis of impact categories from LCA is not restricted to GWP and AP.

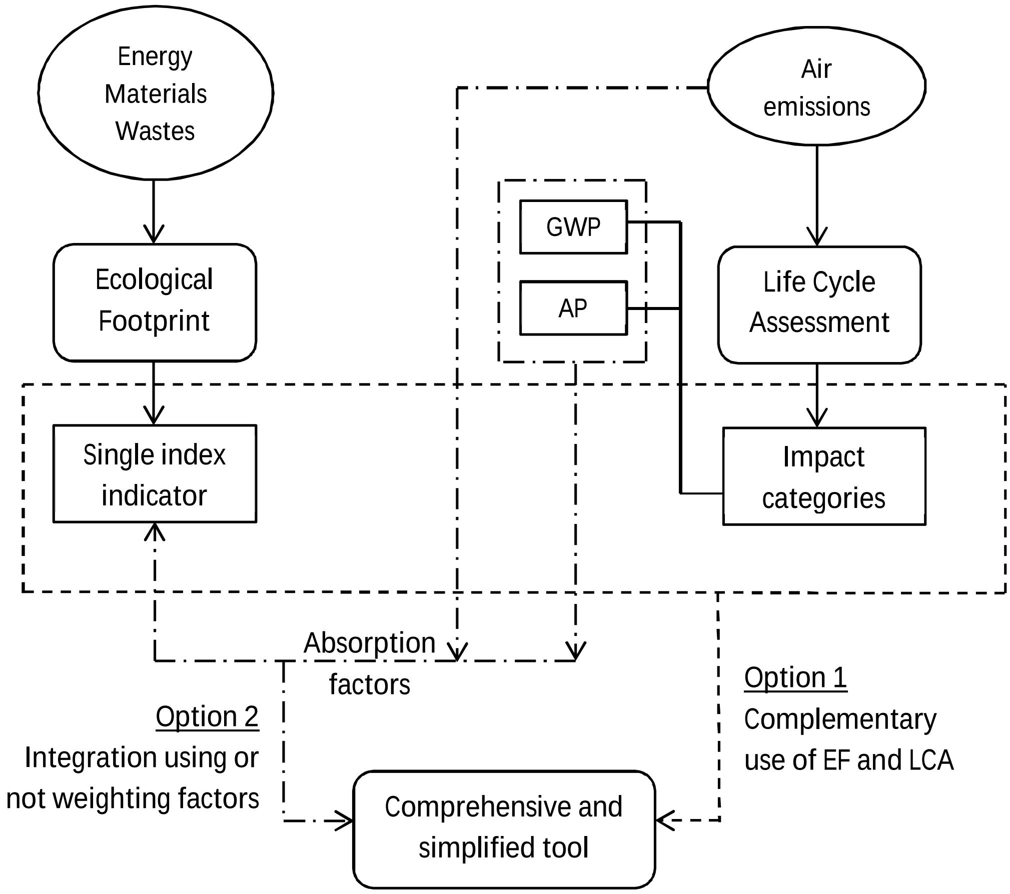


Figure 6.2. Scheme for the integration of EF and LCA.

6.2.4. Sensitivity analysis

Sensitivity analyses based on Monte Carlo simulations were carried out to assess how changes in the variables may affect the final value of the EF. To do this, the software Crystal Ball® 7.2.2 Edition was used (Decisioneering, 2005). All variables were varied randomly at the same time (using 10,000 iterations and setting the confidence level at 95%) according to the distributions stated. On the basis of the previously developed tool, data from year 2005 were used as reference since they were considered the most representative of future tendencies. Triangular distributions, with a general 20% margin of variation, were supposed for each variable (Table 6.3).

Table 6.3. Distributions considered for the sensitivity analysis (data from year 2005).

		Variable	Units	Type	Distribution[(a)]
INPUT	Raw materials	Cotton fabric	kg	Triangular	919,504 (735,603-11,034,045)
		Wool stitch	kg	Triangular	17,500 (0-35,000)
		Lining	kg	Triangular	350,000 (280,000-420,000)
		Paper & cardboard	kg	Triangular	7,173 (5,738-8,608)
		Plastic (packaging)	kg	Triangular	39,313 (31,450-47,176)
		Plastic accessories	kg	Triangular	15,932 (0-31,864)
		Metal accessories	kg	Triangular	7,164 (0-8,597)
		Wood accessories	kg	Triangular	15,932 (0-31,864)
	Energy	Electricity	kWh	Triangular	386,621 (309,297-463,945)
		Wind power	kWh	Triangular	15,244 (12,195-18,293)
		Propane	kg	Triangular	67.0 (0-133.9)
		Gasoil	m^3	Triangular	17.0 (0-34.0)
		Natural gas	kWh	Triangular	1,105,012 (884,010-1,326 014)
		Water	m^3	Triangular	124.6 (99.7-149.5)
OUTPUT		Products	items	Triangular	635,055 (508,044-762,066)
	Similar to urban waste	Textile	kg	Triangular	119,065 (95,252-142,878)
		Paper & cardboard	kg	Triangular	7,173 (5,738-8,608)
		Plastic	kg	Triangular	740 (592-888)
		Textile recycling	%	Triangular	50 (0-100)
		Paper recycling	%	Triangular	50 (0-100)
		Plastic recycling	%	Triangular	50 (0-100)

[(a)]Data are expressed in a yearly basis.

Simulations considering and excluding emissions from the EF estimate were assessed and compared. In order to correlate emissions with the sources of energy employed in the factory, emissions factors (Table 6.4) were introduced instead of using the inventoried data for these particular input variables.

Table 6.4. Emission factors for the sensitivity analysis.

Energy source	**Pollutant**	**Emission Factor**	**Units**	**Source**
Natural gas	CO_2	56.1	t/TJ	IPCC, 1996
	CO	40	lb/billion Btu	US EIA, 1999
	NO_x	92	"	"
	SO_2	1	"	"
Electricity	CO_2	0.4556	kg/kWh	IDAE, 2007
Gasoil	CO_2	74.1	t/TJ	IPCC, 1996
	NO_x	61	g/GJ	US EPA, 1998
	SO_2	[a]$4.92 \cdot S$	g/kWh	US EPA, 1985
Propane	CO_2	63.1	t/TJ	IPCC, 1996
	NO_x	66	g/GJ	US EPA, 1998
	SO_2	7.2	"	"

[a]S = sulfur content in fuel. A limit of 0.2% was considered (Spanish Government, 2005).

6.3. Results and discussion

The integration of LCA and EF methodologies led to the development of a comprehensive tool for the environmental evaluation of the dressmaking process. Thus, it would be useful for decision making in some industrial environmental concerns as process selection, product comparison or operational unit burdens analysis. A more in depth exploration is provided by means of an EFMA conducted in Umberto®.

6.3.1. EMFA analysis

The main input and output categories were used to model the network presented in Figure 6.3, where the dressmaking process (transition T1) and the energy supply (transition T2) were included as subnets. Input flows were energy (P1), raw materials (P2) and water (P3); meanwhile, output flows were the products – jackets- (P4), air emissions (P5), urban or similar to urban waste (P6) and hazardous waste (P7). The number of products obtained was selected as reference flow; thus, the corresponding arrow was specified (it is marked in pink in Figure 6.3).

Different sources of energy (collected in place P8) were consumed in the process but all of them were finally used in the form of electricity. Thus, a subnet was created to homogenize the energy supply (Figure 6.4).

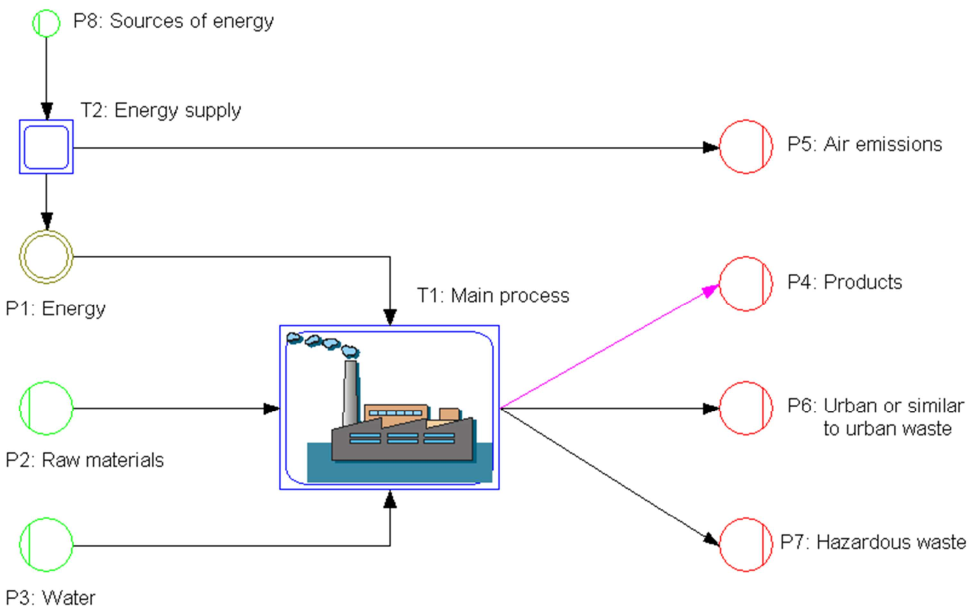


Figure 6.3. Main network of the tailoring process implemented in Umberto®.

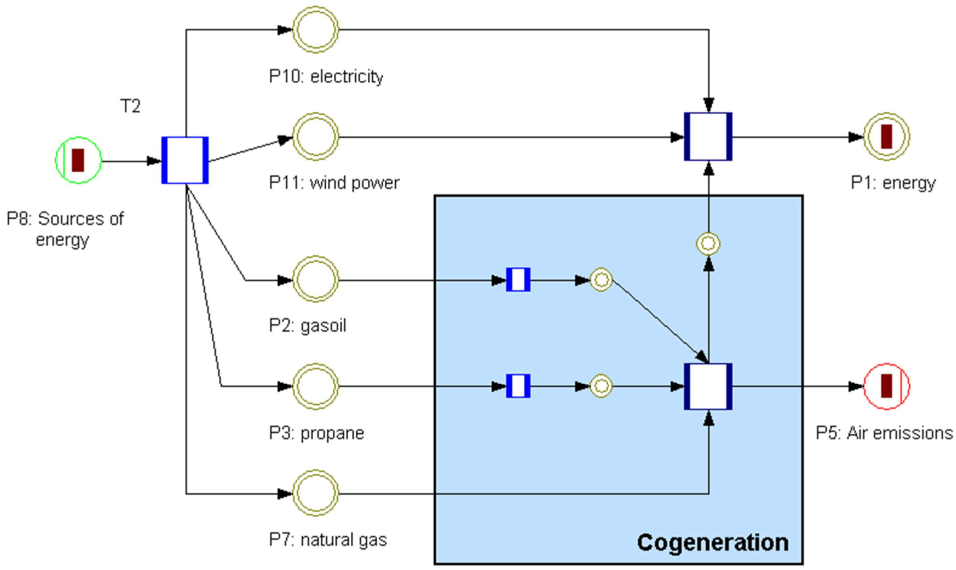


Figure 6.4. Subnet modeling the energy supply to the dressmaking factory.

Gasoil (P10), propane (P11) and natural gas (P7) were combusted in cogeneration units to obtain electricity. The consumption of the former two was originally given as material flows; therefore, intermediate transitions were defined to do proper conversions into energy units. An additional transition was included to aggregate the total electricity generated in cogeneration, to which air emissions were assigned (P5). Finally, the electricity from wind power and from the external power supplier was added to appraise the total flow of energy (in the form of electricity).

The stages that compound the main process were modeled in the subnet presented in Figure 6.5. A similar structure to that used in Chapter 3 to schematize the flowsheet of the dressmaking factory was maintained. Apart from cutting (T1), external tailoring (T2), ironing (T3), packing (T4) and labeling (T5), an additional transition (T7) was included to represent those general operations implicit in the factory operation but that could not be allocated to any of the other stages (e.g. hazardous wastes generation or electricity consumption for lighting and heating). Duplicates of “Raw materials” and “Energy” places were created to avoid an excessive use of arrows crossing the network that may generate confusion.

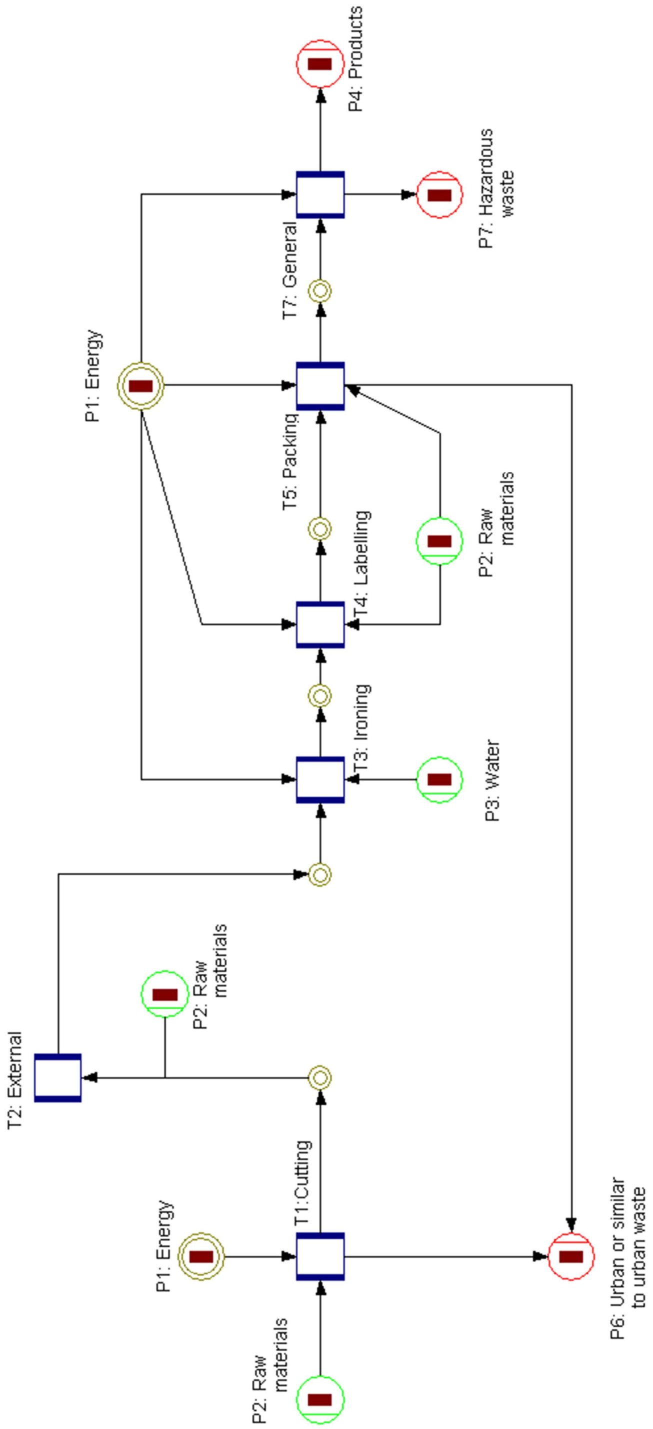


Figure 6.5. Subnet modeling the dressmaking process.

The specifications for transitions T1 and T7 from the dressmaking process subnet are shown in Figures 6.6 and 6.7, respectively, as an example. It must be observed that these specifications were expressed on the basis of the functional unit selected, i.e. an item of clothing produced. Thus, by specifying the arrow for final production (pink arrow in Figure 6.3) all flows were calculated. In Figure 6.6, the quantities of resources consumed during the cutting stage are shown. The pieces of the jacket are first drawn on paper and then placed on the fabric, putting plastic over them to avoid undesired movements during the cutting process. Therefore, the paper and plastic used are converted into residues at the end of the operation, as well as the textile discards from the cotton fabric and the lining. All these wastes (red triangle) are sent to place P6, while the pieces of the jacket follow the production process to the external tailoring (T2).

Similarly, the specification for transition T7 (Figure 6.7) shows the finished products, which do not suffer from any transformation in this transition (they flow from P5 to P4), the energy consumption (supplied by place P1) and the hazardous wastes generated in maintenance operations (sent to place P7).

Input / Output | Allocation Rules | Cost Center Costs | Cost Drivers | Constraints

Place	Var	Coefficient	Material	B. Unit
P2	X00	0.01	paper & cardboard	kg
P2	X01	1.45	cotton fabric	kg
P2	X02	0.00	plastic	kg
P1	X03	4,917.57	energy	kJ
P2	X04	0.55	lining	kg

Var	Place	Material	Coefficient	B. Unit
Y03	P9	cotton fabric	1.26	kg
Y04	P6	paper & cardboard out	0.01	kg
Y05	P6	plastic out	0.00	kg
Y06	P6	textile	0.19	kg
Y07	P9	lining	0.55	kg

Figure 6.6. Specification of transition “T1: Cutting” from the subnet in Figure 6.5.

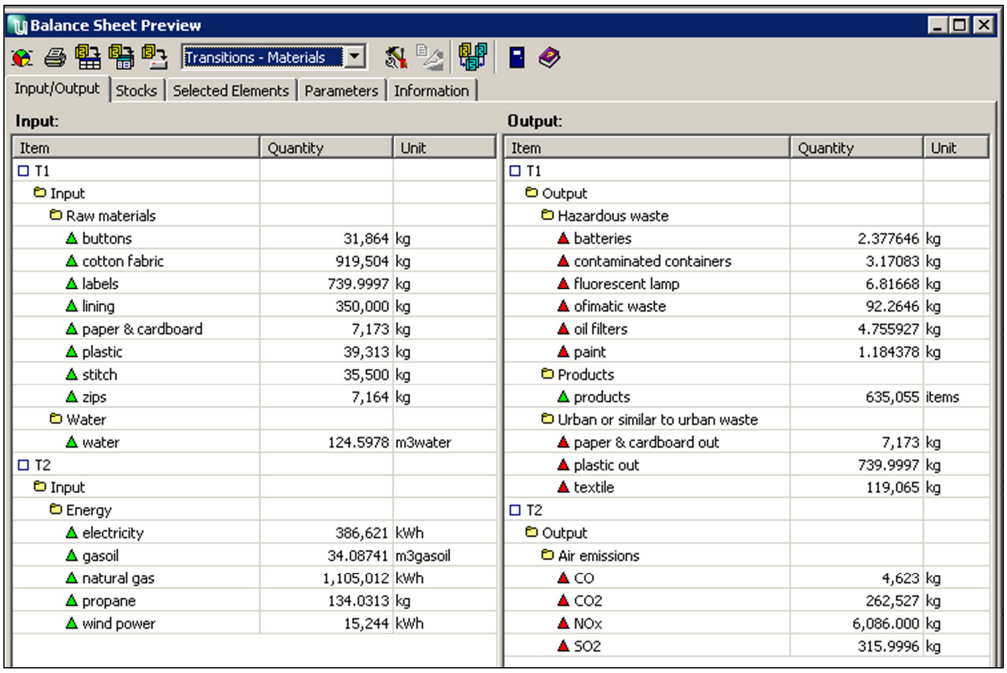
Input / Output | Allocation Rules | Cost Center Costs | Cost Drivers | Constraints

Place	Var	Coefficient	Material	B. Unit
P1	X00	2,783.530	energy	kJ
P5	X01	1	products	items

Var	Place	Material	Coefficient	B. Unit
Y00	P7	batteries	3.744E-6	kg
Y01	P7	contaminated containers	4.993E-6	kg
Y02	P7	fluorescent lamp	1.0734E-5	kg
Y04	P7	ofimatic waste	0.000145286	kg
Y05	P7	oil filters	7.489E-6	kg
Y06	P7	paint	1.865E-6	kg
Y10	P4	products	1	items

Figure 6.7. Specification of transition “T7: General” from the subnet in Figure 6.5.

Once all the transitions were specified, the network was calculated and the balances were obtained. Figure 6.8 shows the balance sheet for the main network, grouping the materials by transition. The correspondence with inventory data in Table 6.1 was very good. The total energy estimated in subnet T2 did not appear here since it was defined as a connection place (P1) in the main network. Conversely, when the balance sheet for the dressmaking process subnet was conveyed (Figure 6.9), the flow representing the total electricity (energy) consumed in the factory appears instead of the original sources of energy as they were not input flows to this subnet.



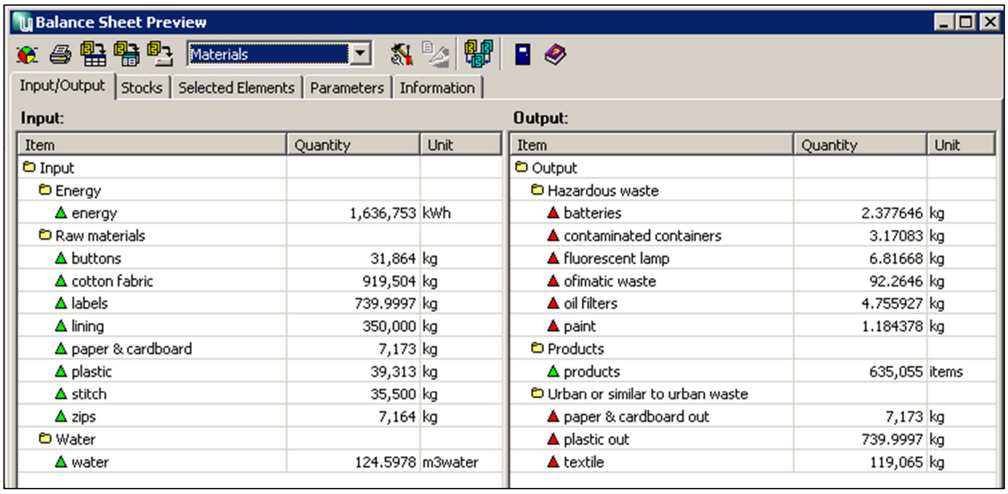
Input:

Item	Quantity	Unit
T1		
Input		
Raw materials		
buttons	31,864	kg
cotton fabric	919,504	kg
labels	739.9997	kg
lining	350,000	kg
paper & cardboard	7,173	kg
plastic	39,313	kg
stitch	35,500	kg
zips	7,164	kg
Water		
water	124.5978	m3water
T2		
Input		
Energy		
electricity	386,621	kWh
gasoil	34.08741	m3gasoil
natural gas	1,105,012	kWh
propane	134.0313	kg
wind power	15,244	kWh

Output:

Item	Quantity	Unit
T1		
Output		
Hazardous waste		
batteries	2.377646	kg
contaminated containers	3.17083	kg
fluorescent lamp	6.81668	kg
ofimatic waste	92.2646	kg
oil filters	4.755927	kg
paint	1.184378	kg
Products		
products	635,055	items
Urban or similar to urban waste		
paper & cardboard out	7,173	kg
plastic out	739.9997	kg
textile	119,065	kg
T2		
Output		
Air emissions		
CO	4,623	kg
CO2	262,527	kg
NOx	6,086.000	kg
SO2	315.9996	kg

Figure 6.8. Balance sheet for the main network for the year 2005 scenario.



Balance Sheet Preview

Materials

Input/Output | Stocks | Selected Elements | Parameters | Information

Input:

Item	Quantity	Unit
Input		
Energy		
energy	1,636,753	kWh
Raw materials		
buttons	31,864	kg
cotton fabric	919,504	kg
labels	739.9997	kg
lining	350,000	kg
paper & cardboard	7,173	kg
plastic	39,313	kg
stitch	35,500	kg
zips	7,164	kg
Water		
water	124.5978	m3water

Output:

Item	Quantity	Unit
Output		
Hazardous waste		
batteries	2.377646	kg
contaminated containers	3.17083	kg
fluorescent lamp	6.81668	kg
ofimatic waste	92.2646	kg
oil filters	4.755927	kg
paint	1.184378	kg
Products		
products	635,055	items
Urban or similar to urban waste		
paper & cardboard out	7,173	kg
plastic out	739.9997	kg
textile	119,065	kg

Figure 6.9. Balance sheet for the dressmaking process subnet for the year 2005 scenario.

Figures 6.10 and 6.11 represent the Sankey diagrams for the energy and the dressmaking process subnets, respectively. Sankey diagrams are flow charts in which the width of the arrow is proportional to the flow quantity. Therefore, they provide a very intuitive graphical analysis of the distribution of energy and material flows in a network. Thus, from Figure 6.10 it could easily be observed that natural gas was the main supplier of energy to the factory, followed by the electricity from an external power supplier. Figure 6.11 shows how the total energy consumed in the factory is distributed among the main stages of the dressmaking process. The allocation of electricity consumption was estimated on the basis of the power of the machines employed in each section of the factory, the number of machines and the working hours. The cutting stage represented the major consumption of energy, mostly due to the heat sealer used to fix the plastic over the fabric to avoid undesired movements. Precision during the cutting of the fabric was essential to prevent errors that may originate important economic losses and the generation of high amounts of textile waste. This equipment worked continuously because it was proved that the consumption of energy was lower than when it was turned on and off periodically. In this respect, strategies for energy savings were difficult to establish.

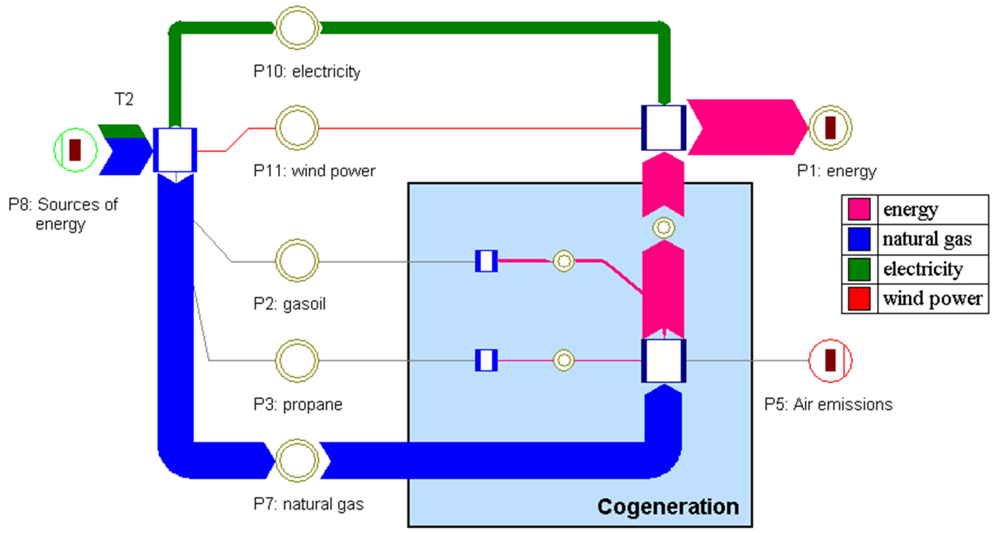


Figure 6.10. Sankey diagram for energy flows in the energy subnet for the year 2005 scenario.

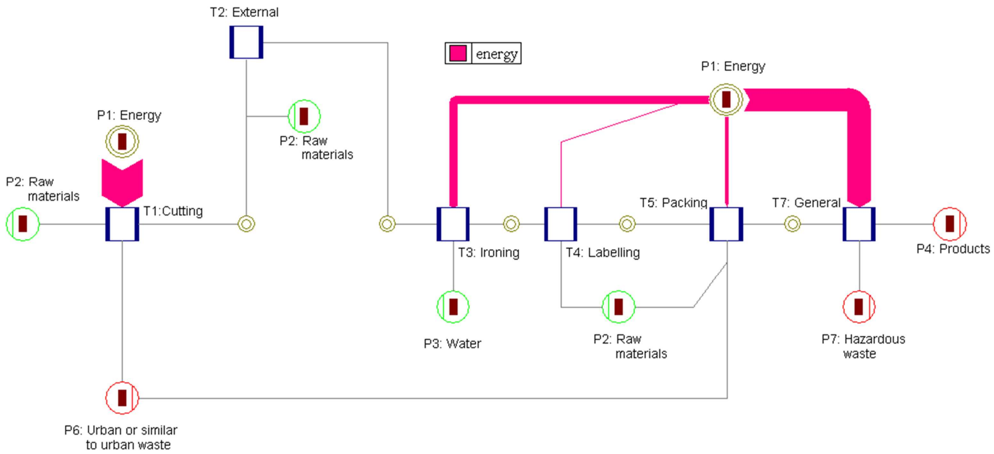


Figure 6.11. Sankey diagram for energy flows in the dressmaking process subnet for the year 2005 scenario.

6.3.2. EF estimates

The results indicating the contribution to the total EF of the considered categories are shown in Figure 6.12, where EF is expressed in global hectares (gha) to allow different lands comparison (Rees and Wackernagel, 1996; Wiedmann and Lenzen, 2007; Galli et al., 2007). Be aware that results from this section refer to the energy, waste (including hazardous waste) and resources categories in Figure 6.12, and that the emissions contribution is later analyzed in section 6.3.4. An increasing tendency in the total EF was observed (from 1,965 gha in year 2002 to 2,868 gha in year 2005), mainly due to an increase in the amount of fabric employed. This strongly influenced the overall result since the resources category represented more than the 80% of the total value of the EF, mainly because of the high weight of the cotton textile, which requires an important area of productive arable land for its growth. The increment in the relative EF was a little bit softer because of the increase in the annual production (the production varied from 519,399 jackets in 2002, with a relative EF of 37.8 gm^2/jacket, to 635,055 in 2005, posing to a relative EF of 45.2 gm^2/jacket). When analyzing a dressmaking factory, the particularity of fashion tendencies dependence has to be considered. Thus, during the design phase, the type and amount of materials to be employed are defined according to aesthetic criteria rather than environmental ones. Consequently, it may happen that the jacket design in a certain year employs a major quantity of material, this not strictly meaning that efficiency in the resource use has decreased (the percentage of material discarded during the cutting phase is maintained), but that fashion preferences have changed. However, a worsening of the environmental performance occurs, situation reflected in the EF figures obtained, from which the recommendation of moving towards an ecodesign of products approach is derived. The design of the jacket should pursue the fulfillment of a certain function, i.e. serving as a piece of clothing, rather than following fashion tendencies without considering that these may lead to a worse environmental performance.

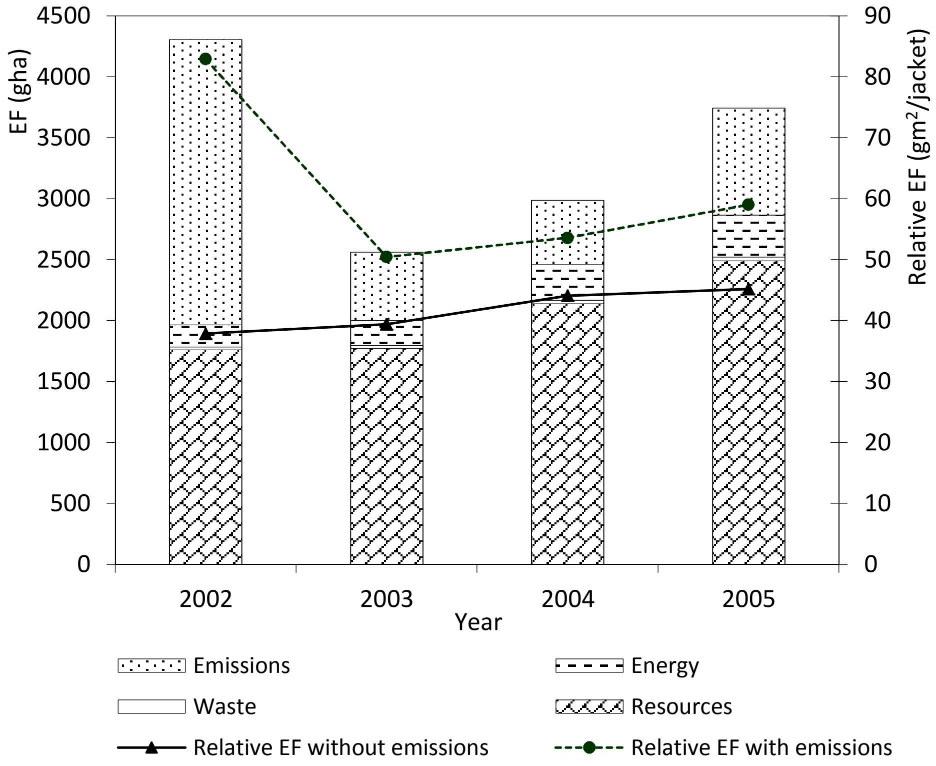


Figure 6.12. Contribution of main categories to the total EF during the period 2002-2005. When emissions (dots) are considered, the new energy category comprises the old energy category (dashes) and the emissions.

Through the analysis of the period 2002-2005, it can be stated that the EF acted as a sensitive indicator to changes in the operational conditions of the plant. Thus, it offered a good measure of its environmental performance (mainly energy and raw materials consumption) and made it easy to analyze its evolution throughout the years. However, the study could be improved if other environmental burdens not included in the EF methodology, like the impact associated to air emissions, were also taken into account.

6.3.3. LCA estimates

The results obtained using the CML factors in SIMAPRO are presented in Figure 6.13. Gases released affected 5 impact categories: global warming, human toxicity, photochemical oxidation, acidification and eutrophication. To assess their degree of significance, the normalization graph using West Europe factors was analyzed (Figure 6.14). The three more significant impacts here were acidification, eutrophication and global warming; hence, it was observed that the human toxicity category was almost negligible and that photochemical oxidation was not a relevant impact at all. The subjectivity inherent to the normalization phase (at deciding the reference area if data for the local region is not available) must not be forgotten, and it is the author of the LCA who must finally decide whether an impact is important for the study or not. Furthermore, when the impact categories are established it is important to avoid overlaps (Guineé, 2001). In this case, GWP and AP included all the emissions released in the factory and therefore their impact was totally measured. These were selected instead of any others because the different stakeholders are especially concerned with global warming (which may even have economic implications), and acidification is a more direct impact than eutrophication; besides, acidification presented a higher value than eutrophication in all the years considered (Figure 6.14). On the other side, human toxicity, for example, would be important if the fabric manufacturing was included in the boundaries of the study. For cotton, the application of pesticides during the cropping stage may be of concern if organic cotton is not employed. Dyeing and printing during the manufacturing of the fabric also require the application of chemicals that may pose a hazard to human health (Chouinard and Brown, 1997; Harner et al., 2001).

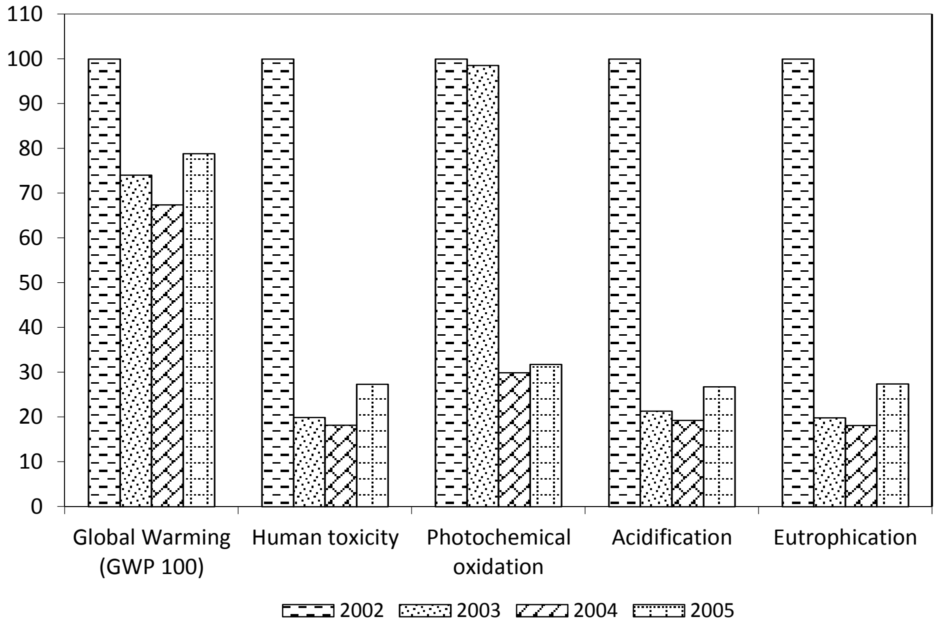


Figure 6.13. LCA characterization phase using SIMAPRO 7.0 and CML methodology.

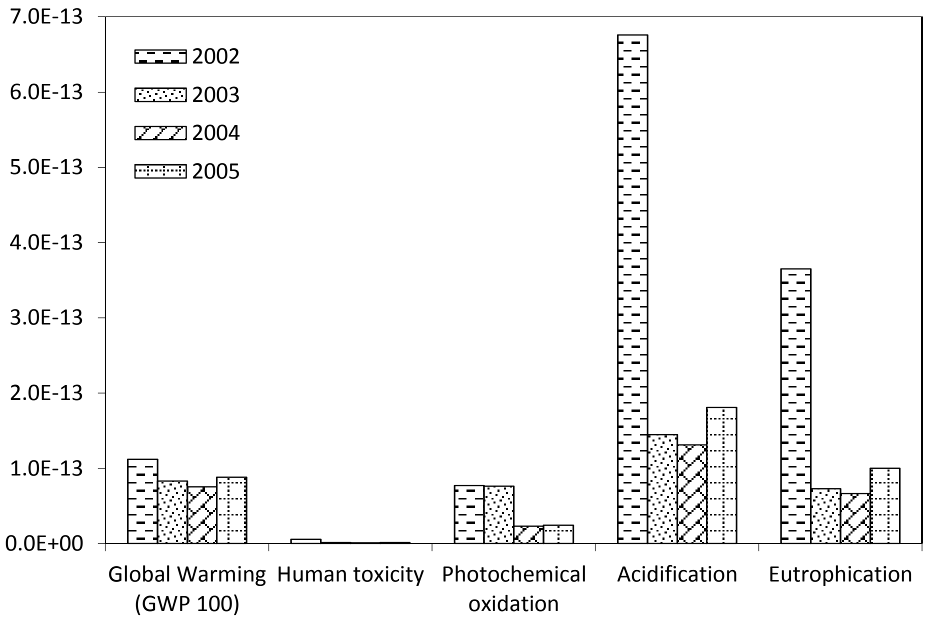


Figure 6.14. LCA normalization phase using SIMAPRO 7.0 and CML methodology.

Once the categories were selected in Simapro, the LCA methodology was implemented in the simplified tool, together with the EF assessments, to appraise the potential damages the air emissions may cause in the environment. The characterization phase was carried out for the GWP and the AP categories (Figure 6.15). Acidification emissions run parallel to the gas oil consumption. Thus, the company's policy of replacing this kind of fuel by cleaner ones, like natural gas or alternative energies (e.g. wind power or solar energy), meant and important reduction of the AP category characterized value. However, the unexpected rise in 2005 could be explained by an additional demand of energy caused by the increasing textile production (a 13.8% increase in relation to 2004 production).

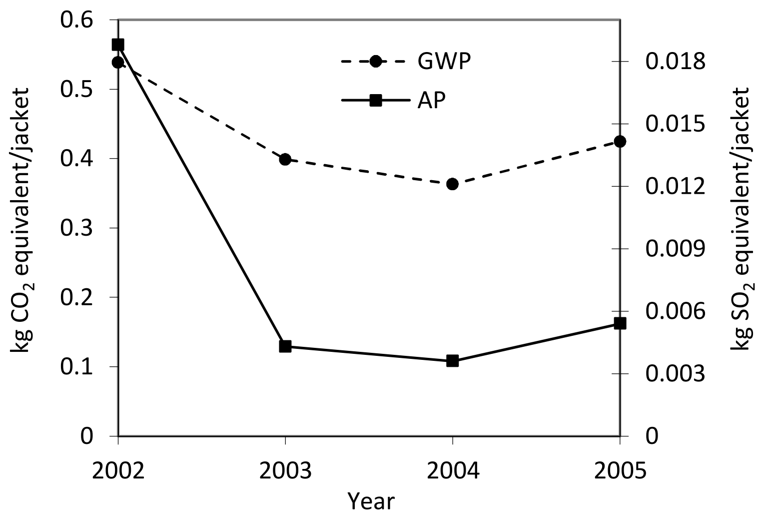


Figure 6.15. Air emissions environmental burden analyzed through LCA.

6.3.4. EF of emissions

The integration of emissions released in the cogeneration was carried out using absorption factors for greenhouse gases (GHG) and acidifying emissions. The estimates obtained taking LCA categories as basis or emissions independently were collected in Table 6.5.

Table 6.5. Emissions released in the factory converted into required space of land using absorption factors.

Year	A. Emission flows (ha)		B. LCA categories (ha)		Difference A-B (%)	
	GHG	Acidifying emissions	GWP	AP	GHG	Acidifying emissions
2002	76.37	1596.0	76.24	1529.2	0.17	4.18
2003	55.38	338.6	55.24	338.6	0.24	1.79
2004	55.26	311.8	55.22	311.8	0.07	3.37
2005	73.58	534.7	73.53	534.7	0.07	3.34

With a characterization factor of 1.53 for CO, the effect of using any of the two options is almost negligible in this work for GHG, also considering that, according to the inventory in Table 6.1, CO emissions in the factory were much lower than CO_2 emissions. The effect is not significantly noticeable for acidifying emissions either. Weighting the severity of NO_x emissions meant halving their contribution (characterization factor of 0.5); however, by considering them as SO_2 equivalents the number of H^+ equivalents was duplicated. To be consistent with Chapters 4 and 5, a precautionary approach was preferred in this case and the incorporation of weighting factors was avoided.

After converting the area required for the assimilation of emissions into gha, the contribution of this category was conveyed in Figure 6.12, together with the previous EF estimates. This contribution was particularly noticeable in 2002. In this year, natural gas and wind power had not been incorporated as energy sources in the tailoring plant; consequently, a higher quantity of gasoil was employed in the cogeneration units thus increasing the amount of emissions released (especially SO_2 and NO_x) and their contribution to the EF.

After incorporating emissions, the energy category would acquire major relevance and the sustainability of the tailoring process would not only remain on the resources consumption, as it seemed to happen according to first assessments (further detail on this analysis was provided in Chapter 3). It is observed that the exclusion of emissions from the EF accounts clearly underestimated the real environmental impact. Nonetheless, these estimates were made under the assumptions underlying EF methodology that suggest that these areas of land are mutually exclusive (Venetoulis and Talberth, 2008), i.e. an area of land used for

CO_2 absorption cannot also be used for SO_2 absorption or cotton growing. This assumption may lead to an overestimation of the EF (Holmberg et al., 1999), but if an additive approach had been selected instead, the effects of incorporating emissions to the analysis would be hidden, since the area required for the production of resources consumed in the process is higher than the area estimated for the absorption of emissions (Figure 6.10). Given that the developed tool was intended to monitor the environmental performance of the factory and to provide supporting information to the company, the results of using an additive approach would offer misleading advise (e.g. the substitution of gasoil for natural gas would not be reflected in reports obtained from EF appraisals, and the company could think that it is not worthy to carry out such initiative). Consequently, the mutually exclusive approach was found more meaningful and preferable in this case. Finally, in all cases a negligible contribution of the waste category, that included the hazardous wastes evaluated in Chapter 4, was detected (Figure 6.12). Although the impact of hazardous wastes in terms of area requirements was not very significant, their potential hazard derived from their toxic nature would be of major concern and, therefore, the application of other methodologies like Environmental Risk Assessment (ERA) would be recommended.

In general, the results obtained show to what extent the EF was being underestimated and how important it would be to include all the inventory data in order to achieve a global indicator for measuring the degree of sustainability of the process and its whole environmental impact.

6.3.5. Sensitivity analysis

The sensitivity analysis to the EF of the dressmaking process was carried out including and excluding emissions.

6.3.5.1. Initial EF estimates excluding emissions

First of all, the influences in the total EF were analyzed (Figure 6.16). Two inputs, belonging to the resources category, were the major contributors to the EF variance. Cotton was the principal component of the jackets manufactured in the factory (20% variation margin was taken into account); meanwhile, the wool was employed in additional accessories depending on fashion tendencies (therefore the minimum value considered was zero). The high requirements of direct land (pasture land for the wool and arable land for the cotton) were responsible for this result. Dahllöf (2004) also found that the environmental burden incorporated by the use of natural fiber is higher than that obtained by the use of synthetic fiber.

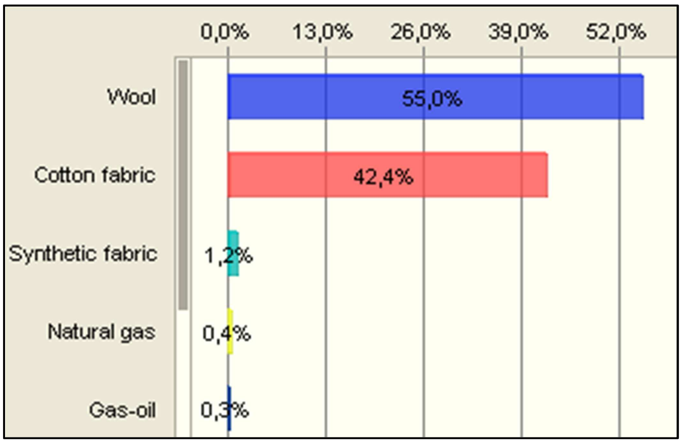


Figure 6.16. Main contributors to the total EF variance (without emissions).

However, before stating that synthetic fabric is environmentally better, other criteria should be taken into account (Nieminen-Kalliala, 2003), like the maximum content of specific harmful substances considered (e.g. in eco-labeling). On the other hand, EF accounts do not differentiate between more or less environmentally friendly agricultural production systems; i.e. an organic cotton cropping would not be better than a conventional one from the EF perspective, unless a higher yield is achieved (Venetoulis and Talberth, 2008).

The general examination conveyed that energy consumption was not significant in terms of sustainability. This was because the EF especially accounted for the intensity of consumption and depletion of natural resources and did not include air emissions in the estimate. Air emissions depended on the amount and sort of energy employed, as well as on the technology used. Thus, if these issues were taken into account, as proposed in this work, the results would be noticeably different (this was analyzed later on).

To be able to scrutinize the relative contribution of the different sources of energy, the energy main category was analyzed separately (Figure 6.17). A 20% variation margin was considered for each source of energy, although for propane and gasoil a minimum value of zero was stated since it was expected that these sources of energy were eliminated (Table 6.3). Natural gas was the main contributor to the energy category variance. This was expected since it was the main source of energy employed (60.4%). Although it only represented the 17.6% of the energy consumed in the factory, gasoil showed a similar contribution to the EF variation. Therefore, to diminish the EF, apart from prioritizing energy savings, higher efforts in substituting gasoil for cleaner sources of energy (e.g. natural gas, renewable energies) should be made.

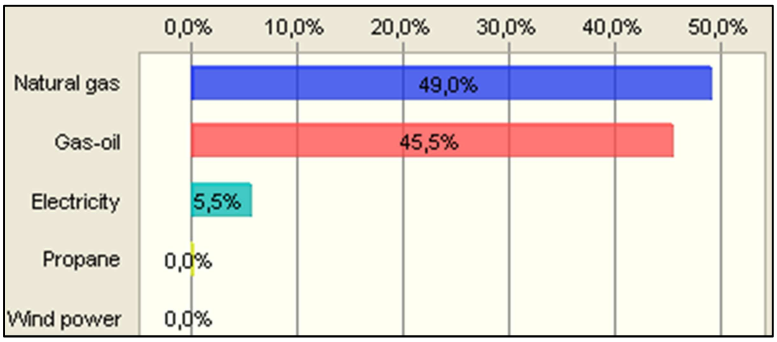


Figure 6.17. Contributions to energy category variance.

The waste category was also studied separately. Given the low contribution obtained for hazardous wastes, these were excluded from this analysis for simplicity. The results obtained in Figure 6.18 corresponded to a sensitivity analysis in which each variable was tested independently in the range of specified percentiles. This meant that, while analyzing one variable, the tool froze the other ones at their reference values. The higher the slope of the curve, the larger the effect on the forecast. Hence, the textile waste and the paper and cardboard waste were the variables that affected the most the contribution to the EF within the waste category. In addition, the waste category analyzed using the same correlation-based method employed in Figure 6.17 led to a contribution to the variance of the textile waste higher than 90%. This result was reasonable because it represented the greatest amount of waste generated in the factory (between 10% and 15% of the textile material processed). However, limited actions could be adopted to improve this environmental aspect, since during the cutting stage patterns were already carefully distributed so that losses of fabric were minimized to a level difficult to reduce. This was an example of how the application of the Best Available Techniques (BATs) allowed for the minimization of environmental burdens (European Commission, 2003b; Nieminen et al., 2007). A negligible effect of waste recycling was observed in Figure 6.18, since the curves were almost horizontal (the slope of the curves was softly negative because recycling percentage had a negative influence in the EF). Actually, this was the reason why a different analysis (fixing only one variable at a time) was preferred to test the waste category sensitivity.

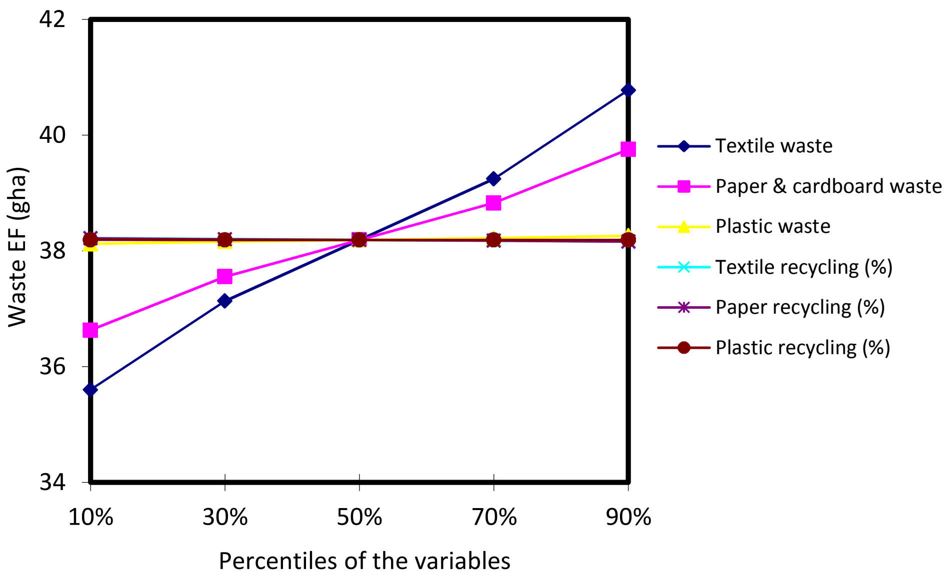


Figure 6.18. Sensitivity analysis in the EF waste category.

6.3.5.2. EF estimates integrating emissions

The energy supply network was modeled again using the emissions factors collected in Table 6.4. In this case, electricity generation from propane and gasoil, as well as the emissions associated to all sources of energy, were calculated by defining mathematical equations in the transitions. The specification of the transition for propane is showed in Figure 6.19 as an example. The Sankey diagram in Figure 6.20 shows the distribution of emission flows. The major quantity of CO_2 emissions was associated to the consumption of natural gas, followed by electricity; meanwhile, NO_x emissions were mainly released in the gasoil cogeneration. Gasoil combustion was an important source of pollution in spite of its low contribution to energy supply, as it could be observed in the Sankey diagram for energy flows (Figure 6.10).

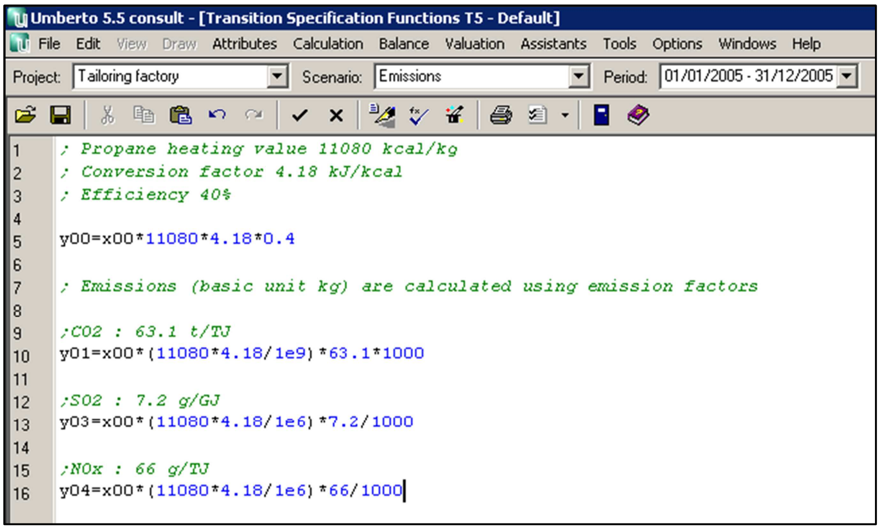


Figure 6.19. Specification of transition simulating the propane cogeneration unit.

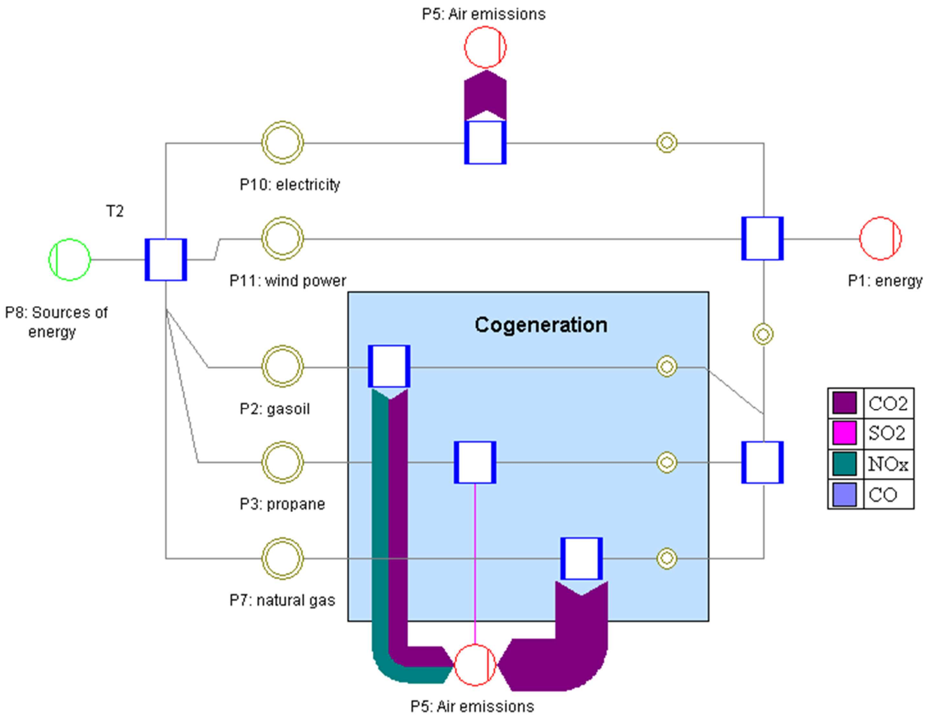


Figure 6.20. Sankey diagram (substance flows) for the energy supply network modeled using emission factors.

The new model was implemented in the simplified tool and the sensitivity analysis was conducted again. Air emissions were not independent variables but they varied according to energy consumption, thus being implicitly included in the energy category. As a consequence, a quantitative change in the kind of variables that mainly explained the variance of the total EF was detected. According to the analysis in section 6.5.3.1, input variables belonging to the resources category (e.g. wool and cotton) explained most of this variance; however, when including the emissions within the energy category, it was clear that gasoil was the contribution that mostly affected this variability (Figure 6.21). This was due to the contribution of acidifying emissions, which presented a high requirement of land for their assimilation. Therefore, in order to achieve a more sustainable management of the dressmaking process, strategies should not only focus on the reduction of resources consumption, but also in the incorporation of more environmentally friendly energies. In this respect, efforts should be carried out to remove gasoil and to promote a higher contribution of renewable energies.

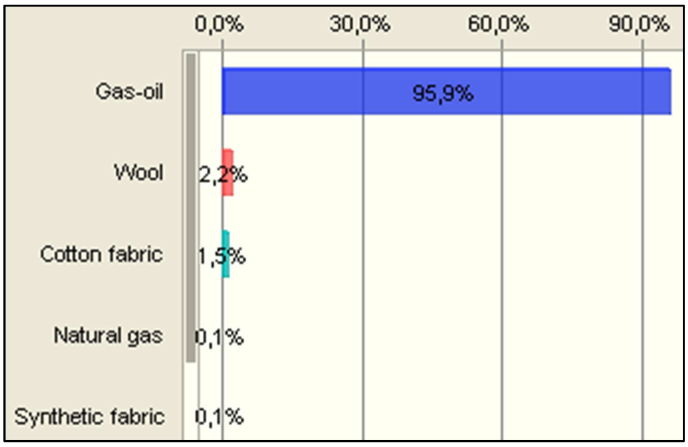


Figure 6.21. Principal contributors to the total EF variance (including emissions).

6.4. Conclusions

Three environmental evaluation methodologies were applied to complementary assess the performance of a dressmaking factory and to improve the simplified tool developed in Chapter 3. The EF widely fulfilled the characteristics of aggregated and simplified indicator sought for the case study. Prior estimates obtained from the application of this methodology conveyed its suitability to measure the impact derived from resources consumption. In this respect, the materials consumption was demonstrated to be a priority issue when measuring the sustainability of a dressmaking process. The introduction of the methodological proposals from Chapters 4 and 5 proved that the ability of the indicator to properly account for the area required for wastes (solid waste, air emissions, etc.) assimilation could be enhanced. As a consequence, a more comprehensive indicator was obtained, sensitive to changes in the different variables implied in the production process (fabric material, sources of energy, etc.) and thus providing more meaningful information at decision making.

The developed tool incorporated the complementary vision from LCA by means of two impact indicators, GWP and AP, based on the same gate-to-gate boundaries of the study. The possibility of their integrations into the EF was explored using absorption factors. Other impact categories identified when conducting the LCA were not included in the EF figure due to the lack of an appropriate method to do so. Although not included in the single indicator, the complementary approach provided by LCA was also presented. Further, in the near future, it was expected the removal of the sources of energy from which the emissions responsible for such environmental impacts were derived. Thus, the usefulness of the environmental evaluation tool was ensured.

Besides, EMFA was applied for a detailed study of energy and material flows, providing useful information at detecting stages where prior actions to reduce environmental loads should take place. The higher energy consumption in the production process occurred during the cutting stage, but minimization strategies had already been implemented. General activities like lighting and heating of the factory represented the second major energy consumption. Hence, actions like installing low power consumption lights, constraining the lighting to the areas that strictly need it or regulating the use of heating would lead to a significant decrease in energy consumption. Further, the mass balance conveyed that gasoil, in spite of its low contribution to energy flow, was one of the main sources of pollution of the factory. As a consequence, it was recommended its substitution for cleaner sources of energy.

Relative indicators were also derived to observe their evolution with time and to facilitate benchmarking among different similar production factories. This information would be useful for internal management or communication purposes. Finally, the findings obtained in the sensitivity analysis could also assist decision making to define sustainable management policies. In addition to those strictly related to the performance of the factory (e.g. energy consumption patterns already discussed), the management policies were strongly related to the incorporation of environmental criteria in the design of products. Therefore, the materials consumption should be minimized and the selection of fabrics should be based on environmental aspects rather than on tendency patterns.

As a final remark, the benefit of a joint application of environmental evaluation methodologies was proved. A more comprehensive analysis was obtained without adding unnecessary complexity to the developed tool.

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Chapter 7

Application of fuzzy logic to integrate environmental criteria in ecodesign

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Abstract

The ecodesign of a product implies that different potential environmental impacts of diverse nature must be taken into account considering its whole life cycle, apart from the general design criteria (i.e. technical, functional, ergonomic, aesthetic or economic). In this sense, an ecodesign tool integrating the criteria provided by three environmental evaluation methodologies, namely Ecological Footprint (EF), Life Cycle Assessment (LCA) and Environmental Risk Assessment (ERA), was developed on the basis of Fuzzy Logic (FL) reasoning and features. This idea enabled the decision making at process and product level taking into account the values of the different indicators at a time. The relative importance of each of them was established through the definition of membership functions as inputs to the fuzzy inference reasoning in the case of a specific product. As a result, a Fuzzy EcoDesign Index (FEcoDI) was obtained.

A well-known case study was used to support the development of the tool and to test it. In this respect, different packaging materials for a beverage bottle were considered to identify the most environmentally friendly option. After refinement on the basis of the feedback from this first case study and following the same protocol and features, the tool was enhanced and further developed to be applied in the ecodesign of footwear. Four models of children shoes were analyzed and compared according to the FEcoDI obtained. The tool properly identified those proposals of design that should be rejected (mainly because of the likely damage to human health during use) and provided a ranking based on their more or less suitability from an environmental and safety point of view. This information was considered by the design team to incorporate the environmental perspective in their decisions.

7.1. Introduction

Ecodesign may be defined as the systematic introduction of environmental concerns during product design and development (AENOR, 2003). This means to bear in mind the environmental impacts at all stages of the product life cycle, starting at the designing and development phases. The objective is to create sustainable solutions that satisfy human needs and desires (Karlsson and Luttropp, 2006). The identification and appraisal of the environmental burdens requires the application of evaluation tools. The different available indicators offer complementary visions of the studied scenario; therefore, they cannot be replaced by each other and, in most cases, more than one should be applied at a time (Byggeth and Hochschorner, 2006).

The complementary characteristics of Ecological Footprint (EF), Life Cycle Assessment (LCA) and Environmental Risk Assessment (ERA) were discussed in Chapter 1 (section 1.4); particularly, the combination of EF and LCA to appraise the environmental performance of a production process was explored in Chapter 6. In this chapter, EF and LCA were applied to account for environmental burdens associated to the manufacture of the products; meanwhile, ERA was used to estimate the risk derived from the exposure to certain hazardous substances that raw materials may contain (organic compounds, heavy metals, etc.) and that would affect the final consumers of the product or factory employees (Franco et al., 2007a).

When more than one indicator is handled at a time, the difficulty arises when a decision has to be made based on the information provided by all of them. Methodologies of multi-criteria analysis have proved to be efficient in the definition of integrative frameworks, but their application requires processing imprecise, uncertain, qualitative or vague data (Lahdelma et al., 2000; Greening and Bernow, 2004). Fuzzy Logic (FL) is one of the most common methodologies used to address uncertainty matters (Bellman and Zadeh, 1970; Enea and Salemi, 2001; Ekel, 2002; Benetto et al., 2008). The use of FL techniques allows obtaining a quantitative approach using a qualitative representation (Zadeh, 1965); thus, it is able to simultaneously handle numerical data and linguistic knowledge (Carrasco et al., 2002).

FL techniques have been applied in a number of studies in the environmental field: to derive a Fuzzy Water Quality (FWQ) index (Ocampo-Duque et al., 2006);

to estimate aggregative risk of various environmental activities, pollution sources and routes in a given process (Sadiq and Hussain, 2005); to create a model called Sustainability Assessment by Fuzzy Evaluation (SAFE) in which ecological and human inputs were treated individually and then combined with the aid of fuzzy logic to provide an overall measure of sustainability (Phillis and Andriantiatsaholiniaina, 2001; Andriantiatsaholiniaina et al., 2004); to create a user-friendly software framework (F-IND) for the development of multivariable indices (Marchini et al. 2009); to derive an integrated fuzzy hazardous waste index (FHWI) as a measure of hazardousness of a given composite waste derived from the crisp inputs of its component's flammability, corrosivity, toxicity and reactivity attributes (Musee et al., 2006). The fuzzy multi-objective model proposed by Kuo et al. (2009) aimed at considering not only environmental criteria through a LCA but also the customer needs and cost considerations in the ecodesign of products. They developed an Eco-quality function deployment (Eco-QFD) to aid product design teams in seeking the overall customer satisfaction.

This chapter focuses on the ecodesign of products, being the first time ever that EF, LCA and ERA methodologies were integrated with this purpose on the basis of FL techniques. As case studies two kinds of products were analyzed: packaging materials and footwear. Packaging materials content substances that, under specific conditions, may be released to other compartments and, subsequently, reach the human feed chain. Monarca et al. (1994) analyzed PET bottles for carbonated beverages and identified some potentially genotoxic compounds (acetaldehyde, dimethyl terephthalate, terephthalic acid) among the migrant compounds. Also in a study considering PET as case study, Westerhoff et al. (2008) proved that small fractions of antimony migrated from bottles. Wagner and Oehlmann (2009) observed the contamination of mineral water with xenoestrogens that partly originated from compounds leaching from the plastic packaging material. In the study by Ahmad and Bajahlan (2007) it was proved that styrene and some other aromatic compounds leached continuously from polystyrene bottles. Meanwhile, Maia et al. (2009) demonstrated that dishwashing detergents may increase the bisphenol A (BPA) released from polycarbonate baby bottles. BPA is one of the chemicals that has generated more controversy and received major attention from authorities in the last years. Recently, the European Commission announced that from March 2011 the manufacture of baby feeding bottles containing endocrine disruptor BPA in the EU will be outlawed, and from June 2011 the importation and sale of such bottles will be prohibited. Therefore, the selection of the materials to be used in packaging of food or drinks should be particularly concerned with the potential hazard to human health, apart from considering other environmental impacts related to their production processes and recyclability.

The perception from consumers is of major relevance for the shoemaking industry. In this respect, Alcántara et al. (2005a; 2005b) scrutinized the semantic space of casual shoes regarding the customer's preferences. They found that the semantic space of casual footwear was described by 20 independent axes related to use and context of use, aesthetics, performance, quality, social context and gender, which formed a sound basis for emotional design and evaluation of shoes. However, the environmental criteria are also awaking the interest of the customers. Hence, different companies have presented initiatives on the development of environmentally friendly shoes, like the ECCO Group, Nike, Timberland or Camper. The main strategies followed were the use of non-hazardous and low energy content materials, keep to a minimum the use of harmful chemicals and the recycling of materials. In this respect, the footwear industry is responsible for a large waste stream at the end of the functional life of shoes, which are often disposed of in landfills. The potential reuse or recycling of these products is conditioned by the type of materials they are composed of, their diversity and feasibility to be separated, aspects that should be considered during the design stage (Staikos and Rahimifard, 2007a; 2007b).

As for other products, a European Eco-label exists for footwear (European Commission, 2009a) that provides guidance on the ecological goals that should be pursued, as recommended by the Integrated Product Policy -IPP- (European Commission, 2003). Besides, the content of certain hazardous substances is regulated: e.g. dimethyl fumarate (Spanish Government, 2009; European Commission, 2009b), phthalates (European Commission, 2005) or azo colorants (European Commission, 2004). Allowable thresholds are always more restrictive for children's products. Multinational corporations must bear in mind that legal restrictions vary among countries. Hence, the development of an *ad hoc* tool results appealing to simplify the process and to gather together all the relevant information for the particular case study.

In the present work, a framework based on the integration of EF, LCA and ERA was proposed. This approach was built on the basis of FL reasoning and features.

The objective was to obtain a final Fuzzy EcoDesign Index (FEcoDI) for ranking options from an environmental point of view. To test the tool, two packaging materials for a water bottle were evaluated. After refinement of the ecodesign tool on the basis of the feedback from this first case study and following the same protocol and features, the tool was enhanced and further developed to be applied in the ecodesign of footwear.

7.2. Methodology

The general protocol followed for the development of the ecodesign tool is presented in Figure 7.1. EF and LCA were used to analyze the environmental loads associated to the manufacture of the products; meanwhile, ERA was employed to assess the potential harm to human health derived from the use stage. The methodologies were implemented in a spreadsheet in MS Excel®. The indicators (I_n) obtained from the application of EF, LCA and ERA were integrated by means of a fuzzy inference engine, using the Fuzzy Logic Toolbox of Matlab®. As a result, an ecodesign indicator that aggregated the information from the different indicators was obtained.

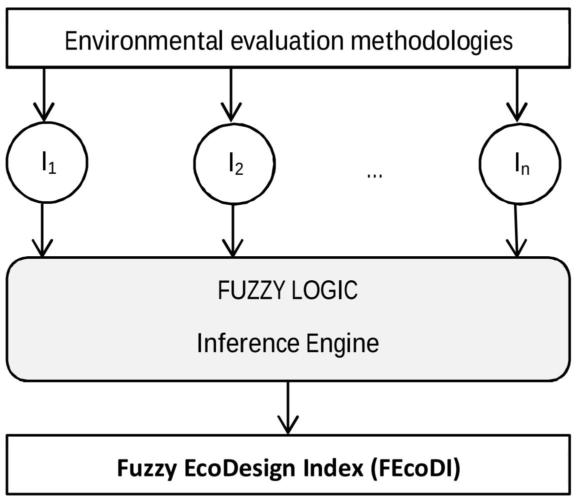

Figure 7.1. Protocol for the integration of environmental indicators.

The detailed description of the steps followed for the development of the tool and its application to the case studies is provided below. The section first deals with the environmental methodologies applied. Given the particularities of the products analyzed, it was preferred to present the methodologies separated by case study. Finally, the FL features were presented.

7.2.1. Case study 1: bottle of drinking water

The case study is based on a 2 liter bottle of drinking water as functional unit. Two kinds of plastic materials were assessed: PET and PVC. Based on the life cycle inventory of these products (Table 7.1), EF and complementary LCA were estimated. For ERA, particular acknowledged risk problems were taken into account, like the migration of bisphenol A (Le et al., 2008) and aldehydes in PET (Dabrowska et al., 2003), or vinyl chloride monomer in PVC (Fayad et al., 1997).

7.2.1.1. EF and LCA

For the EF appraisal the component approach based on life cycle inventory was employed (Monfreda et al., 2004). Thus, individual EFs were calculated for each material and energy flow in the inventory data, and then they were aggregated to estimate the total EF of each bottle. This indicator was employed to evaluate the energy and materials consumption, as well as the solid waste generation.

Emissions released during the manufacture were evaluated via LCA following the structured established by the ISO 14040 standards (ISO, 2006). During the life cycle impact assessment, only the compulsory characterization phase was considered in the development of the ecodesign framework. However, the normalization phase was also conducted to identify the most significant environmental impacts. Characterization and normalization factors from the Dutch Institute of Environmental Sciences (CML) were applied (Table 7.2).

7.2.1.2. ERA

For the environmental risk appraisal, migration rates from the bottle material to water, as well as final concentrations in water for the compounds considered (bisphenol A, vinyl chloride monomer and aldehydes), were found in the literature (Table 7.3).

Table 7.1. Inventory data for the PVC and PET systems considering a 2 liter bottle of drinking water as functional unit (Feijoo and Roca, 2005).

			PVC		PET	
INPUT VARIABLES	Raw Materials	Iron ore	0.0118	g	13.75	g
		Limestone	4.8	g	6.75	g
		Sand	0.032	g	0.5	g
		Water	640	g	438	g
		Bauxite	7.11	mg	7.75	mg
	Energy		1.978	MJ	1.868	MJ
OUTPUT VARIABLES	Air emissions	CO_2	57.6	g	53	g
		CH_4	0.182	g	0.0925	g
		N_2O	0.0002	g	0.0001	g
		NO_x	0.511	g	0.475	g
		SO_x	0.416	g	0.55	g
		Halon 1301	0.0012	mg	0.0018	mg
		Metals	0.0438	mg	0.0174	mg
		Aromatic compounds	1.487	mg	0.087	mg
		Others	0.0081	g	0.0028	g
	Water emissions	COD[(a)]	0.0352	g	0.0780	g
		Phosphates	0.0022	g	0.0022	g
		Nitrates	0.0003	g	0.0003	g
		Ammonium	0.0005	g	0.0008	g
		Others	0.0083	g	0.0027	g
	Solid waste		4.16	g	1.03	g

[(a)]COD: Chemical Oxygen Demand

Table 7.2. Characterization and normalization factors employed (CML, 2010).

	Pollutant	Characterization factors					
		GWP100 CO_2 eq	AP SO_2 eq	ODP CFC-11 eq	EP PO_4 eq	HTP 1,4-DB eq	POCP C_2H_4 eq
Air	CO_2	1					
	CH_4	25					
	N_2O	300					
	NO_x		0.5		0.13	1.2	
	SO_x		1.2			0.096	0.048
	Halon 1301	7100		12			
Water	COD				0.022		
	Phosphates				1		
	Nitrates				0.1		
	Ammonium				0.33		
Normalization factors EU25[(a)]		$5.02 \cdot 10^{12}$	$2.81 \cdot 10^{10}$	$8.94 \cdot 10^{7}$	$1.32 \cdot 10^{10}$	$7.78 \cdot 10^{12}$	$8.48 \cdot 10^{9}$

[(a)]Normalization factors calculated for EU25 based on Western Europe data and using GDP.

GWP100: Global Warming Potential 100 years
AP: Acidification Potential
ODP: Ozone layer Depletion Potential
EP: Eutrophication Potential
POCP: Photochemical Oxidation Potential
HTP: Human Toxicity Potential
CFC: chlorofluorocarbon
DB: Dichlorobenzene

Table 7.3. Data used for the ERA in the ecodesign of bottles.

Compound	**Material**	**Migration rate or concentration in water**			**RfD[(a)]** mg kg^{-1} day^{-1}	**SF[(a)]** kg day^{-1} mg^{-1}
		Value	Units	Source		
Bisphenol A	PET	0.19±0.13	µg l^{-1}	Le et al. 2008	$5.00 \cdot 10^{-2}$	-
Vinyl monomer	PVC	0.6	µg l^{-1}	Fayad et al. 1997	$3.00 \cdot 10^{-3}$	1.50
Acetaldehyde	PET	60.0±6.0	µg l^{-1}	Dabrowska et al. 2003	-	-
Formaldehyde	PET	78.1±7.8	µg l^{-1}	Dabrowska et al. 2003	$2.00 \cdot 10^{-1}$	-

[(a)]Source: ORNL, 2010.

The concentration of these compounds in water stored depended mainly on initial concentrations in the bottle material, as well as on temperature and time of storage. Estimations were made under the worst case scenario conditions (i.e. major concentrations reported) and only considering the oral pathway. Thus, during the exposition evaluation phase, equation [7.1] was used to estimate the daily dose due to ingestion of water:

$$Dose = WIF \cdot CW \qquad [7.1]$$

Where *Dose* is expressed in mg kg^{-1} day^{-1}, *WIF* is the human water intake factor -a value of $2.5 \cdot 10^{-2}$ l kg^{-1} day^{-1} was considered (Clark et al., 2003)- and *CW* is the final concentration in water of each compound expressed in mg l^{-1}.

For the risk characterization, reference doses (*RfD*) for non-carcinogenic effects, and slope factors (*SF*) for carcinogenic effects were used in order to calculate the Hazard Quotient (*HQ*) and the Cancer Risk factor (*CR*) as explained in Chapter 2 (ORNL, 2010; US EPA, 2010). *HQ* and *CR* values calculated for the different compounds associated to each material were added to obtain the final indicators.

7.2.2. Case study 2: footwear

Given the major concern on children's products, four models of size 20 shoes were evaluated: leather red, leather white, synthetic pink and synthetic white. The former two and the latter two were only differentiated by the color. The production process where these shoes were manufactured, located in Elche (Alicante, SE Spain), was studied (Figure 7.2). On the basis of a pair of shoes as functional unit, inventory data regarding the production process was provided by the factory. A wide number of operations are required for making a pair of shoes and they are generally performed by a separate machine.

After the design is finished, the first stage of the shoemaking process is the cutting of the pieces that will take form of uppers. This operation needs a high level of skill, especially when the material is leather, to minimize the generation of waste and to avoid the likely defects on the surface that cannot be part of the shoe. Therefore, it is hardly automatized; however, algorithms exist to deal with this nesting problem to minimize the trim loss (Yang and Lin, 2009). Next, the component pieces are sewn together to produce the completed upper. The completed uppers are molded into a shape of foot with the help of a last, a plastic shape that simulates the foot shape, which is later removed from the finished shoe to be used in making other shoes. The surplus of material is trimmed off the seam and the other parts of the shoes (toe cap, stiffener, insole, sole, etc.) are sequentially attached. After gluing steps, a thermal treatment takes places to assure a good adherence. Once the main structure of the shoe is ready, accessories (rings, eyelets, laces, rivets, etc.) are incorporated. At the finishing stage, shoes are cleaned; then, depending on the material and their final use, they could be stained, polished or waxed to ensure an attractive finish. After visual control, the footwear is packed.

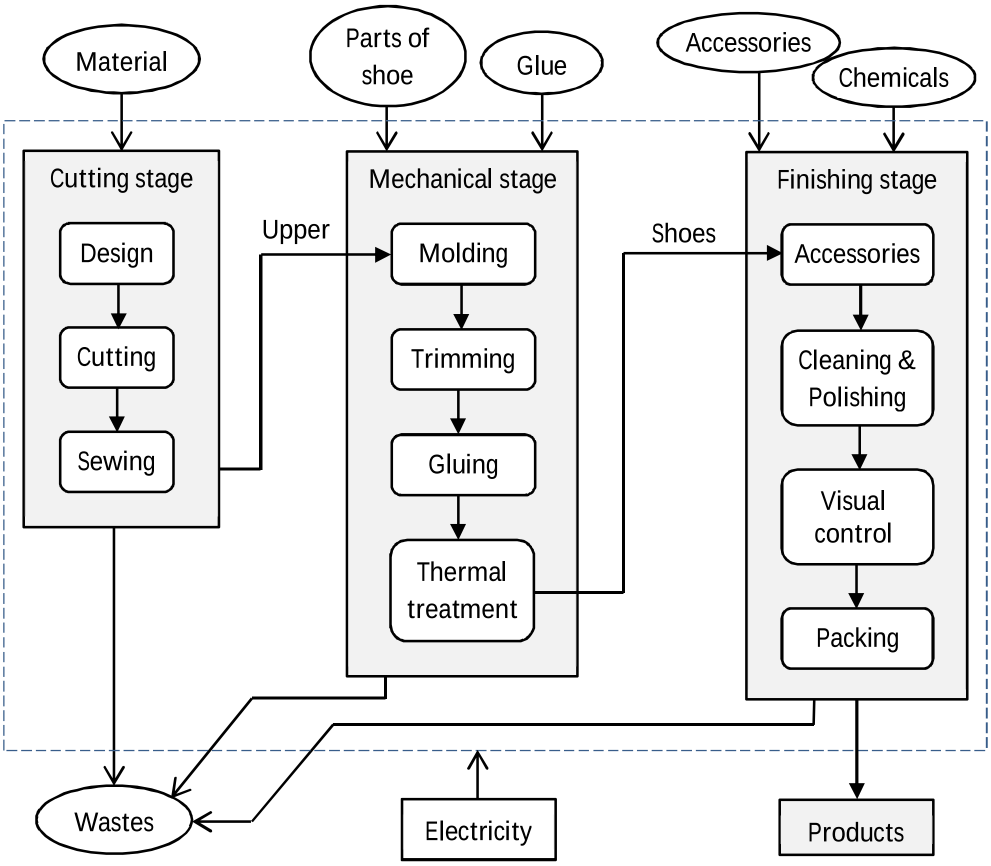

Figure 7.2. Flowsheet of the shoemaking factory.

7.2.2.1. Inventory data collection

Samples of the materials employed in the manufacture of the shoes, as well as a finished shoe, were provided, including a technical description (e.g. type of material, supplier, composition, color, etc.). Given the different nature of the elements which the shoe was composed of, the Input Module of the ecodesign tool was formatted accordingly. Further, their transformation into mass units was required for the application of the environmental evaluation tools. Given that samples of the entire sole were provided, they were weighted directly. For the main parts of the shoe, the shoemaking factory provided average consumption values of materials for a size 20 pair of shoes. These data were expressed in ft^2, units usually handled in the footwear industry. To convert these consumption values into mass units it was necessary to determine the surface density of the materials. Thus, the mass of samples with known dimensions was determined in the laboratory using a precision balance (AND model EK-1200G) and then the surface densities were estimated. For special elements like eyelets, rivets or rings, the mass of a single piece was measured and then multiplied by the number of pieces in the pair of shoes. Finally, the mass of laces was measured for a given length and then multiplied by the total length employed.

Electricity was the only source of energy used in the factory. The shoemaking process took place in two different buildings to which electricity was supplied by different companies. Therefore, both electricity mixes were included in the tool.

7.2.2.2. EF

As in the case of the bottles, the consumption of materials and energy was evaluated by means of the EF. Given that emissions were not released during the manufacture of shoes, the complementary use of LCA was not required in this case study.

7.2.2.3. ERA

The dermal route was considered to estimate the exposure to contaminants due to the use of footwear. Total doses were calculated under the approach of worst case scenario, i.e. considering direct contact skin-shoe (this may be more close to reality during warm seasons). Daily doses of exposure to contaminants via dermal absorption were determined according to equation [7.2] (US EPA, 2004).

$$Derms = CS \cdot \left(\frac{SAfeet}{BW}\right) \cdot AdhF \cdot CT \cdot DAF \cdot 0.1 \qquad [7.2]$$

Where $Derms$ (mg kg^{-1} day^{-1}) is the estimated daily dose of each contaminant due to dermal contact with shoes and CS is the concentration of the contaminant in shoes. $SAfeet$, BW and CT were defined in Table 7.4 and DAF is the dermal absorption factor (Table 7.5). $AdhF$ (mg cm^{-2}) is the adherence factor; in the particular case of shoes, feet were considered to be totally in contact with the components and the values of surface densities of materials were used as adherence factors. The factor 0.1 is the result of the homogenization of units to make the equation consistent. CS values were determined by an external laboratory.

Table 7.4. Parameters (age group 1-2 years) for the estimation of the daily dose of each contaminant due to dermal contact with shoes (US EPA, 2008).

Parameter		**Units**	**Value**
Mean surface area for feet	$SAfeet$	m^2	0.033
Mean body weight	BW	kg	11.4
Contact time	CT	h day^{-1}	8

For the risk characterization data from Table 7.5 were applied. Route-to-route extrapolations were employed in those compounds with unavailable specific dose-response assessment for the dermal route. Oral to dermal reference doses (*RfD*) and slope factors (*SF*) were extrapolated by multiplying and dividing the oral *RfD* and *SF* by the gastrointestinal absorption factor (*GIAB*), respectively (ORNL, 2010). The list of compounds initially included in the tool was established on the basis of the substances likely to be present in the shoes, but it could be extended if necessary.

$$DerRfD = OralRfD \cdot GIAB \qquad [7.3]$$

$$DerSF = OralSF / GIAB \qquad [7.4]$$

From this, the *HQ* and the *CR* for each compound were estimated by comparing the estimated doses with the reference values. They were later added to obtain the final indicators.

Table 7.5. Data used for the risk characterization in the ecodesign of shoes (ORNL, 2010).

Compound	***DAF***	***GIAB***	***OralRefD*** **(mg kg^{-1} day^{-1})**	***OralSF*** **(mg^{-1} kg day)**	**[b]*DerRefD*** **(mg kg^{-1} day^{-1})**	**[c]*DerSF*** **(mg^{-1} kg day)**
DEHP	$1.00 \cdot 10^{-1}$	1.00	$4.00 \cdot 10^{-2}$	$1.40 \cdot 10^{-2}$	$4.00 \cdot 10^{-2}$	$1.40 \cdot 10^{-2}$
BBP	$1.00 \cdot 10^{-1}$	1.00	$2.00 \cdot 10^{-2}$	$1.90 \cdot 10^{-3}$	$2.00 \cdot 10^{-2}$	$1.90 \cdot 10^{-3}$
DBP	$1.00 \cdot 10^{-1}$	1.00	$2.00 \cdot 10^{-1}$	-	$2.00 \cdot 10^{-1}$	-
DNOP	$1.00 \cdot 10^{-1}$	1.00	$1.00 \cdot 10^{-1}$	-	$1.00 \cdot 10^{-1}$	-
Formaldehyde	$1.00 \cdot 10^{-1}$	1.00	$2.00 \cdot 10^{-1}$	-	$2.00 \cdot 10^{-1}$	-
Cadmium	$1.00 \cdot 10^{-3}$	$2.50 \cdot 10^{-2}$	$1.00 \cdot 10^{-3}$	-	$2.50 \cdot 10^{-5}$	-
Chromium VI	$1.00 \cdot 10^{-3}$	$2.50 \cdot 10^{-2}$	$3.00 \cdot 10^{-3}$	-	$7.50 \cdot 10^{-5}$	-
Lead[a]	$1.00 \cdot 10^{-3}$	$1.50 \cdot 10^{-1}$	$3.60 \cdot 10^{-3}$	-	$5.40 \cdot 10^{-4}$	-

[a]Data for lead were extracted from WHO (2003) except for GIAB (ORNL, 2010)
[b]Estimated according to equation [7.3]. [c]Estimated according to equation [7.4].
DEHP: Di(2-ethylhexyl) phthalate BBP: Benzyl butyl phthalate
DBP: Dibutyl phthalate DNOP: Di-n-octyl phthalate

7.2.3. Fuzzy logic structure

The linguistic variables were those corresponding to the indicators derived from the application of the environmental evaluation methodologies explained previously: *EF*, impact categories from LCA, *HQ* and *CR*. LCA was applied only in the case of bottles and, given that the most relevant LCA impacts categories were GWP and AP, they were integrated into the *EF* figure as explained in previous chapters. Therefore, membership functions were defined for *CR*, *HQ* and *EF*. The *EF* contribution to the ecodesign indicator was measured in terms of *EF* variation (*ΔEF*) in relation to a base case (options that present an *EF* lower than the base case will be better valued than those with a higher *EF*).

Triangular and trapezoidal functions were selected in all cases. These straight line membership functions have the advantage of simplicity, but provide detail enough to describe the input variables considered in this case. The Mamdani inference system (Mamdani and Assilian, 1975) was selected and the Fuzzy Logic Toolbox of Matlab® v.7.7 was used to assist calculations (a file named ecodesigner.fis was created).

The main structure of the fuzzy inference engine was maintained from case study 1 to case study 2; however, due to the different characteristics of the products being analyzed and to the refinement of the tool, changes in membership functions were conducted and, consequently, in the decision tree and rules defined. Hence, the specific features for each case are presented separately.

7.2.3.1. Case study 1: bottle of drinking water

Membership functions defined for the input variables are shown in Figures 7.3 to 7.5, where μ represents the membership degree. A major division of the universe of discourse of *CR* was considered given the major relevance and incidence of this indicator in the safe use of products. For *HQ* only three categories were created, assuming that values under 0.5 would not be of concern. Finally, a maximum variation of ±20% for *EF* was considered.

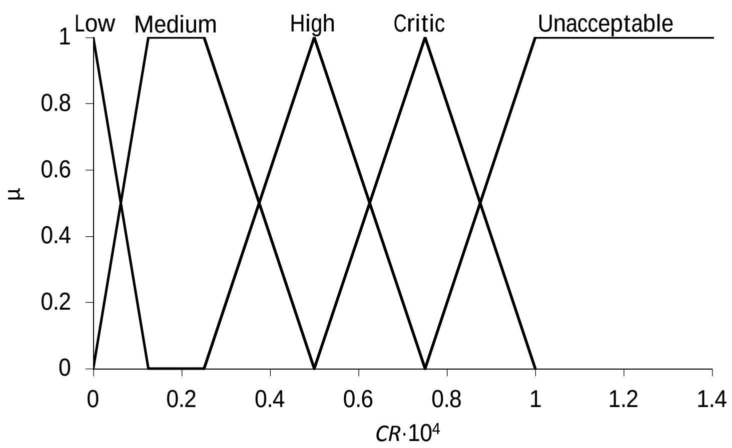

Figure 7.3. Membership function and universe of discourse for input variable *CR*, in case study 1.

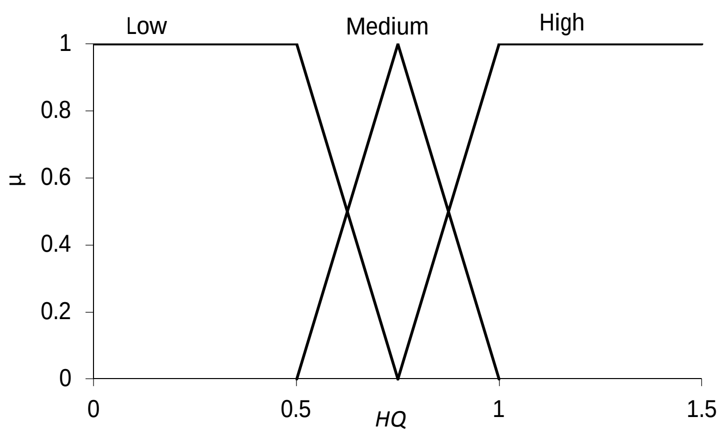

Figure 7.4. Membership function and universe of discourse for input variable *HQ* in case study 1.

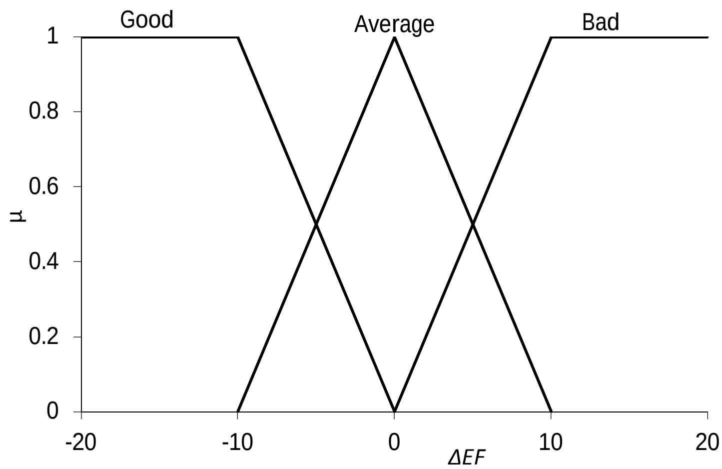

Figure 7.5. Membership function and universe of discourse for input variable *ΔEF* in case study 1.

The output to the inference engine was the Fuzzy EcoDesign Index (FEcoDI) ranging from 0 to 100. According to the definition conveyed in Figure 7.6, the higher the FEcoDI the better the option from an environmental point of view.

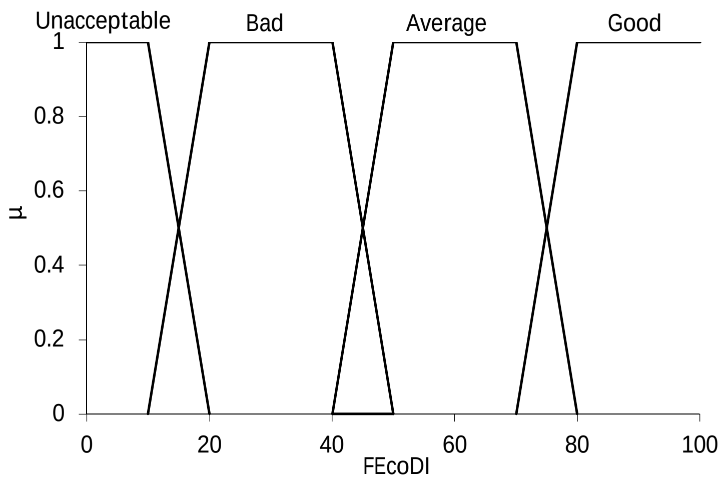

Figure 7.6. Membership function and universe of discourse for FEcoDI in case study 1.

7.2.3.2. Case study 2: footwear

In this case, a major division of the universe of discourse for *HQ* was required to make the tool more sensitive to the presence of chemicals (Figure 7.7). Moreover, given that only one route of exposure (dermal) was considered and that, therefore, the total hazard index could be increased, a more precautionary approach was adopted and more significance was provided to *HQ* values. Besides, the universe of discourse for *ΔEF* was extended since a major variability in *EF* values may occur for different models of shoes; as a consequence, more categories were defined for this variable (Figure 7.8). To cover the extended variability in the categories of the linguistic variables, the membership functions for the FEcoDI were also modified (Figure 7.9). The functions for *CR* were maintained (Figure 7.3).

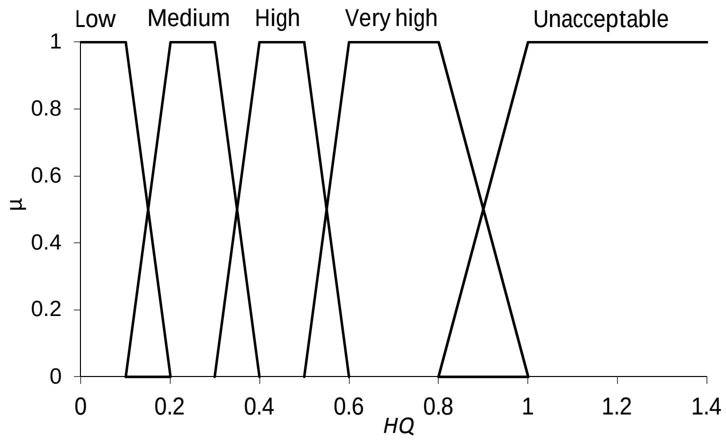

Figure 7.7. Membership function and universe of discourse for input variable *HQ* in case study 2.

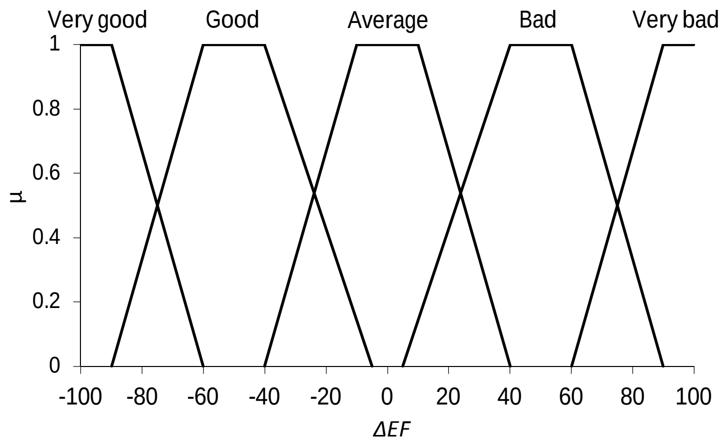

Figure 7.8. Membership function and universe of discourse for input variable *ΔEF* in case study 2.

The refinement of membership functions in study case 2 lead to the modification of the decision tree. Also, the number of if-then rules increased from 23 to 45. Some of these rules are shown in Table 7.6 as an example, and a branch of the decision tree in case study 2 is presented in Figure 7.10.

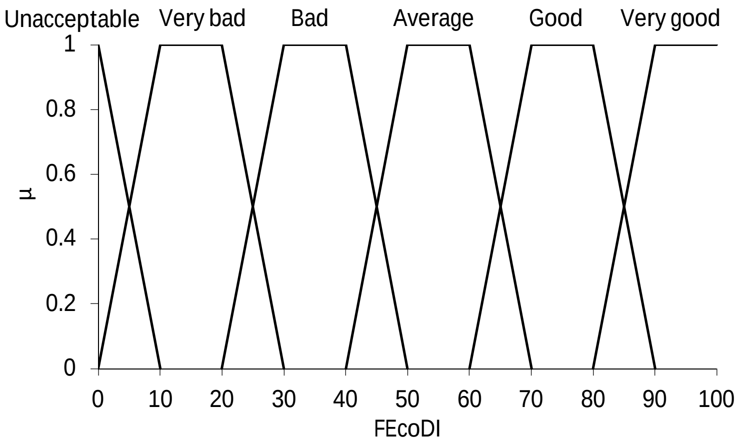

Figure 7.9. Membership function and universe of discourse for FEcoDI in case study 2.

Table 7.6. Examples of the if-then rules defined in case study 2.

IF	**THEN**
CR *is* Unacceptable	FEcoDI *is* Unacceptable
CR *is* High *and* HQ *is* High	FEcoDI *is* Very bad
CR *is* Low *and* HQ *is* High *and* ΔEF *is* Average *or* Bad	FEcoDI *is* Very bad
CR *is* Low *and* HQ *is* Low *and* ΔEF *is* Bad	FEcoDI *is* Average
CR *is* Low *and* HQ *is* Medium *and* ΔEF *is* Good	FEcoDI *is* Good
CR *is* Low *and* HQ *is* Low *and* ΔEF *is* Very good	FEcoDI *is* Very good

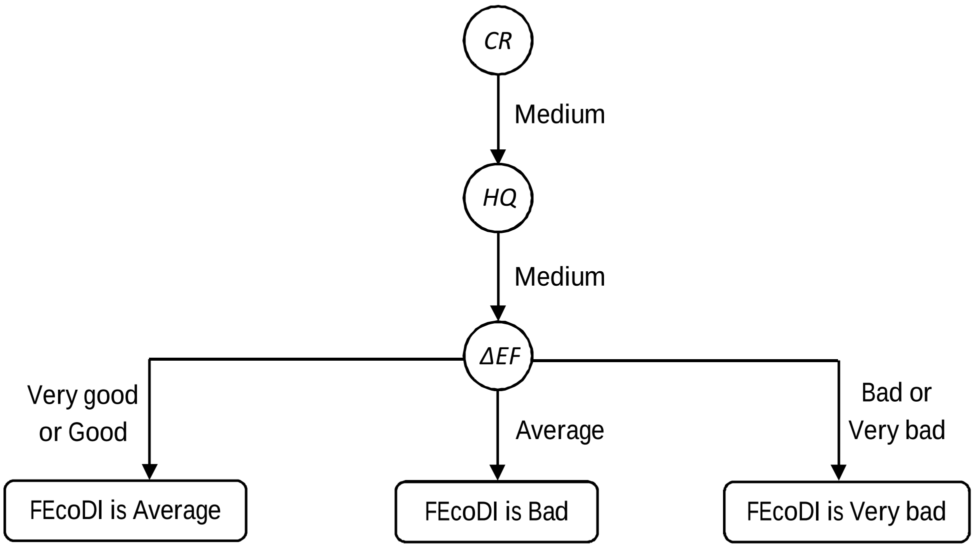

Figure 7.10. Branch of the decision tree of the fuzzy inference engine refined in case study 2.

7.3. Results and discussion

The general structure of the developed ecodesign tool is shown in Figure 7.11 and the particular results obtained for each case study are presented in sections 3.1 and 3.2.

7.3.1. Case study 1: bottle of drinking water

On the basis of the inventoried data for the manufacture of a 2 liter bottle of these materials, the EF and ACV were calculated, the former accounting for materials and energy consumption, and the latter for emissions to air and water. LCA results for the characterization and normalization phase are shown in Table 7.7 and Figure 7.12, respectively.

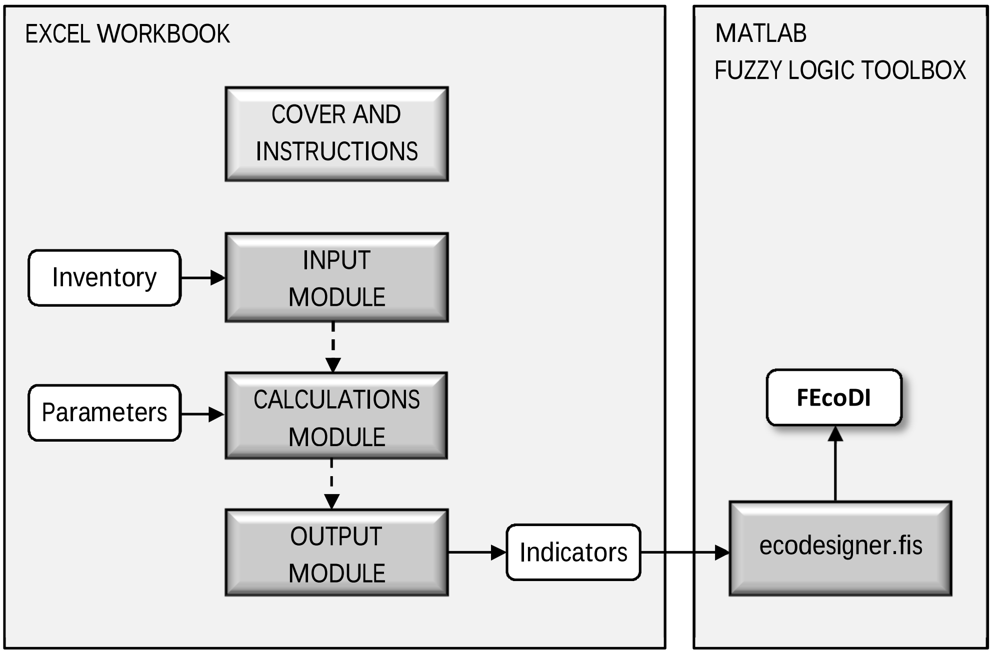

Figure 7.11. Simplified flowchart of the tool developed for the ecodesign of products.

Table 7.7. Characterization stage for PVC and PET bottle.

Impact category	Units	PVC	PET
GWP100	g CO_2	62.22	55.36
AP	g SO_2	0.75	0.90
ODP	g CFC-11	$1.44 \cdot 10^{-5}$	$2.16 \cdot 10^{-5}$
EP	g PO_4	$6.96 \cdot 10^{-2}$	$6.60 \cdot 10^{-2}$
HTP	g 1,4-DB	0.65	0.62
POCP	g C_2H_4	$2.00 \cdot 10^{-2}$	$2.64 \cdot 10^{-2}$

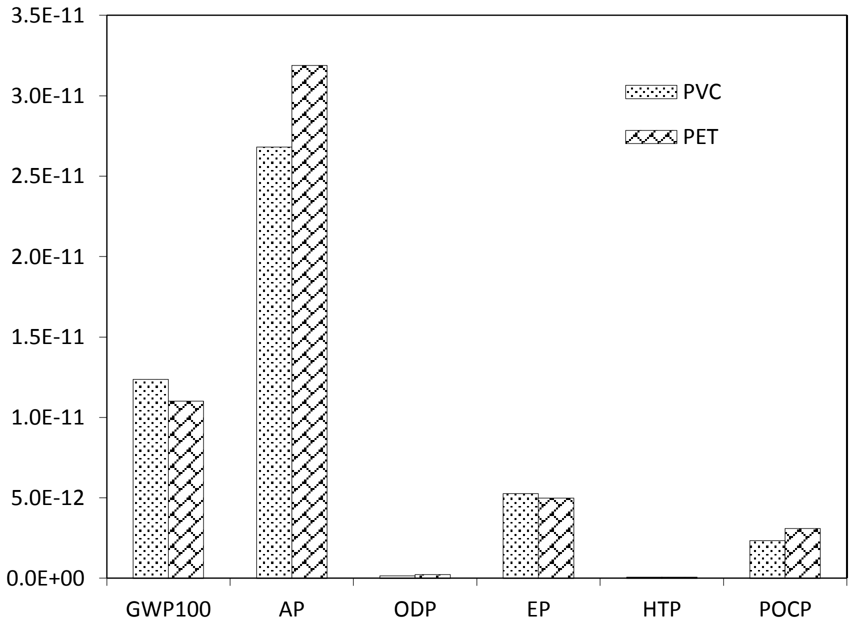

Figure 7.12. Normalization stage for PVC and PET bottle.

Observing the major importance of GWP100 and AP categories, and the low relevance of ODP and EP categories, the latter were discarded from this first approach of the ecodesign tool for simplicity. After the integration of the former two into the EF estimates using absorption factors, the total amount of productive land required resulted in 2.27 m^2/bottle for PVC and 2.43 m^2/bottle for PET. Since in the fuzzy structure EF is measured in terms of variance with respect to a base case, the lowest value of the two (PVC) was considered as reference level. Consequently, the input for the PET bottle would correspond to a 7% increase in relation to the EF of the PVC bottle.

Results from ERA were collected in Table 7.8. Acetaldehyde and formaldehyde have the R40 risk phrase associated (ESIS, 2010), this meaning that there is limited evidence of a carcinogenic effect (they are catalogued as carcinogenic type 3). In addition, slope factors exist for both compounds for the inhalation route (ORNL, 2010). However, there is currently no slope factor available for the dermal route recognized by an international organization for these substances and, therefore, no contribution of these substances to cancer risk was considered. Furthermore, acetaldehyde did not have chronic *RfDs* either; thus, it was not possible to assess acetaldehyde did not have chronic *RfDs* either; thus, it was not possible to assess the potential environmental risk contribution of this compound. The *HQ* and *CR* were below the safety limits stated by the US EPA in all cases. It must be noticed that for PET the risk contribution of two compounds was considered, while for the PVC only the potential risk derived from the vinyl monomer migration was appraised. This was made according to main risk problems acknowledged in the literature indicated in the methodology section, and taking into account that the purpose of the analysis was to test the tool, not to present definite results on the evaluation of the two materials.

Table 7.8. Results from the ERA for the two bottles analyzed.

Material	**Compound**	***HQ***	***CR***
PET	Bisphenol A	$9.50 \cdot 10^{-5}$	-
	Formaldehyde	$9.76 \cdot 10^{-3}$	-
	Total	$9.86 \cdot 10^{-3}$	-
PVC	Vinyl monomer	$5.00 \cdot 10^{-3}$	$2.25 \cdot 10^{-5}$

The results obtained applying each environmental evaluation methodology were introduced into the ecodesign tool. As a result, a FEcoDI of 30.0 for the PVC bottle was obtained, while a value of 66.6 was estimated for the PET bottle. Consequently, this would lead to select the PET bottle as the best option from an environmental point of view.

The main difference between the two materials studied was the existence of a carcinogenic slope factor for the vinyl monomer. In the definition of membership functions, the universe of discourse for *CR* was divided into more categories than for the other variables. The *CR* has a probabilistic nature; this means that if it has a value different from zero, a risk of someone suffering from cancer will exist. However, in the case of *HQ*, effects on exposed population will occur only if the reference dose is exceeded. As a result, fewer precautions were taken for *HQ* values lower than 1, while for the *CR* a more strict characterization was considered. The tool seemed to be sensitive to changes in the *EF* only when *CR* and *HQ* were low enough. This may be because of the way the decision tree was constructed, first evaluating the *CR* and then the *HQ*, and only allowing the *EF* appraisal of those products that had passed the first barriers. Therefore, materials that may cause carcinogenic effects will receive a bad evaluation from the tool. Thus, in this case, the PET bottle obtained a better FEcoDI in spite of having a higher *EF* and *HQ*. The revision of this imbalanced weight for the input variables may lead to variations in the structure of the decision tree in the fuzzy reasoning. It could be more adequate to expand the number of levels for the *HQ* membership function in order to properly differentiate between products with different risk characteristics under the safety limits (*HQ* = 1), depending on the case study. These considerations were taken into account during the refinement of the tool for its application in the ecodesign of footwear.

7.3.2. Case study 2: footwear

The inventory (based on a functional unit of a pair of shoes) regarding the consumption of materials for the different models of shoes analyzed was collected in Tables 7.9 and 7.10. An average consumption of 0.75 kWh of electricity was allocated for a pair of shoes manufactured.

Table 7.9. Inventory data to estimate the EF of the synthetic model.

Element	**Material**	**Consumption (ft^2/pair)**	**Surface density (g/cm^2)[(a)]**	**Consumption (g/pair)**
Sole	Rubber			[(b)]29.5208
Upper	Polyurethane	0.70	0.0379	24.6472
Insole	Cotton & polyester	0.18	0.0193	3.2274
Lining A	Leather	0.74	0.0333	22.8932
Lining B	Polyester	0.74	0.0181	12.4434
Velcro	Nylon & polyester	0.02	0.0745	1.3142
Element	**Material**	**Consumption (no. pieces)**	**Mass (g/piece)**	**Consumption (g/pair)**
Rings	Zinc	1	0.4183	0.4183

[(a)]Experimentally determined. [(b)]Directly measured.

Table 7.10. Inventory data to estimate the EF of the leather model.

Element	Material	Consumption (ft^2/pair)	Surface density (g/cm^2)[(a)]	Consumption (g/pair)
Sole	Rubber			[(b)]29.5208
Upper	Leather	0.76	0.0714	50.4129
Insole	Cotton	0.18	0.0339	5.6723
Lining A	Leather	0.12	0.0365	4.0692
Lining B	Polyamide	0.12	0.0115	1.2821
Velcro	Nylon & polyester	0.02	0.0692	1.2199
Element	**Material**	**Consumption (no. pieces)**	**Mass (g/piece)**	**Consumption (g/pair)**
Rings	Zinc	1	0.4183	0.4183
Eyelets	Nickel	12	0.1568	1.8816
Rivets	Nickel	4	0.1192	0.4768
Element	**Material**	**Consumption (cm)**	**Mass (g/cm)**	**Consumption (g/pair)**
Lace	Viscose & polyurethane	40	0.0286	1.144

[(a)]Experimentally determined. [(b)]Directly measured.

The results from the external laboratory regarding the concentration of hazardous compounds in shoe samples were collected in Table 7.11, indicating the model of shoe, the part of shoe, the substance that was detected and the concentration. These values were used to determine the exposure to each contaminant due to dermal contact.

The EF assessment yielded the following results: 6.5 gm^2/pair of shoes for the synthetic model and 11.1 gm^2/pair of shoes for the leather model. The former presented the lowest value and, therefore, it was taken as base case (final values for the indicator entering the fuzzy toolbox are indicated in Table 7.13). Meanwhile, results from ERA are presented in Table 7.12.

Table 7.11. Inventory data for the ERA of the models of shoes studied.

Model	**Part**	**Substance**	**ppm**
Pink synthetic	Insole	Phthalates	1300
	Lining	Formaldehyde	25
	Lining	Chromium	9.82
White synthetic	Lining	Formaldehyde	131
	Insole	Formaldehyde	92
White leather	Lining	Formaldehyde	22

Table 7.12. Results from ERA for the four models of shoes analyzed.

Model	**Piece**	**Substance**	***HQ***	***CR***
Pink synthetic	Insole	Phthalates	29.84	$8.35 \cdot 10^{-3}$
	Lining	Formaldehyde	0.06	-
	Lining	Chromium	1.04	-
		Total	30.94	$8.35 \cdot 10^{-3}$
White synthetic	Lining	Formaldehyde	0.34	-
	Insole	Formaldehyde	0.23	-
		Total	0.57	-
White leather	Lining	Formaldehyde	0.06	-

The summary of input variables to the fuzzy inference engine is presented in Table 7.13, together with the FEcoDI appraised for each model. None of the models analyzed obtained a good evaluation from the ecodesign tool. The pink synthetic model clearly fell inside the unacceptable region. This was due to the presence of high levels of phthalates in the insole (specifically in the printing) that made the *CR* and *HQ* exceed the tolerable thresholds. It must be remarked that, given that the kind of phthalate was not specified, it was assumed to be DEHP under a worst case scenario approach (this compound presented the lowest *RfD* and the highest *SF*). Phthalate esters are predominantly used as plasticizers due to the flexibility and softness they add to PVC resins (Franco et al., 2007b). They have a high potential to diffuse out of plastic materials since they are not covalently bound to the polymeric matrix, what may be of major concern in children's products because of mouthing (oral pathway of exposure to contaminants). Therefore, the presence of this kind of substances, especially DINP (Di-isononyl phthalate), in toys has been the subject of criticism and different studies have dealt with the assessment of the risk associated (Wilkinson and Lamb, 1999; Babich et al., 2004). Although these studies concluded that oral exposure to DINP from mouthing soft plastic toys is not likely to present a health hazard to children, it is also important to highlight that the total hazard index is the result of the contribution from different pathways to which human beings may be exposed in daily activities. The European Directive 2005/84/EC (European Commission, 2005) stated that the precautionary principle should be applied and established the introduction of restrictions of phthalates for toys and childcare articles, being more severe for DEHP, DBP and BBP, which had been identified as repro-toxic substances. Thus, the presence of phthalates in children's footwear (which apart from the dermal contact could also be put in the mouth) should be avoided.

The white synthetic model was evaluated as very bad. The reason in this case was the *HQ* associated to the formaldehyde present in the shoe lining and in the insole. Finally, the tool could not distinguish between the red leather and the white leather models, since the *HQ* for the latter was low enough. The absence of hazardous substances meant that a better FEcoDI was obtained, in spite of having a higher EF than the synthetic models. Nonetheless, the major area requirement prevents this option from reaching a better FEcoDI.

Table 7.13. Input variables to the fuzzy toolbox and FEcoDI obtained.

Model	ΔEF	HQ	CR	FEcoDI
Pink synthetic	0	30.5	$8.35 \cdot 10^{-3}$	3
White synthetic	0	0.51	-	15
Red leather	70.8	-	-	47.5
White leather	70.8	0.06	-	47.5

The results obtained from the ecodesign tool would lead to immediately discard the pink and white synthetic models. Regarding the leather options, they could be taken into account but new proposals that meant a reduction of the EF should be sought to achieve a better environmental evaluation (higher FEcoDI). Besides, once more models were evaluated using the developed tool, the establishment of the base case to assess the *ΔEF* could be redefined and better founded based on the range of EF values observed. Hence, the applicability of the tool would benefit from the feedback of its users.

Cherrett et al. (2005) studied five fiber types, namely conventional cotton, organic cotton, conventional hemp, organic hemp and polyester, and ranked them with regard to the EF (gha) of producing one ton of spun fiber. The lowest EF figure was 1.5 gha t^{-1} for organic hemp. Polyester produced in Europe presented an EF of 1.67 gha t^{-1}, in spite of its higher energy requirements. The difference was that polyester did not require the land area for cultivation that cotton and hemp did. Actually, crop cultivation represented the greatest proportion of the EF in the cotton case studies, which ranged from 2.17 gha for organic cotton to 3.57 gha for conventional cotton. Similarly, the main component of the EF of leather was the pasture land for cattle raising; nonetheless, the raw material in the production of leather is a by-product of the mead industry, and allocation of burdens should be considered (Joseph and Nithya, 2009). The consequence was that the synthetic models analyzed obtained a better evaluation from the EF perspective. Therefore, the results provided by EF assessments must be read carefully and with a deep understanding of their meaning so as not to lead to erroneous interpretations. Also, the perspective from other indicators must also be considered during the ecodesign stage, as proposed in this chapter, to draw a more complete map of the environmental implications of making one or another choice. Thus, using leather for which phosphonium instead of chromium was used as tanning agent would make a difference. The presence of chromium would not only pose a hazard during the using stage but also at the end-of life if shoes are disposed of in landfill, because of the likely spontaneous oxidization of Cr^{3+} to Cr^{6+} in the open-air dumps (Kolomaznik et al., 2008). Hence, although the entire cradle to grave cycle was not analyzed (Milà et al., 1998), the effects of the methods employed during the processing of materials will be transferred to any extent to the final product in terms of the presence of hazardous substances, thus being considered during the ERA.

Selecting fibers produced under sustainable criteria and reducing the materials consumption as much as possible, would lead to the better values of FEcoDI, since the area requirements would be minimized and the absence of hazardous compounds would ensure safety conditions during the use stage. Another option would be to use recycled materials, like rubber from tires in soles.

By incorporating the perspective of EF and ERA in the ecodesign tool, many of the aspects also evaluated in the European Ecolabel are considered (European Commission, 2009a). Thus, the use of sustainable materials, their durability or recyclability is measured by means of the EF. Also, the effect of energy consumption is included in the EF appraisal. Besides, the limitation of substances harmful for health and the environment (e.g., chromium) is established by the *HQ* and *CR* maximum allowable thresholds.

The tool is flexible to the introduction of more indicators that could be identified as relevant after its applications to a number of cases. Further, other criteria to be considered during the design stage, e.g. economic or fashion tendencies, could also be integrated into the final ecodesign index. In this respect, FL techniques were successfully applied in the apparel industry in the work by Wong et al. (2009), in which an expert system was developed to provide customers with professional and systematic mix-and-match recommendations based on attribute data such as color, pattern or type.

7.4. Conclusions

A tool based on EF, LCA and ERA was built on the basis of FL reasoning and features. The output obtained was a Fuzzy EcoDesign Index (FEcoDI) that could range between 0 and 100, and that collected the criteria offered by the different environmental evaluation methods. This allowed for the conversion of the expert knowledge of technicians, not familiar with the theoretical fundamentals of FL, ERA, LCA and EF, into an ecodesign index in a simple way.

In general terms, the constructed tool seemed to work properly. However, when testing it with the case study of bottles some limitations were detected. Thus, the tool was not sensitive enough to different concentrations of hazardous substances in terms of the *HQ*. As a consequence, during the refinement of the tool to be applied in the ecodesign of shoes, a major division of the universe of discourse of *HQ* was carried out. The same had to be done for the EF, given that more variability in materials consumption (amount, composition, etc.) could be expected and, therefore, the *EF* was allowed to range in a ±100% interval. This meant that more categories had to be defined for the linguistic variable *ΔEF*. The major division in input variables required a correspondent division in the output FEcoDI to properly cross categories and to redefine the rules. A new decision tree was constructed accordingly.

Future revisions of the tool should deal with the integration of more LCA impact categories that may be significant in the evaluation of other products. Similarly, energy flows could be introduced as individual indicators (or using the Carbon Footprint as indirect measure) to better reflect the importance of this indicator, helping to better identify materials with less embodied energy. Another option would be to separate the Carbon Footprint (CF) from the EF (it is currently integrated in the aggregated EF figure) and to use it as indirect measure of energy consumption.

Another proposal to improve the tool that will be considered in future research is the use of continuous functions (e.g. Gaussian type) instead of triangular or trapezoidal types to define the membership degree. The former would continuously range between 0 and 1, and their relative contribution to the output FEcoDI would be established by means of the definition of weights. The main benefit of this option would be that more input variables could be added to the inference engine without requiring the redefinition of a decision tree and the associated decision rules, thus making the tool more flexible.

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Chapter 8

MSW treatment options: environmental assessment and MCA

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Abstract

The choice of a municipal solid waste (MSW) treatment option is a complex process in which a widespread set of criteria must be taken into account. Additionally to economic, geographical situation or social aspects, the decision process should consider the environmental perspective. With the purpose of quantifying these environmental burdens, a wide variety of environmental and sustainability indicators have been developed in the last years. However, integrative frameworks have been signaled as the best option to achieve more comprehensive assessments. To this respect, Multi Criteria Analysis (MCA) provides a family of flexible analytical tools that can effectively support decision making with regard to complex sustainability issues.

In the first part of this chapter, a case study extracted from the literature was analyzed to prioritize among four different options of (MSW) treatment processes. The Ecological Footprint (EF) was applied as single indicator; then more indicators were included and a ranking of alternatives was established using MCA methodologies (AHP, ELECTREE and PROMETHEE). The ranking was (from best to worst): thermal plasma gasification, biological treatment of organic fraction with energy recovery from refuse derived fuel, incineration with energy recovery and landfilling. This was in agreement with the commonly recommended hierarchy.

In the second part of the chapter, the real case of LIPOR (Porto, Portugal) was assessed. A joint application of EF and Life Cycle Assessment (LCA) was proposed with this purpose. The main activities of the integrated management system of LIPOR, namely multi-material valorization, organic valorization, energy valorization and landfilling, were analyzed; consequently, stages that presented a major contribution to environmental burdens could be identified.

8.1. Introduction

Economic and human population growths, as well as changes in lifestyles and in consumption patterns, have been the main drivers of the progressive increase in wastes generation (particularly in packages, which is one of the major problems). One of the major challenges for municipalities in the 21st century is to collect, recycle, treat and dispose of these increasing quantities of solid waste (Cherubini et al., 2009). The sustainable management of municipal solid waste (MSW) has become necessary at all phases of impact from planning to design, to operation and to decommissioning (Pires et al., 2011). Waste causes a number of impacts on the environment, including pollution of air, soil, surface and ground water; meanwhile, valuable space is taken up by landfills and poor waste management causes risks to public health (Daskalopoulos et al., 1997; EEA, 2007). This is conventionally the catalyst to handle the problem; otherwise, waste is treated as irrelevant to production (Seadon, 2010).

Landfill is still the most common waste management method used across the pan-European region. The EU directives and national policies developed since the beginning of the 1990s set targets for recycling and recovery and restrictions on waste to landfill. As a result, the percentage of municipal waste recycled (including composting) has increased significantly. In EU‑15 + EFTA (European Free Trade Association), the percentage of recycling reached 40 % in 2004. In EU‑10, however, recycling and incineration are minimal (EEA, 2007). The increasing pressure on waste managers, planners and waste regulators to deliver a sustainable approach has spanned the spectrum of new and existing waste treatment technologies and managerial strategies from maintaining environmental quality at present to meet sustainability goals in the future (Barton et al., 1996; Pires et al., 2011).

The waste hierarchy defined in the Directive 2008/98/EC on waste establishes the following priority order to be considered in waste prevention and management legislation and policy: 1) prevention; 2) preparing for re-use; 3) recycling; 4) other recovery, e.g. energy recovery; and 4) disposal. Nonetheless, this new Directive also addresses the possibility of altering the stated hierarchy in a specific situation, if justified by a life-cycle thinking study (European Commission, 2008; Tarantini et al., 2009). Thus, this kind of studies can be used to test the waste hierarchy and identify situations where it may be modified, as for exchanging order between recycling and incineration, or to place biological treatments such as anaerobic digestion and composting (Moberg et al., 2005; Finnveden et al., 2005). This may depend on the waste itself, on the location where the waste arises and its timing, as well as priorities in cases of conflicting results. Alternatives should be examined systematically so that waste is put to the use which is most beneficial in resource and environmental terms, rather than accepting a simple hierarchy, thus pursuing integrative strategies (Clift et al., 2000; Cherubini et al., 2009; Pires et al., 2011).

Tools are needed to predict the likely overall environmental burdens of any waste management system (Thomas and McDougall, 2005). Given its broad approach, LCA has widely been applied all over the world in the MSW field: Arena et al. (2003), Eriksson et al. (2005), Moberg et al. (2005), Bovea and Powell (2006), Hong et al. (2006), Özeler et al. (2006), Cherubini et al. (2009), Banar et al. (2009), Khoo (2009), Rigamonti et al. (2009), Zhao et al. (2009). Cleary (2009) conducted a comparative analysis of 20 process-based LCAs of MSW published between 2002 and 2008 (include some of the previously mentioned studies). In this review, system boundaries, data sources and impact assessment methods were compared to appraise the transparency of LCA studies. Although differences were appreciated on assumptions made by practitioners, it did not seem to affect the order of preferred options for MSW treatment. It was also observed that the human and ecological toxicity impact categories were much less common than global warming potential, acidification, and eutrophication. The applicability of LCA for waste management presents some other limitations. The dependence on a certain quantity of waste treated makes it inadequate for the identification and assessment of waste prevention strategies. Further, the spatial information is lost by summarizing emissions and the long-term effects are not evaluated. Besides, the use of average data may be misleading (Ekwall et al., 2007).

LCA is not the only methodology available to appraise environmental burdens associated to MSW treatment processes. Cherubini et al. (2009), for instance, included the complementary perspectives of material flows and ecological footprint, also based in a life-cycle inventory. Further, in other cases it may be of interest to consider other criteria like rates of energy and material recovery (Karagiannidis and Perkoulidis, 2009; Ekmekçioğluet al., 2010) or the distance to protected areas and nearby towns (Aragonés-Beltrán et al., 2010). However, considering a number of criteria at a time certainly complicates the decision making process. In this respect, Multi Criteria Analysis (MCA) provides a family of flexible analytical tools that can effectively support decision making with regard to complex policy and planning problems (Ladehlma et al., 2000; Greening and Bernow, 2004).

Several examples of application of MCA techniques related to waste treatment can be found in the recent literature. One of the most frequently applied MCA method is ELECTRE III. Franca Norese (2006) used it to consider environmental, social and technical criteria to select the adequate localization of an incinerator and a waste disposal plant in Turin (Italy); similarly, Banias et al. (2010) employed ELECTRE III to define a MSW management strategy in the region of Central Macedonia (Greece) considering 19 criteria including environmental, economic and social aspects; Bolinger and Pictet (2008) also used this method to include inputs from policy and technology on the problem of waste incineration. Regarding specific waste types, Karagiannidis and Perkoulidis (2009) applied ELECTRE III to rank different technologies for the anaerobic digestion for energy recovery of the organic fraction of municipal solid wastes; Roussat et al. (2009) applied ELECTRE III to select among 9 alternatives for sustainable demolition waste management using 8 criteria from the three dimensions of sustainability; meanwhile, Archillas et al. (2010) used this method in the development of a methodology for the optimal location of units of treatment and recycling of WEEE (Waste of Electrical and Electronic Equipment). Vego et al. (2008) applied another outranking method, PROMETHEE combined with the GAIA plane, to define a MSW management strategy in Croatia. Other authors combined GIS (Geographical Information Systems) and MCA techniques to deal with the problem of landfill site selection: Guiqin et al. (2009) used AHP (Analytic Hierarchy Process) to establish the criteria weight, while Geneletti (2010) used a linear weight scoring. Kijak and Moy (2004) applied MAUT (Multi Attribute Utility Theory) to develop a decision support framework for the evaluation of scenarios for the integrated management of MSW, combining LCA with other environmental, social and economic tools. Aragonés-Beltrán et al. (2010) used ANP (Analytic Network Process), based on AHP features, to select the best location for the construction of a MSW treatment plant in Valencia. They classified criteria in 4 main clusters in the hierarchy: plant exploitation costs, facilities and infrastructures, environmental issues and legal requirements. In Turkey, Ekmekçioğluet al. (2010) employed fuzzy AHP to determine criteria weight and then applied TOPSIS

(Technique for Order Preference by Similarity to Ideal Solution) for the selection of the best alternative for MSW disposal (i.e. landfilling, composting, conventional incineration, RDF – Refuse Derived Fuel - combustion). There are other examples in which less common MCA methods were applied or self-developed models were developed for decision making and planning of waste management strategies (Lahdelma et al., 2002; Vaillancourt and Waaub, 2002; Simões Gomes et al., 2008). In general, MCA methods proved to be a practical and feasible method for the integrated assessment and ranking of alternatives. A wide review on models to support decision making in the area of MSW management was conducted by Morrisey and Browne (2004). They classified these models into three main categories (based on cost benefit analysis, based on LCA and based on MCA) and highlighted some of their major shortcomings and benefits. Regarding the MCA type, ELECTRE III was signaled as the most suitable given its superior features when compared with others (e.g. AHP which is also frequently used in the waste management field).

This chapter focuses on the MSW treatment and management problem from an environmental point of view. In a first part, four different options of MSW treatment were ranked using data from the literature. First, the Ecological Footprint (EF) was used as single indicator; then, other indicators were included in the appraisal and were integrated using MCA methods. In a second part, a real case scenario corresponding to LIPOR, the waste management service in Oporto (Portugal), was assessed.

8.2. Methodology

The environmental evaluation methodologies applied in this chapter were: EF, Life Cycle Assessment (LCA) and Material Flows Analysis (MFA). The MCA methods were AHP, ELECTREE I, ELECTREE III and PROMETHEE/GAIA. These all were explained in Chapter 2.

8.2.1. Case study 1: MSW treatment alternatives from the literature

Four different options of MSW treatment, namely landfilling of MSW with energy recovery, incineration of MSW with energy recovery, biological treatment of the organic fraction of MSW (OFMSW) with energy recovery from RDF and thermal plasma gasification were selected to be evaluated from an environmental approach. The former three were largely based on the study by Arena et al. (2003) for the area of Regione Campania with the composition of MSW shown in Table 8.1. Meanwhile, the analysis for the latter was based on the methodology developed in Chapter 4.

Table 8.1. Composition of MSW in Regione Campania, Italy (Arena et al., 2003)

Waste component	**%**	**Waste component**	**%**
Glass	5.7	Plastics, hard	2.84
Metals	3.25	Textiles	4.48
Wood	1.75	Leather	1.76
Food wastes	30.1	Oversize	0.7
Greens	3.88	Inert materials	1.26
Paper and cardboard	23.15	Miscellaneous	4.49
Plastics, light	7.92	Fines	8.7

8.2.1.1. System boundary definition

Given that the scope of this chapter was to compare and rank MSW treatment techniques, the system boundaries were established at the treatment plant, assuming that previous stages of generation, collection and transport were common in all cases and did not add any useful information to distinguish one alternative from others.

The functional unit was 1 kg of MSW treated in all cases.

8.2.1.2. Description of alternatives

The alternatives considered for the MSW treatment were: landfilling with energy recovery (a_1), incineration with energy recovery (a_2), biological treatment of the OFMSW with energy recovery from RDF (a_3) and thermal plasma gasification (a_4). They were extracted from the work by Arena et al. (2003) and are briefly described below.

8.2.1.2.1. Landfill with energy recovery

The treatment plant was constituted by a landfill properly equipped with high quality bottom and top barriers with low permeability material such as high density polyethylene (HDPE or clay) to avoid leaching leaks and was operated by dumping vehicles. The leachate was collected and sent to a specific treatment unit. Besides, the biogas (mainly composed of CO_2 and CH_4) was collected with 55% efficiency; 60% of this was burned in a gas engine with an electric conversion efficiency of 35%, whereas the remaining biogas was sent to a flare to reduce greenhouse gas emissions. A scheme of the process is presented in Figure 8.1.

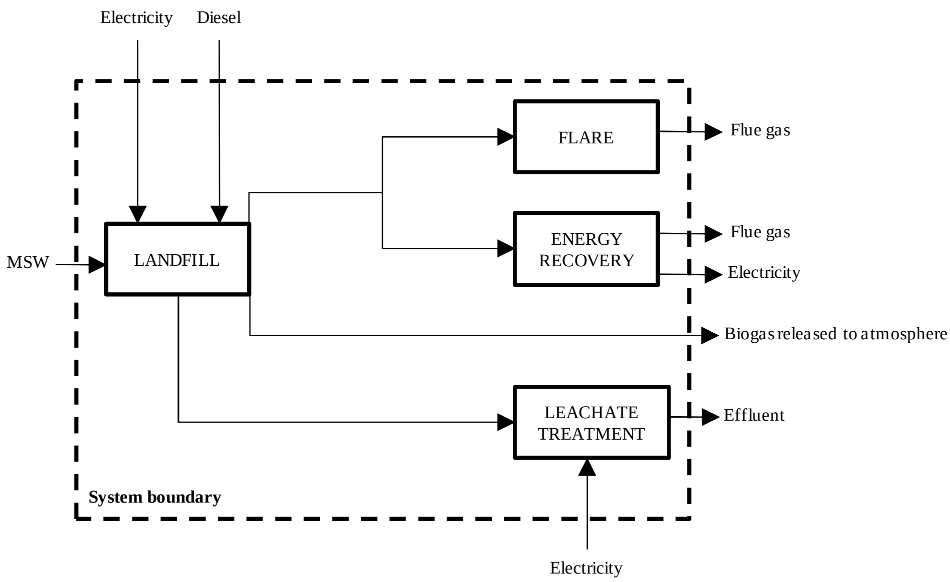


Figure 8.1. Flowsheet of the landfill with energy recovery process (adapted from Arena et al., 2003).

8.2.1.2.2. Incineration with energy recovery

Incineration is a waste valorization alternative that allows for energy recovery, although this is highly influenced by the water content of the OFMSW (organic fraction of the MSW), without a careful preliminary sorting process. The incineration process is quite simple and, if properly managed, it is feasible to fulfill legal requirements. The scheme considered here comprised four stages: pre-sorting of waste, combustion, energy recovery, flue gas treatment and ash conditioning (Figure 8.2). The technology selected for the combustion stage was a furnace with a mobile grate cooled by water. In a secondary chamber, the combustion of volatile unburned compounds was ended by adding a secondary air stream. The flue gases treatment consisted of a semi-dry scrubber for acid treatment, a fabric filter for fly ash removing and a catalytic reduction of NO_x and organic micro-pollutants. Water was totally vaporized and fly ashes recovered in the flue gas treatment stage were conditioned before landfill dumping, while slag and ash from the combustion stage were directly landfilled.

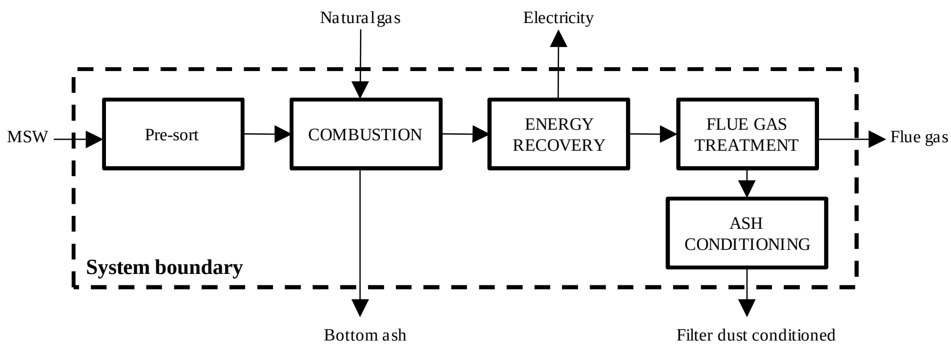


Figure 8.2. Flowsheet of the incineration process (adapted from Arena et al., 2003).

8.2.1.2.3. Biological treatment of the OFMSW with energy recovery from RDF

This alternative consisted of two main process lines: RDF production and combustion and biological treatment of the OFMSW (Figure 8.3). In the sorting stage four output streams were obtained: RDF bales for combustion, rest fraction for landfill dumping, ferrous material that could be recovered and the organic fraction for biological stabilization. The latter was stabilized under aerobic conditions (air was continuously forced through the waste pile to maintain a high decomposition rate) to produce compost that could be used for soil remediation. The exhaust air was treated in a scrubber and a bio-filter to reduce odors and pollutants before being released. For the RDF combustion, the same technology described for the incineration scenario was considered.

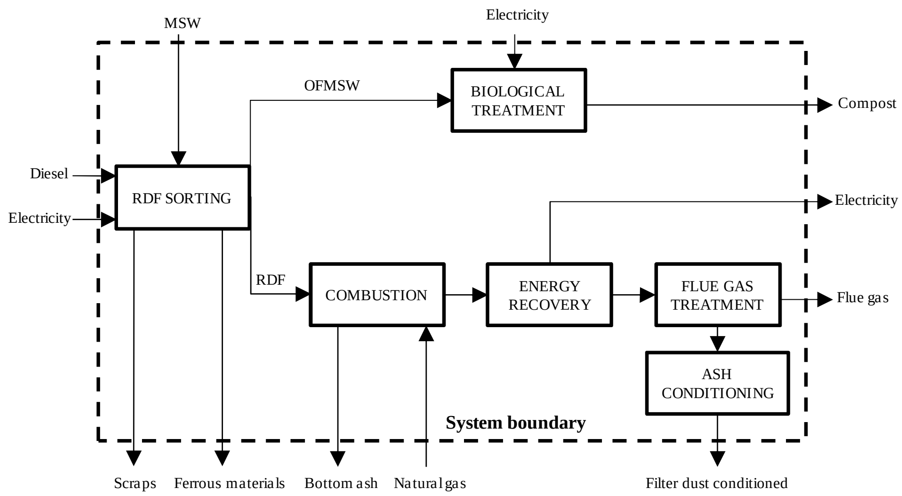


Figure 8.3. Flowsheet of the biological treatment of the OFMSW with incineration of RDF incineration process (adapted from Arena et al., 2003).

8.2.1.2.4. Thermal plasma gasification

A simultaneous dual reaction process takes place in a plasma reactor: the organic compounds are thermally decomposed into their constituent elements (syngas with more complete and advantageous conversion of carbon into gas than in incinerators), while the inorganic materials are melted and converted into a dense, inert and nonleachable vitrified slag, which does not require controlled disposal (Tendler et al., 2005). Plasma treatment is ideally suited for toxic wastes and complex waste streams that have recoverable energy content. The high temperature of the plasma arc greatly reduces the amount of undesirable by-products that are generated (Vaidyanathan et al., 2007).

The syngas cleaning system removed particles and other substances that could harm the energy recovery equipment or that could imply the release of undesired pollutants (e.g. acidifying gases). After cleaning, the syngas (mainly composed of CO and H_2) was combusted in a combined heat and power cycle to produce electricity. Vitrification is the result of the interaction between the plasma and inorganic materials. Because the inert fraction is vitrified and harmful substances can barely leach from the lava, this product can be used for road construction or as a building material (Tendler et al, 2005). The main stages of the process were: plasma gasification/vitrification, syngas cleaning and energy recovery from syngas (Figure 8.4).

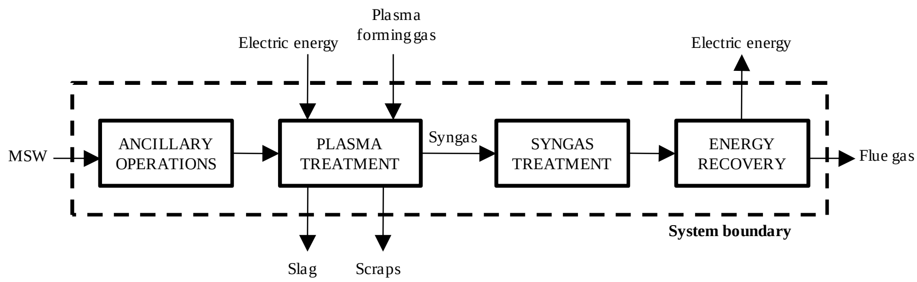


Figure 8.4. Flowsheet of the thermal plasma gasification process.

8.2.1.5. EF analysis

Input and output flows included in the EF estimates were those that could properly be assessed with this methodology (as discussed in previous chapters of this thesis) and reflected the most significant environmental burdens of the waste treatment alternatives considered. Energy balances were carried out to take into account electricity generation in all the alternatives. Among air emissions, only those with global warming and acidifying effect were accounted for by means of assimilation factors. Finally, the EF for solid waste was the area required for landfill dumping.

Inventory data considered for alternatives a_1, a_2 and a_3 were collected in Tables 8.2 to 8.5. The transformation of energy consumption, global warming and acidifying emissions into area units was conducted similarly to previous chapters. For solid wastes conversion factors were extracted from Huijbregts et al. (2008) and from Ecoinvent database: 0.860 kg m^{-2} yr^{-1} for filter dusts, bottom ash and scraps; 0.053 kg m^{-2} yr^{-1} for stabilized solid waste. For those materials that were recycled a counter footprint (CF) was appraised by means of avoided impacts: 171.8 kg m^{-2} yr^{-1} for the stabilized organic fraction; 0.11 kg m^{-2} yr^{-1} for ferrous material; 1228.1 kg m^{-2} yr^{-1} for slag. Equivalence factors from Kitzes et al. (2007) were applied.

Table 8.2. Inventory for the EF estimate of the landfill with energy recovery alternative (adapted from Arena et al., 2003).

Input flow	Value	Units	Output flow	Value	Units
MSW	1	kg	Electricity generated	$3{\cdot}10^{-1}$	MJ
Energy			Air emissions		
Electricity	$2.6{\cdot}10^{-2}$	MJ	CH_4	$2.10{\cdot}10^{-2}$	kg
Diesel (dumping vehicles)	$7.4{\cdot}10^{-3}$	MJ	CO_2	$1.78{\cdot}10^{-1}$	kg
			CO	$1.19{\cdot}10^{-1}$	kg
			NO_x	$1.07{\cdot}10^{-1}$	kg
			SO_x [(a)]	$8.06{\cdot}10^{-6}$	kg
			Stabilized solid waste	$4.9{\cdot}10^{-1}$	kg

[(a)]Estimation based on US EPA (1998).

Table 8.3. Inventory for the EF estimate of the incineration alternative (adapted from Arena et al., 2003).

Input flow	Value	Units	Output flow	Value	Units
MSW	1	kg	Electricity generated	2.42	MJ
Energy			Air emissions		
Natural gas (heat)	$3.6{\cdot}10^{-2}$	MJ	CO_2	$9.53{\cdot}10^{-1}$	kg
			NO_x	$1.97{\cdot}10^{-3}$	kg
			SO_2	$1.97{\cdot}10^{-4}$	kg
			CO	$9.80{\cdot}10^{-5}$	kg
			Solid waste		
			Filter dusts	$9.00{\cdot}10^{-2}$	kg
			Bottom ash	$1.70{\cdot}10^{-1}$	kg

Table 8.4. Inventory for the EF estimate of the biological treatment of the OFMSW with incineration of RDF alternative (adapted from Arena et al., 2003).

Input flow	**Value**	**Units**	**Output flow**	**Value**	**Units**
MSW	1	kg			
RDF production + biological treatment					
Energy			Air emissions		
Diesel	$1.00 \cdot 10^{-2}$	MJ	CO_2	$2.00 \cdot 10^{-1}$	kg
Electricity	$8.30 \cdot 10^{-2}$	MJ	Solid waste		
			Scraps	$5.00 \cdot 10^{-2}$	kg
			Compost	$3.70 \cdot 10^{-1}$	kg
			Ferrous materials	$5.00 \cdot 10^{-2}$	kg
			RDF	$4.00 \cdot 10^{-1}$	kg
RDF combustion					
Energy					
Natural gas (heat)	$1.44 \cdot 10^{-2}$	MJ	Electricity generated	1.64	MJ
			Air emissions		
			CO_2	$6.06 \cdot 10^{-1}$	kg
			NO_x	$1.33 \cdot 10^{-3}$	kg
			SO_2	$1.33 \cdot 10^{-4}$	kg
			CO	$6.70 \cdot 10^{-5}$	kg
			Solid waste		
			Filter dusts	$3.60 \cdot 10^{-2}$	kg
			Bottom ash	$4.40 \cdot 10^{-2}$	kg

For the estimate of the EF associated to the thermal plasma gasification the model developed in Chapter 4 was used. Thus, the average carbon content for the MSW stream was appraised using data from Table 8.1 and Table 4.1. This value, together with the amount of MSW used as functional unit (1 kg), was introduced into the tool to obtain the EF.

8.2.1.6. MCA analysis

In addition to the EF (g_1), other 5 criteria were employed to conduct the MCA analysis of the alternatives studied in this chapter: water consumption (g_2), air emissions of organic compounds (g_3), air emissions of dusts (g_4), water emissions of suspended solids (g_5) and occupied landfill volume (g_6). This way, impact burdens not evaluated by the EF were included in the analysis. Besides, these criteria were selected in accordance with those employed in the life-cycle analysis conducted by Arena et al. (2003), thus favoring comparability of conclusions. For the plasma technology, water consumption was extracted from MPM Tech. (2005); meanwhile, the occupied landfill volume was considered to be zero (Plasco, 2008). To estimate the air emissions of organic compounds data from Plasco (2008) and Gallego (2008) were employed. During the thermal plasma gasification no water streams are generated and the release of dusts to air does not occur.

Legal thresholds existed for the majority of the criteria selected, especially associated to the incineration of waste (European Commission, 2000). Particular concern is regarded to air emissions of organic compounds and dusts because of their toxic effects in human health. Suspended solids in water reduce the photosynthetic activity of aquatic vegetation and may cause technical problems. The importance of water consumption will depend on the conditions of the area (in case of scarcity the environmental impact will gain in significance). Finally, the occupied landfill volume offers a measure of the capability to reduce the volume of waste; apart from visual impact in landscape and odor, this criterion is important because of the requirement of land to dispose of waste.

The most restrictive limits exist for air emissions of organic compounds and dusts, followed by suspended solids in water and closely by CO, SO_2 and then NO_x (European Commission, 2000). Thus, the priority of criteria was established in the following order: $g_3 > g_4 > g_1 > g_5$. EF was situated before suspended solids in water because it aggregated a number of environmental impacts, including CO, SO_2 and NO_x. Regarding g_2 and g_6, it was difficult to establish a clear order of priority. None of them affected human health or ecosystems (if landfill is properly managed) and therefore, they were placed at the end. To distinguish between them, three hypotheses were considered based on the space and water availability in a given scenario: H_1) the two criteria are equally important; H_2) g_6 is moderately more important than g_2; H_3) g_2 is moderately more important than g_6.

Given that all criteria were quantitative, the maximum range for each criterion was used to relate the variability among the different alternatives and the Saaty's scale in AHP. Weights were normalized to make them range between 0 and 1.

The same weights derived from the AHP hierarchy were employed to apply the outranking methods ELECTRE I, ELECTRE III and PROMETHEE/GAIA. The establishment of thresholds required for the application of ELECTRE III was also based on criteria amplitude: indifference (10%), preference (50%) and veto (80%). For PROMETHEE I and II the software Decision Lab 2000 (Visual Decision Inc., 2009) was employed: all criteria were set to be minimized, linear functions were selected (this definition in the software corresponds to the V-shape with indifference criterion described in Chapter 2) and *p* and *q* parameters were established as for ELECTREE.

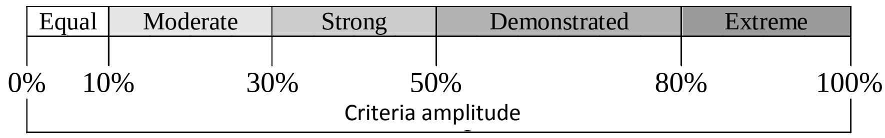


Figure 8.5. Equivalence between Saaty's scale and criteria amplitude.

8.2.2. Case study 2: LIPOR

LIPOR – Intermunicipal Waste Management of Greater Porto (Portugal) – is the entity in charge for the management, recovery and treatment of the MSW produced by the eight partner municipalities: Espinho, Gondomar, Maia, Matosinhos, Porto, Póvoa de Varzim, Valongo and Vila do Conde (Figure 8.6). Built as a Municipalities' Association in 1982, LIPOR treats every year about 480 thousand tons of MSW produced by about 1 million inhabitants. LIPOR has developed an integrated strategy of valorization, treatment and confinement of MSW, based on three components: multi-material valorization, organic valorization and energy valorization, completed by a sanitary landfill to dispose of refuse and waste previously prepared (LIPOR, 2011).

Table 8.5 summarizes the generation of wastes and the distribution among the different waste streams and treatment processes during the years 2007-2009.

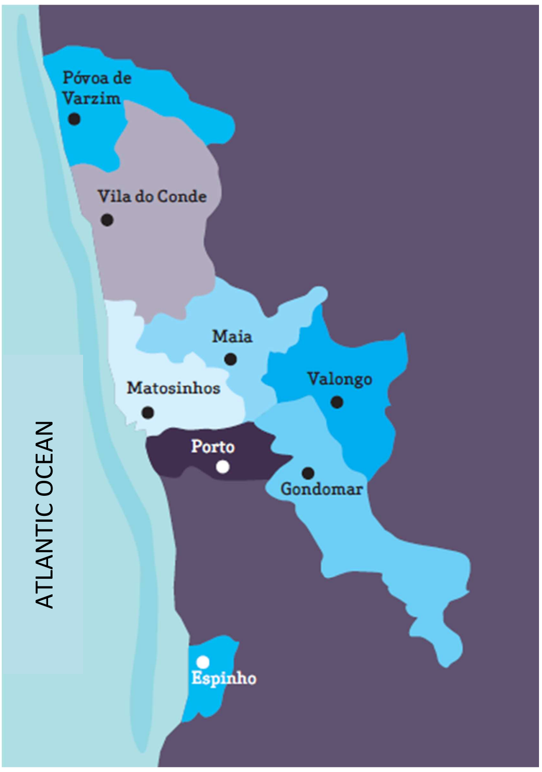


Figure 8.6. Partner municipalities of LIPOR (LIPOR, 2011).

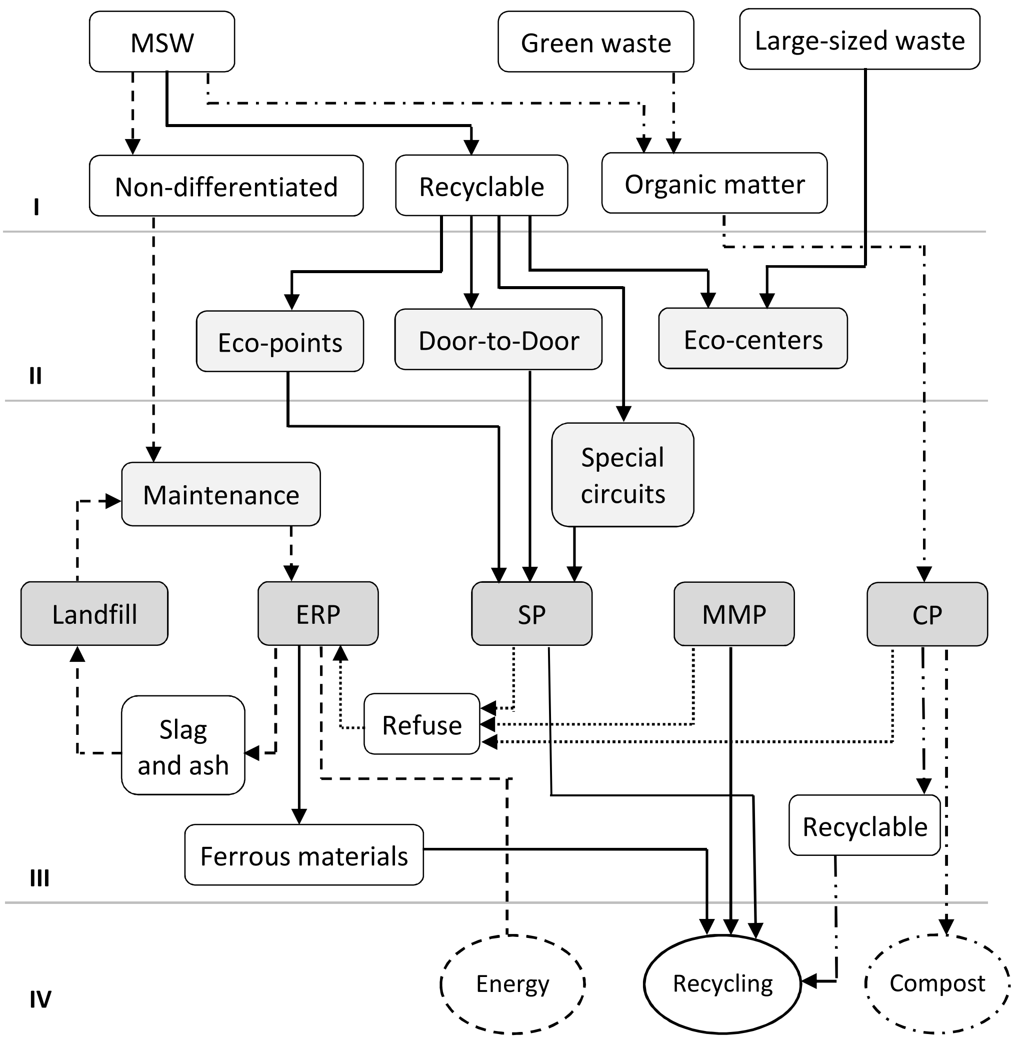
Table 8.5. Generation and distribution of wastes in LIPOR (LIPOR, 2007-2009).

Parameter	**2007**	**2008**	**2009**	**Units**
Population (inhabitants)	972,301	972,301	970,704	
Waste streams				
Multi-material valorization	49,884	55,470	59,966	t
Organic valorization	30,730	37,146	42,215	t
Energy valorization	419,389	383,553	398,392	t
Landfilling	27,185	63,308	39,339	t
Total	527,188	539,477	539,912	t

The multi-material valorization process starts at the separated disposal and collection of wastes, for which adequate infrastructure is available for citizens. LIPOR's Sorting Plant (SP), prepared to treat 35.000 tons of waste per year, makes a complementary separation, sorting the materials from the selective collection, baling and sending them to the recycling plants. The facility, in a closed building of 4000 m^2, is equipped with two sorting lines: flat products line (paper and cardboard); large products line (packages made of plastic, metal and packs for liquid food products).

The biowaste recovery is ensured through a Composting Plant (CP) that can process nearly 60.000 t $year^{-1}$ of organic waste, associated with the implementation of organic fraction removal circuits at great producers (restaurants, hypermarkets, markets) and in the areas of door-to-door selective collection (collection of the organic fraction of the household waste), complemented by local initiatives of homely composting. The application of the organic matter in the soil is highly beneficial, namely on the improvement of the soil characteristics (porosity, water retention ability) and in the prevention of the soil degradation, contributing, effectively, for the minimization of erosion, aggregation, salinization and desertification of these sites.

The main purpose of the Energy Recovery Plant (ERP) is the controlled thermal treatment of MSW without recovery potential through organic and multi-material recycling processes, recovering their endogenous energy for the production of electrical energy. Waste arrives to the ERP coming from the several LIPOR partner municipality circuits and is stored in a reception pool with capacity for a 6 day-production amount. This waste deposition occurs inside a closed building that is kept under negative pressure to avoid the spread of odors. It is then transferred by special lifting equipment into two lines of treatment where it is incinerated at high temperatures (1000º C to 1200ºC) in an oxygen saturated environment. The plant is energetically self-sufficient and sends 90% of its production to the Portuguese National Electrical Network. Inert gases and materials originated by the combustion process undergo a strict control and environmental monitoring system. Before being released into the atmosphere, gases are neutralized and filtered through highly effective equipment, while ashes (once made inert) together with slag are confined in a sanitary landfill.



I – Citizen: production and disposal
II – Councils: production and disposal
III – LIPOR: valorization and treatment
IV – Clients: products and resources
MMP = Multi-material platform

Figure 8.7. IWMS of LIPOR (adapted from LIPOR, 2009).

Figure 8.7 shows the whole Integrated Waste Management System (IWMS) of LIPOR. In the second part of this chapter an environmental assessment of the main processes of this system was conducted based on a joint application of EF, MFA and LCA. Inventory data for the period 2007-2009 were employed (LIPOR, 2007-2009). For clarity, these were collected in Annex A8. Transport stages were excluded from the analysis. Waste water treatment was not included because this was not conducted in a homogenous way, showing a clear tendency to be totally externalized (therefore it was set outside the system boundaries considered). Finally, the biogas plants recently implemented as pilot studies were also excluded because of lack of data. To conduct the EF appraisal, a similar protocol as for the previous case study was followed, accounting for energy and materials consumption. The sub-products of each treatment processes were not included given that these streams moved from one plant to another (Figure 8.7) until their final disposal or reuse. The Portuguese national electricity mix was used to break down electricity into the primary sources of energy (Table 8.6). LCA was applied to assess the environmental impacts derived from air emissions, mainly released in the ERP, using the impact assessment methodology of CML (Table 8.7). It must be noticed that, for global warming gases, the CO_2 equivalent value was directly supplied. Further, these CO_2 emissions reported were the result of energy consumption in the different sections of LIPOR; therefore, given that energy consumption was assessed by means of EF, CO_2 emissions were not further considered to avoid double counting. A tool adapted to the case study was developed in MS Excel®, in which the process and year could be selected to easily conduct the calculations.

Table 8.6. Portuguese national electricity mix (LIPOR, 2009).

Sources	**Annual average (%)**
Fossil	
Carbon	17.85
Natural gas	34.39
Fuel-oil	1.46
Nuclear	5.95
Cogeneration	8.29
Renewable	
Hydro power	16.44
Wind power	13.75
Others	1.86

Table 8.7. Characterization and normalization factors employed to appraise the environmental impact of air emissions in the ERP (CML, 2010).

Pollutant	Characterization factors (baseline)							
	GWP 100 CO_2 eq	AP SO_2 eq	HTP 1,4-DB eq	POCP C_2H_4 eq	EP PO_4 eq	FAETP 1,4-DB eq	MAETP 1,4-DB eq	TETP 1,4-DB eq
CO_2	1							
CH_4	25			$6.0 \cdot 10^{-3}$				
N_2O	300							
HCl			0.5					
NO_x		0.5	1.2		0.13			
HF			$2.9 \cdot 10^{3}$			4.6	$4.1 \cdot 10^{7}$	$2.9 \cdot 10^{-3}$
SO_2		1.2	$9.6 \cdot 10^{-2}$	$4.8 \cdot 10^{-2}$				
Particulates			0.82					
CO				$2.7 \cdot 10^{-2}$				
Dioxins[(b)]			$1.9 \cdot 10^{9}$			$2.1 \cdot 10^{6}$	$3.0 \cdot 10^{8}$	$1.2 \cdot 10^{4}$
Normalization factors EU25[(a)]								
	$5.02 \cdot 10^{12}$	$2.81 \cdot 10^{10}$	$7.78 \cdot 10^{12}$	$8.48 \cdot 10^{9}$	$1.32 \cdot 10^{10}$	$5.19 \cdot 10^{11}$	$1.17 \cdot 10^{14}$	$4.86 \cdot 10^{10}$

[(a)]Normalization factors calculated for EU25 based on Western Europe data and using GDP.
[(b)]The material flow “dioxins and furans” (Table A8.1) was considered to be dioxins.

GWP100: Global Warming Potential 100 years
POCP: Photochemical Oxidation Potential
FAETP: Fresh water Aquatic EcoToxicity Potential
MAETP: Marine Aquatic EcoToxicity Potential
TETP: Terrestrial EcoToxicity Potential
AP: Acidification Potential
HTP: Human Toxicity Potential
EP: Eutrophication Potential
DB: Dichlorobenzene

The application of MCA among the MSW treatment options did not make sense in this case, since all of them were part of the IWMS of LIPOR and treated specific waste streams (sorted according to their characteristics that made them more or less suitable for the different treatment alternatives).

8.3. Results and discussion

This section first addresses the results for the first case study based on data from the literature; then, the environmental analysis for LIPOR is presented.

8.3.1. Case study 1: MSW treatment alternatives from the literature

The results from EF assessments are detailed and a ranking of alternatives is established on the basis of the values obtained, latter compared to that from MCA analysis.

8.3.1.1. EF of alternatives of MSW treatment

The EF values obtained for 1 kg of MSW treated were 13.2, 4.9, 3.3 and 3.4 gm^2 for alternatives a_1 to a_4, respectively (to apply the plasma methodology 37% average carbon content was estimated). Figure 8.8 shows the gross EF, CF and net EF for the four alternatives evaluated.

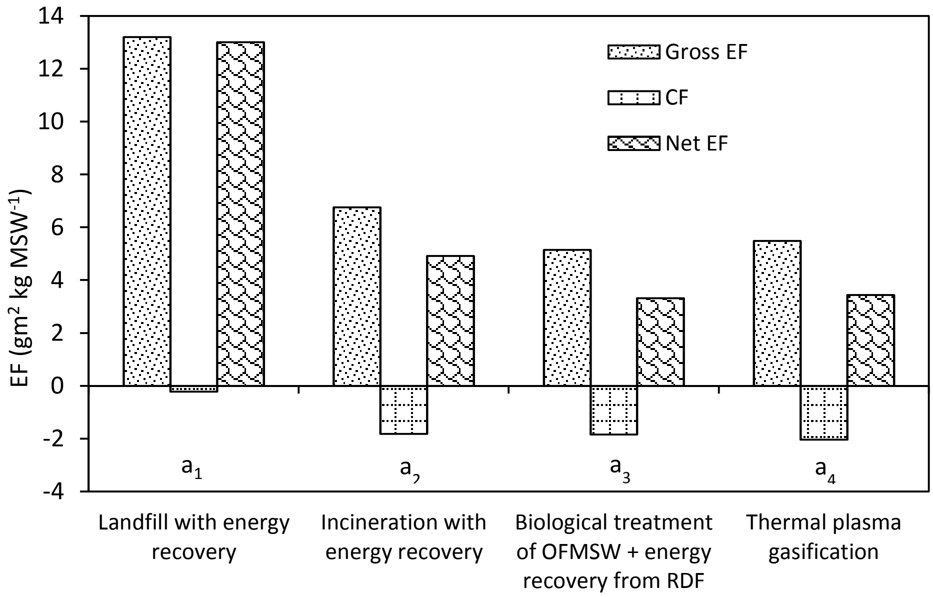


Figure 8.8. EF of the analyzed MSW treatment alternatives (results are expressed for 1 kg MSW treated).

The worst results in terms of EF were obtained for the landfilling option, with a main contribution from the area required to dispose of the stabilized solid waste. A quite insignificant contribution to CF of the biogas collected was observed. The EF was around three times higher than for the other three alternatives, for which air emissions posed the major environmental pressure. Also, the effect of energy recovery and material recycling had a visible effect on the contribution to CF, thus diminishing the net EF in a range from 27% to 37%. Energy recovery reduces the need for other energy sources, material from recycling processes replaces production of virgin material and biological treatment may reduce the need for production of artificial fertilizers and vehicle fuel (Ekwall et al., 2007). Incineration with energy recovery was undoubtedly ranked in second place, while the difference between biological treatment of OFMSW with energy recovery from RDF and thermal plasma gasification as not so clear. A higher gross EF was appraised for thermal plasma gasification, but CF due to high levels of energy recovery reached in this process reduces it to a net EF quite close to that for biological treatment of OFMSW and RDF combustion. Further, uncertainty associated to the difference in data sources and methodology used to assess the EF for the plasma process must be highlighted.

Cherubini et al. (2009) used the software tool SPIonexcel (Sandholzer and Narodoslawsky, 2007) to evaluate the environmental impact of waste treatment alternatives: landfill, landfill with biogas recovery, MSW sorting plant (separate organic fraction) and incineration. SPI (Sustainable Process Index) is an indicator that belongs to the ecological footprint family (it was presented in Chapter 1, section 1.3.2). The ranking of alternatives obtained by the authors was in agreement with the results of this chapter; nonetheless, the footprint values were very different. The value estimated by Cherubini et al. (2009) for landfill was very similar to that obtained in this chapter for landfill with biogas recovery, while for this latter the study by Cherubini et al. yielded -1.3 ha t^{-1} MSW. Also for sorting and incineration a negative area demand was estimated by these authors, being especially noticeable for MSW sorting plant (near -15 ha t^{-1} MSW).

Huijbregts et al. (2008) studied the EF of a number of processes extracted from Ecoinvent database (v1.2), some of them belonging to the category of incineration (73), landfill (113) and recycling (28). The order of magnitude of the EF values was closer to the range obtained in this chapter, being 5 m^2 kg^{-1} for the former and around 0.05 m^2 kg^{-1} for the latter two.

8.3.1.2. MCA ranking of alternatives

Once the EF was assessed, the multi-criteria decision matrix was completed for criteria defined in the methodology section (Table 8.8).

Table 8.8. Decision matrix for the selection of MSW treatment alternatives.

		Criteria					
		g₁ (gm^2)	***g₂*** (g)	***g₃*** (g)	***g₄*** (g)	***g₅*** (g)	***g₆*** (m^3)
Alternatives	***a₁***	13.2	0	$1.2\cdot10^{-1}$	$2.7\cdot10^{-2}$	0.03	$1.43\cdot10^{-3}$
	a₂	4.9	175.2	$2.0\cdot10^{-3}$	$4.9\cdot10^{-2}$	6.79	$2.7\cdot10^{-4}$
	a₃	3.3	70.8	$1.6\cdot10^{-3}$	$3.32\cdot10^{-2}$	1.23	$4.9\cdot10^{-4}$
	a₄	3.4	151.2	$1.2\cdot10^{-4}$	0	0	0

Preferences were established for the three hypotheses considered (see section 8.2.1.6) using the Saaty's scale (Tables 8.9 to 8.11) and normalized weights were assessed (Table 8.12).

Table 8.9. Matrix expressing intensities of preference among criteria under H_1.

	g_1	**g_2**	**g_3**	**g_4**	**g_5**	**g_6**
g_1	1	5	1/5	1/3	3	5
g_2	1/5	1	1/9	1/7	1/3	1
g_3	5	9	1	3	7	9
g_4	3	7	1/3	1	5	7
g_5	1/3	3	1/7	1/5	1	3
g_6	1/5	1	1/9	1/7	1/3	1

Table 8.10. Matrix expressing intensities of preference among criteria under H_2.

	g_1	g_2	g_3	g_4	g_5	g_6
g_1	1	7	1/5	1/3	3	5
g_2	1/7	1	1/9	1/9	1/5	1/3
g_3	5	9	1	3	7	9
g_4	3	9	1/3	1	5	7
g_5	1/3	5	1/7	1/5	1	3
g_6	1/5	3	1/9	1/7	1/3	1

Table 8.11. Matrix expressing intensities of preference among criteria under H_3.

	g_1	g_2	g_3	g_4	g_5	g_6
g_1	1	5	1/5	1/3	3	7
g_2	1/5	1	1/9	1/7	1/3	3
g_3	5	9	1	3	7	9
g_4	3	7	1/3	1	5	9
g_5	1/3	3	1/7	1/5	1	5
g_6	1/7	1/3	1/9	1/9	1/5	1

Table 8.12. Normalized weights for criteria calculated with AHP and used for all MCA methods.

	g_1	g_2	g_3	g_4	g_5	g_6
W (H_1)	0.14	0.03	0.47	0.26	0.07	0.03
W (H_2)	0.14	0.02	0.46	0.27	0.07	0.04
W (H_3)	0.14	0.04	0.46	0.27	0.07	0.02

Figures 8.9 to 8.11 show the results of the application of AHP and ELECTREE I and III. A robustness analysis was performed to ensure that the parameters and weights selected during the analysis did not significantly influence the ranking of

alternatives. As it can be observed, very similar results were obtained for the three hypotheses proposed.

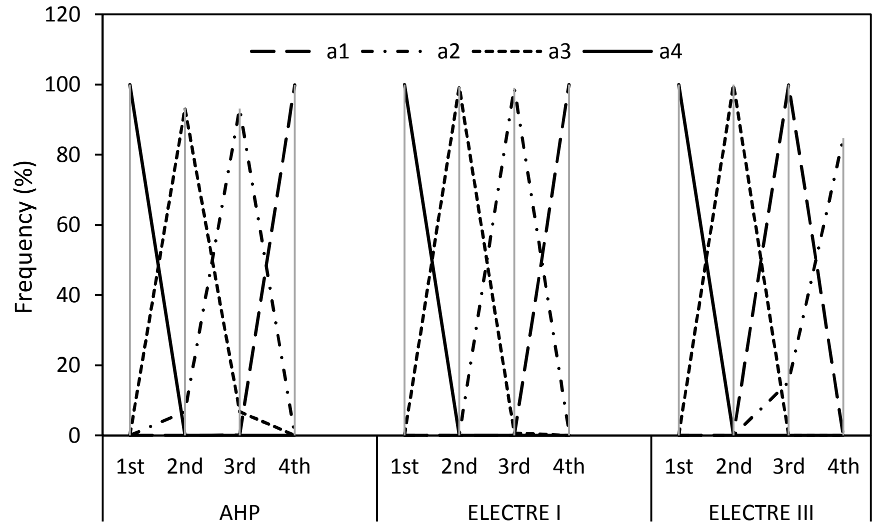


Figure 8.9. Frequency for alternative rankings under H_1, after the robustness analysis.

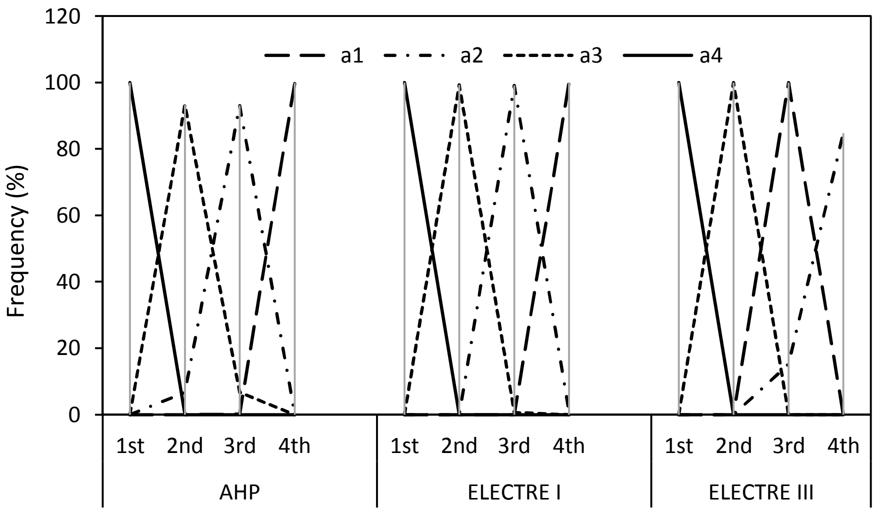


Figure 8.10. Frequency for alternative rankings under H_2, after the robustness analysis.

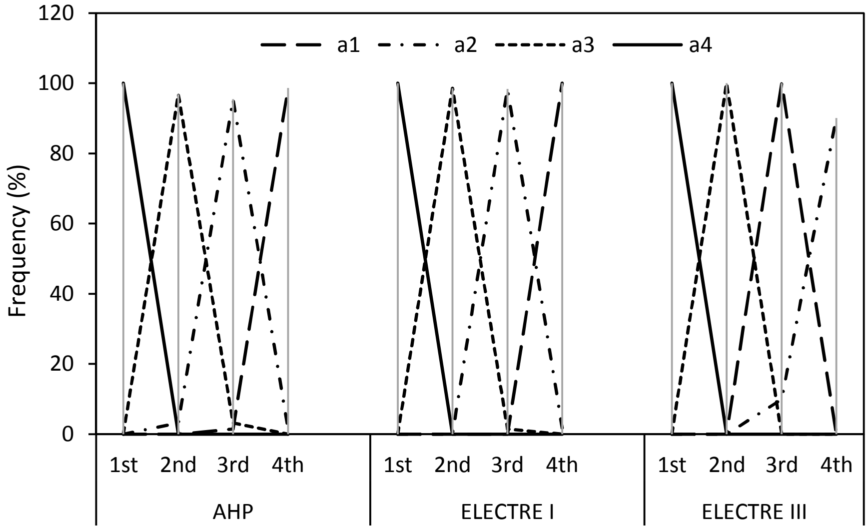


Figure 8.11. Frequency for alternative rankings under H_3, after the robustness analysis.

According to the robustness analysis, a_4 was always ranked in first place and a_3 in second place in the majority of cases (more than 90%) for the three MCA methodologies and under the three hypotheses. However, for the third and fourth positions the MCA methodologies were in disagreement. While AHP and ELECTRE I ranked a_2 and a_1 in third and fourth places, respectively, these alternatives exchanged positions according to ELECTRE III.

Results obtained with Decision Lab for the PROMETHEE family outranking methods are shown in Figures 8.12 and 8.13. Both PROMETHEE I and II yielded the same ranking order than AHP and ELECTRE I. Criteria weights were modified in the software within a reasonable interval to conduct a sensitivity analysis and the ranking of alternatives did not change.

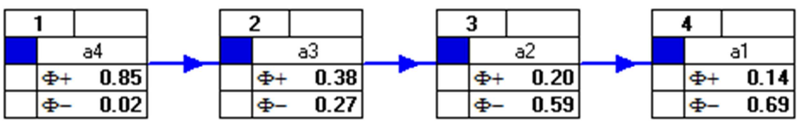


Figure 8.12. Ranking of alternatives according to PROMETHEE I.

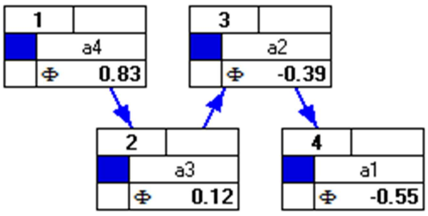


Figure 8.13. Ranking of alternatives according to PROMETHEE II.

Similar conclusions could be extracted from the GAIA plane (Figure 8.14), which had a Δ = 91.57%, thus ensuring its reliability. The *pi* decision axis clearly pointed at a_4 as compromise solution among all criteria, being a_3 quite close and a_2 and a_1 in opposite direction. All criteria seemed to be in agreement with each other as their axes had the same orientation (only water consumption – g_2 – could be considered to be moderately conflicting). Further, a_3 and a_4 conveyed strong features for most of the criteria, conversely to a_1 and a_2.

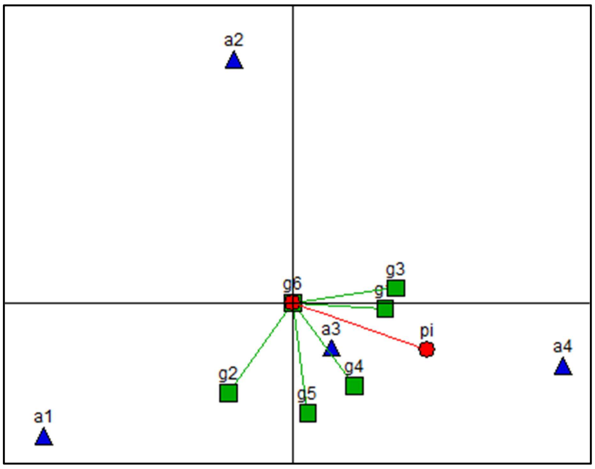


Figure 8.14. Gaia plane for the selection of MSW alternatives problem.

All the MCA employed, except for ELECTRE III, ranked the MSW treatment alternatives in the following order: 1) thermal plasma gasification; 2) biological treatment of the OFMSW with incineration of RDF; 3) incineration; 4) landfilling with energy recovery. This can be considered to be in agreement with the general recommended hierarchy for waste management (European Commission, 2008), since energy recovery and the generation of valuable sub-products is prioritized with respect to landfilling. Besides, when incorporating more environmental indicators than EF, a_4 was more clearly identified as the best option. Nonetheless, the EF proved to be a good screening indicator.

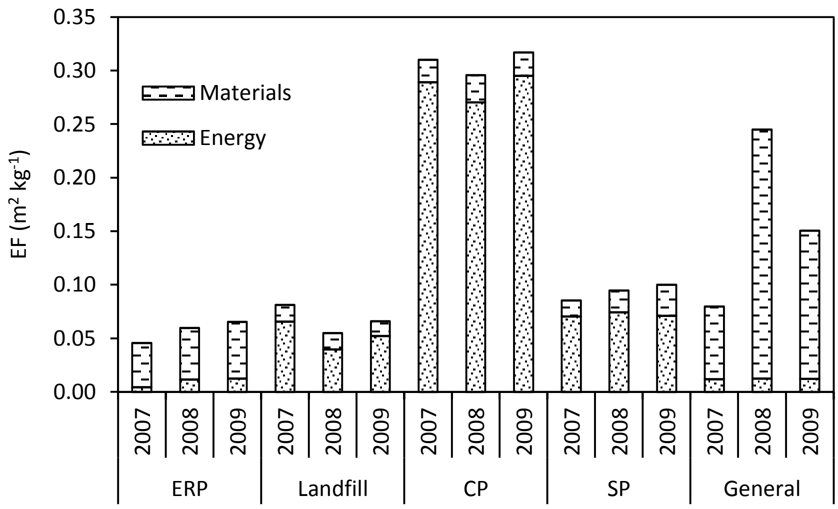
The different ranking offered by ELECTRE III could be explained by the consideration of uncertainty during the analysis by using pseudo-criteria (Figueira et al., 2005). These results could raise doubts on whether a properly managed landfill could be better than incineration without a previous sorting of wastes. The selection of one or other would be conditioned by the geographical location and the resources availability, as well as by social, political or economic aspects not being considered here.

The selection of the plasma process as best alternative must be judged cautiously. This is a novel technology for which a small number of industrial applications exist up to date. Besides, it was not initially conceived as a process for the treatment of MSW but for industrial ones, given its ability to decompose hazardous substances.

8.3.2. Case study 2: LIPOR

In this section the results obtained from the environmental assessment of the MSW treatment processes of LIPOR are presented. Figure 8.15 summarizes the relative EF values for energy recovery, landfill, composting and sorting plant, as well as that associated to general activities of LIPOR (total EF was divided by the amount of wastes treated in each process and by the total waste stream in the last case – Table 8.5). Given the range of values, the EF was expressed in gha kg^{-1}. The contribution of main categories energy and materials is indicated for the three years analyzed. Regarding the time-frame analysis, no clear tendencies throughout the years were observed. For ERP and SP there was a continuous increase in the relative EF. Meanwhile, for landfill and CP the environmental impact decreased in 2008 and rose again in 2009; conversely, the relative EF associated to general activities had the opposite behavior.

The CP clearly presented the highest EF, mainly due to energy consumption; it was followed by the general activities, but in this case the major contribution came from materials. Conversely, the ERP had the lowest EF, closely followed by the landfill (it must be noticed that landfill area was not included in this figure, as it is explained later). It must be highlighted that only the gross footprint is being conveyed in Figure 8.15. Therefore, for ERP and CP a counter footprint term could be considered by means of electricity generation and the compost obtained, respectively. According to the sustainability reports of LIPOR (2009), the ERP generates 200,000 MWh yr^{-1}, from which 90% are exported to the national electrical network. This is translated into an average CF figure of 0.87 $gm^2\ kg^{-1}$, thus confirming ERP as the most beneficial treatment process from an environmental point of view. The CP yields around 20,000 t compost every year. Using the same conversion factor employed in case study 1 (section 8.2.1.5) the average CF estimated was 0.0043 gha kg^{-1}. In this case the gross EF would not significantly be reduced.



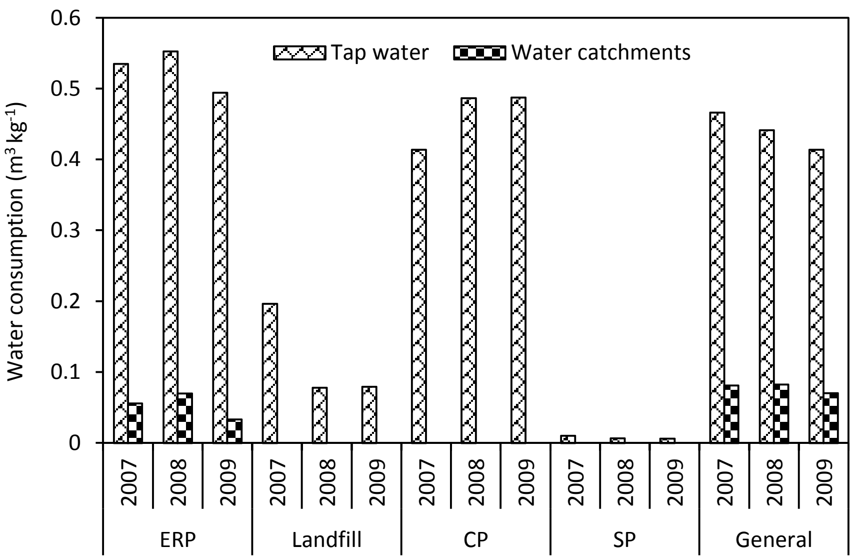
ERP: Energy Recovery Plant CP: Composting Plant SP: Sorting Plant

Figure 8.15. EF estimate of the LIPOR waste treatment processes.

The analysis conducted only referred to operational data. The allocation of the area occupied by buildings and infrastructure is difficult to carry out, since the expected working time-horizon of each installation should be estimated. For example, the landfill area is 520,000 m^2, but it would be necessary to know the volume and density of wastes to estimate when it will be closed, and therefore, appraise a ratio area/kg MSW disposed of. Besides, the space occupied by the installations did not pose a difference among years, although it must be taken into account that this contribution to EF is systematically being excluded in this assessment.

The total EF of the LIPOR system was also calculated and referred to the number of inhabitants, yielding 79.5, 179.8 and 133.2 gm^2 $inhabitant^{-1}$ for 2007, 2008 and 2009, respectively. The worst ratio was obtained for 2008, moderately improved in 2009. When referring the total EF of LIPOR to 1 kg of MSW entering the system (i.e. the totally of wastes treated in the different plants), the figures estimated were 0.15, 0.32 and 0.24 gm^2 kg^{-1} MSW, conveying a similar tendency.

Other material flows collected in inventory tables (annex A8) that were not appraised by the EF were the air emissions released in the ERP and water consumptions. Among the waste treatment processes, the energy recovery clearly represented the major contribution to water requirements in absolute terms; large quantities of water were also consumed in general facilities of LIPOR (Tables A8.1 and A8.5). Tap water was the main source of water, although water from catchments was also supplied in ERP and general activities. However, when analyzing the relative flows, the CP gained in relevance being comparable to the general activities consumption and much closer to the ERP (Figure 8.16).



ERP: Energy Recovery Plant CP: Composting Plant SP: Sorting Plant

Figure 8.16. Water consumption in the different plants of LIPOR (allocated to 1 kg of waste treated in each process).

Regarding air emissions, the evaluation provided by LCA applying the CML impact assessment method is presented in Figures 8.17 and 8.18, corresponding to the characterization and normalization stages, respectively. Figure 8.17 conveys the partial contribution of each emissions flow within an impact category. All impact categories affected by the compounds released in the ERP were included. Hydrogen fluoride was the chemical that more significantly affected the toxicity categories (HTP, FAETP, MAETP and TETP). HF has the R26/27/28 risk phrases associated, this meaning that it is very toxic by inhalation, in contact with skin and if swallowed (ESIS, 2011). The most relevant contribution to AP and EP environmental impact categories was due to nitrogen oxides; meanwhile, sulfur dioxide achieved major importance in the photochemical oxidation category, together with CO. Particulates and HCl did not show noticeable contribution to any of the impact categories.

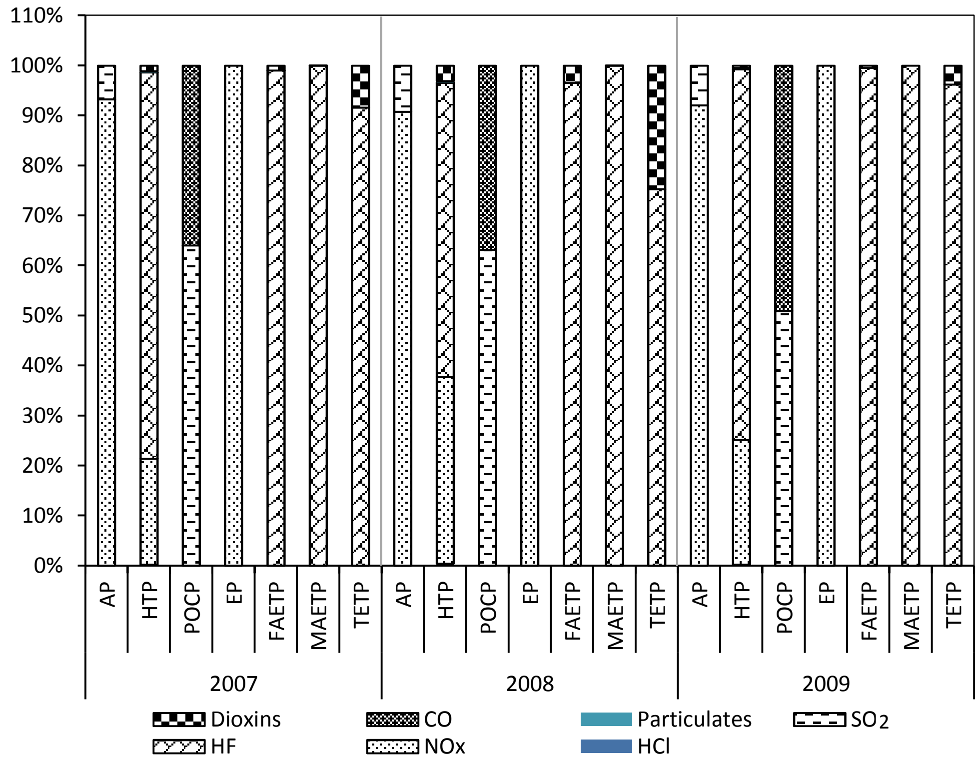


Figure 8.17. Impact assessment of air emissions released in the ERP: characterization stage.

The normalization stage was conducted to evaluate the relative importance of the different environmental impact categories. As indicated in the methodology section, data regarding global warming emissions was not available. Instead, total CO_2 equivalent emissions were reported. This is why GWP100 was not included in Figure 8.17 (the partial contribution of the different gases could not be estimated) but it was in Figure 8.18. However, this was done to be able to compare impact categories, since CO_2 emissions were already appraised by the EF by means of energy consumption. The most relevant impact category was MAETP for the three years studied, although an important decreased occurred during 2008. This may be of concern taking into account that LIPOR is located close to the Atlantic coast. GWP100 was the second most important category, followed by AP and EU. It could be concluded that HF (affecting MAETP) and NO_x emissions (affecting AP and EP) were the air emissions that posed a major potential damage to the environment.

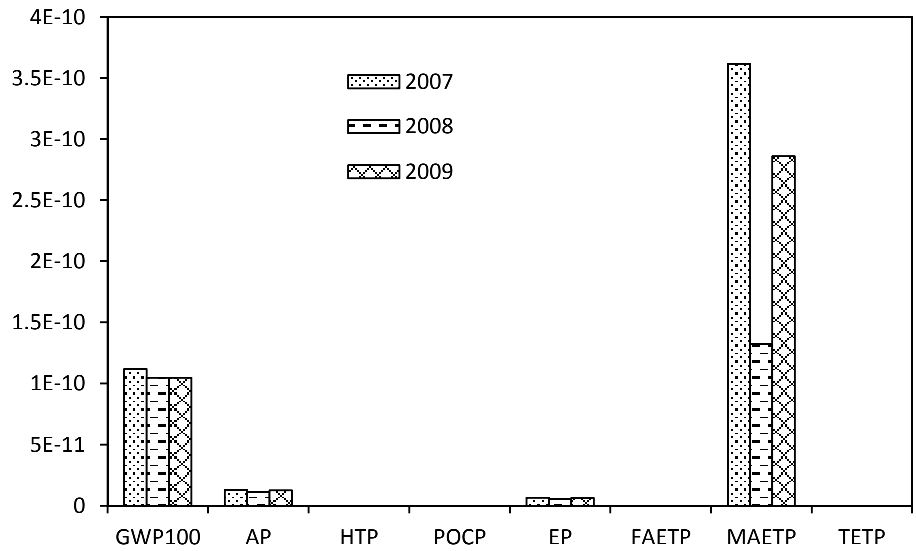


Figure 8.18. Impact assessment of air emissions released in the ERP: normalization stage. Results referred to 1 kg of waste treated in ERP.

The ERP was evaluated as the most environmentally friendly treatment plant from the EF approach. However, other environmental burdens as water consumption and air emissions signaled this treatment process as the most pollutant. On the other hand, unexpectedly, the CP obtained one of the worst evaluations from the EF and water consumption point of view. Nonetheless, the release of air emissions may also occur in other MSW treatment process, although this was not reported by LIPOR. Therefore, faired conclusions would be achieved if these data were also included.

8.4. Conclusions

In a first part of this chapter, a case study extracted from the literature was analyzed from an environmental point of view. The goal was to rank different alternatives for the treatment of MSW. The application of MCA methodologies was found to be a suitable way to integrate the information provided by a set of environmental criteria and to aid decision making. Also, EF proved to be a good screening indicator although it did not provide a comprehensive measure of environmental impacts associated to the waste treatment options considered. Besides, the ranking obtained was in agreement with the general hierarchy recommended by legislation, prioritizing treatment techniques that allow for energy or materials recovery. It must be noticed that inventory data suffered from certain gaps, e.g. the use of chemicals or waste water treatment was not included. A more detailed environmental evaluation or the extension of the system boundaries could lead to obtain a different ranking of alternatives.

MCA methods of different methodological nature were tested, all of them providing similar results except for ELECTRE III. In future research it is recommended to explore this disagreement by conducting a normalization of criteria previous to the analysis, for example. In this way, the influence of using criteria with different ranges and units would be avoided. Further, to carry out a proper selection of alternatives the incorporation of criteria from the other dimensions of sustainability (social en economic) should be considered. Nonetheless, the aim of MCA is to provide guidance in decision making and not irrefutable solutions.

The second part of the paper dealt with a real case scenario: the IWMS of LIPOR (Intermunicipal Waste Management of Greater Porto – Portugal). The system was composed of three waste treatment processes, namely multi-material valorization, organic valorization and energy valorization, completed by a sanitary landfill to dispose of refuse and waste previously prepared. All these processes complemented each other and the purpose of the study was not to select among them but to assess the environmental performance of LIPOR. The joint application of EF, MFA and LCA revealed that the composting plant presented the major contribution to environmental impact in relation to the amount of waste treated. Further, the energy recovery processes was found as very beneficial because of the low gross EF and the high CF due to the large quantity of electricity produced that is exported to the national network. Nonetheless, this result should be considered cautiously given the environmental impacts associated to air emissions released in the ERP and the higher water consumption rate with respect to other plants. The potential toxic effects of HF were found very relevant; therefore, measures to decrease the release of these emissions should be adopted. Therefore, the application of complementary environmental evaluation methods was found necessary and useful to properly evaluate the whole process.

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Annex A8. Inventory data for LIPOR (years 2007-2009)

Table A8.1. Inventory data of the Energy Recovery Plant (ERP).

Input flow	2007	2008	2009	Units
Non-differentiated waste	0	383,553	398,392	t
Chemicals				
Hydrated lime	4,331.3	4,000.7	4,337.2	t
Urea	1,531.1	1,539.9	1,808.4	t
Activated carbon	182.7	179.1	192.6	t
Caustic soda	0	43.1	38.5	t
Hydrochloric acid	0	44.8	44.5	t
Tripoliphosphate	0	17.4	0.7	t
Water				
Tap water	224,481	212,009	197,031	m^3
Water catchments	23,484	26,812	13,368	m^3
Energy				
Natural gas	2,372.63	5,640.81	4,624.75	GJ
Gasoil	56.51	89.68	92.96	GJ
Electricity	348	1009	3180	GJ
Air emissions				
HCl	8,630	6,562.2	6,830.1	kg
NO_x	286,000	222,898.4	264,165	kg
HF	433	145.1	325.4	kg
SO_2	8,570	9,397.9	9,482.3	kg
Particulates	4,140	2,532.4	2,563.2	kg
CO	8,559	9,736.1	16,219.8	kg
Dioxins and furans	$9.6\cdot10^{-6}$	$1.15\cdot10^{-5}$	$3.1\cdot10^{-6}$	kg
CO_2 equivalent	235,778	202,018	209,829	t
Sub-products				
Ash (to landfilling)	32,364	30,037	30,481.9	t
Slag (to landfilling)	76,606	72,798	75,871.8	t
Ferrous materials (recycling)	5,425	4,554	4,265.2	t

Table A8.2. Inventory data of the landfill associated to the ERP.

Input flow	2007	2008	2009	Units
Chemicals				
Sulfuric acid	9.07	18.99	9.77	t
Acetic acid	4.33	15.83	13.13	t
Soda	26.44	57.421	28.27	t
Antifoam	508	495	555	l
Water				
Tap water	5,347	4,944	3,122	m^3
Energy				
Gasoil	1,126.48	1,956.51	1,297.82	GJ
Electricity	1,495	1,496	1,728	GJ
Air emissions				
CO_2 equivalent	183,404	176,027	167,287	t

Table A8.3. Inventory data of the Composting Plant (CP).

Input flow	**2007**	**2008**	**2009**	**Units**
Separated organic wastes	0	37,146	42,215	t
Raw materials				
Pallet	49.95	92.75	106.8	t
Packaging plastic	11.05	7	2.16	t
Big bags	0.57	2.87	1.9	t
Water				
Tap water	12,725	18,082	20,596	m^3
Energy				
Natural gas	995.81	1,124.11	1,105.62	GJ
Gasoil	2,339.04	2,727.55	3,254.77	GJ
Electricity	11,522	12,885	16,550	GJ
Air emissions				
CO_2 equivalent	5,001	6,575	7,472	t
Sub-products				
Ferrous materials (recycling)	18.9	18.7	28.7	t
Light wastes (recycling)	475.5	554.8	645.6	t
Heavy wastes (landfilling)	1,041	233.6	350.9	t

Table A8.4. Inventory data of the Sorting Plant (SP).

Input flow	**2007**	**2008**	**2009**	**Units**
Separated recyclable wastes	0	55,470	59,966	t
Raw materials				
Wire	47.3	70.689	109.3	t
Water				
Tap water	514	386	376	m^3
Energy				
Natural gas	89.69	92.19	143.94	GJ
Gasoil	2,904.57	3,451.99	3,589.58	GJ
Electricity	1713	1957	1924	GJ
Refuse				
Paper and cardboard (to ERP)	1,211.5	1,095.9	719.4	t
Packages (to ERP)	580	431.7	489.1	t
Fines (to ERP)	455.9	580.6	541.4	t
Pre-sorting (to ERP)	1,048.6	1,136.6	1,291.7	t
WEEE (to ERP)	0	72.7	58	t
Particulates (to be made inert)	640	900	1,060	t

Table A8.5. Inventory data of general activities.

Input flow	**2007**	**2008**	**2009**	**Units**
Raw materials				
White paper	1,546.78	4,019	2,960	kg
White stationery	0	521	0	kg
Recycled paper	1,197.5	4,032	2,948	kg
Recycled stationery	0	7,173	1,646	kg
Water				
Tap water	2,813	2,756	2,435	m^3
Water catchments	19,463	17,912	24,702	m^3
Energy				
Natural gas	126.53	181.29	245.8	GJ
Gasoil	5,259.91	5,383.29	5,144.31	GJ
Gasoline	0	317.09	251.28	GJ
Electricity	2,937	3,003	3,139	GJ
Sup-products (managed by authorized companies)				
Refuse from platform	0	5,141.2	6,465.3	t
Mineral oil	7,400	10,200	9,300	l
Oily water	1,000	5,500	10,500	l
Cooking oil	375	755	1130	kg
Sawdust and cloth with oil	41.39	-	-	kg
Polluted containers	5,240	14,946.5	3,181	kg
Batteries	27,260	28,500	31,160	kg
Syringes	56,000	36,900	34,680	kg
Ink and toner	62.8	93	106.6	kg
Tires	580	1260	305	kg
Slag	47,097	36,737	56,516	t
Ferrous materials (recycling)	5,425	4,554	4,265	t

Concluding remarks

Conclusiones

Conclusións

Concluding remarks

Industry faces a major challenge to lead their processes and products towards a sustainable performance, given the increasing awareness on environmental issues of policy makers and the general public. As a consequence, the current positive attitude towards the implementation of solutions for pollution prevention and cleaner production is not just a response to emerging environmental laws and regulations, but also a matter of corporate responsibility and a key factor to be competitive in market. Metrics are required to measure environmental burdens, set goals, analyze trends, make comparisons among factories or for communication purposes.

This thesis poses a contribution to the development of environmental evaluation tools adapted to particular production sectors, aiming at providing metrics to guide decision making for the ecodesign of sustainable processes and products. Integrative frameworks that combine methodologies of different nature were proposed as the most suitable way to achieve comprehensive evaluations. At the same time, the simplicity of tools was pursued to make its application easier and more attractive for enterprises.

The following two core conclusions represent the common thinking-frame underlying in the specific conclusions drawn from this thesis:

- ✓ A wide variety of methodologies and indicators exist to perform environmental evaluation. Every indicator pays special attention to particular burdens (i.e. depletion of resources, impact categories like global warming or acidification, harmful effects to human health, etc.). Therefore, their application can result more or less appropriate depending on the characteristics of the case study or the goals established.
- ✓ The limited information provided by a single indicator can be sufficient in certain circumstances; however, integrative frameworks are require to achieve more comprehensive assessments of natural resources depletion, consequence of anthropogenic activities. The complexity of handling the information provided by different indicators at a time can effectively be addressed by applying Multi-criteria analysis (MCA) methodologies.

The specific conclusions of the thesis were:

- ✓ The Ecological Footprint (EF) was found suitable to be applied in the environmental evaluation of production processes. It properly accounted for energy (by means of the area required for the absorption of CO_2, thus including the Carbon Footprint) and materials consumption. However, when analyzing the results, the kind of metric that is provided by the EF must be looked carefully; thus, synthetic materials may obtain a better environmental evaluation than natural ones because of the area the latter require for their production. Its aggregative nature and the expression in units easy to be understood made the EF very appealing for communication purposes at a corporate level.
- ✓ During the application of EF to assess the environmental performance of a tailoring plant, certain drawbacks of the methodology were detected. Air emissions (apart from CO_2) or hazardous wastes could not be evaluated. In general, EF does not include any substance not meant to have a close cycle in nature. Although the integration with other environmental evaluation tools was proposed, it was also interesting the enhancement of an indicator with a recognized potential for measuring sustainability to achieve more comprehensive evaluations.
  - A methodology to estimate the EF of toxic and hazardous wastes (also suitable for non-hazardous ones) was developed considering a closed cycle modeled through a plasma process; a phenomenon that naturally occurs in stars and volcanoes. After testing it with the textile case study, it was observed that it provided reasonable results in the same order of magnitude than figures yielded by the conventional methodology for non-hazardous wastes.
  - The incorporation of greenhouses gases other than CO_2 and of acidifying emissions demonstrated that, by excluding these streams from EF assessments, the environmental impact was significantly being underestimated. Therefore, the principle underlying in conventional EF methodology of excluding substances with low assimilation capacities was found inappropriate and misleading.

- ✓ Although the application of global (i.e. world-average) factors in EF assessments was found necessary to be able to compare studies all over the world, it was also demonstrated that, by doing so, there is a lost of accuracy on estimates (especially for local problems). The CO_2 absorption factor estimated for Galicia (NW Spain) conveyed that the forests of this region presented a higher capacity to assimilate emissions than the world-average (although CO_2 effects are of global nature, European directives give relevance to regional capacities for carbon storage). The figure could be even higher given that carbon budgets in litter layer and soil were not considered. Therefore, EF figures would decrease as a consequence of sustainable forest management. The development of databases with regional conversion factors was found necessary. Considering the local context could result a key issue for decision making at corporate level.
- ✓ The re-evaluation of the tailoring factory incorporating the EF methodological improvements, as well as the combined use of EF with Energy and Material Flow Analysis (EMFA) and Life Cycle Assessment (LCA), yielded a much more comprehensive evaluation of the process than when EF was initially employed as stand-alone indicator. All inventory data could be assessed and more reliable conclusions could be extracted.
- ✓ The application of MCA methodologies proved to be useful for the integration of criteria that may or not be conflicting.
    - Fuzzy logic techniques enable the construction of a tool delivering an ecodesign index that incorporated the measure of energy and resource consumption (EF), environmental impact categories (LCA) and damage to human health (ERA). Handling a single number, but without losing information and scientific rigor, greatly benefited the work of the design team.
    - AHP, ELECTRE I and III, PROMETHEE I and II/GAIA methodologies conveyed a good behavior for the ranking of alternatives of municipal solid waste (MSW) treatment. Valuable guidance was provided to support decision making, although not for the identification of irrefutable solutions.
- ✓ Waste management was identified as a very complex task. The diversity of waste treatment alternatives, technologies, legal requirements, etc.

requires carrying out very exhaustive and accurate assessment to select the most adequate solution in accordance to the specific characteristics of the case study. In this thesis, the usefulness of MCA to aid this process was observed, although the inventory was not as complete as desirable. Therefore, future research should be focused on a more detailed analysis of the available waste treatment options, to expand the number of criteria considered (under the environmental approach) and to obtain more rigorous conclusions from the ranking of alternatives.

Conclusiones

La industria se enfrenta a un gran reto para dirigir sus procesos y productos hacia un buen comportamiento de sostenibilidad, dada la creciente concienciación social sobre aspectos ambientales tanto de los agentes políticos como del público en general. Como consecuencia, la actitud positiva actual hacia la implementación de soluciones de prevención y control de la contaminación y producción limpia no es sólo una respuesta a las leyes y regulaciones emergentes, sino también una cuestión de responsabilidad corporativa y un factor clave para ser competitivo en el mercado. Se necesitan indicadores que permitan medir las cargas ambientales, establecer objetivos, analizar tendencias, llevar a cabo comparaciones entre fábricas o para comunicar resultados.

Esta tesis supone una contribución al desarrollo de herramientas de evaluación ambiental adaptadas a sectores productivos particulares, con el objetivo de proporcionar medidas que guíen el proceso de decisión para el ecodiseño de procesos y productos sostenibles. Se propuso la combinación e integración de indicadores como la manera más eficaz de alcanzar evaluaciones globales y exhaustivas. Al mismo tiempo, se buscó mantener en lo posible la simplicidad de las herramientas para facilitar su aplicación y hacerla más atractiva a las empresas.

Las dos siguientes conclusiones generales representan el núcleo común de la filosofía que subyace en las conclusiones específicas extraídas de esta tesis:

- ✓ Existe una gran variedad de metodologías e indicadores para llevar a cabo evaluaciones ambientales. Cada indicador presta especial atención a cargas ambientales particulares (p.ej. agotamiento de los recursos, categorías de impacto como el calentamiento global o la acidificación, daños a la salud humana, etc.). Por tanto, su aplicación puede resultar más o menos apropiada dependiendo de las características del caso de estudio o de los objetivos establecidos.

- ✓ La información limitada proporcionada por un solo indicador puede ser suficiente en determinadas circunstancias; sin embargo, la definición de marcos de trabajo integrados se hace necesaria para alcanzar evaluaciones más exhaustivas del nivel de agotamiento de los recursos como consecuencia de las actividades antropogénicas. Las metodologías de Análisis Multi-criterio (AMC) pueden abordar de forma efectiva la complejidad que supone manejar la información proporcionada por varios indicadores al mismo tiempo.

Las conclusiones específicas de la tesis fueron:

- ✓ La Huella Ecológica (HE) resultó apropiada para ser aplicada en la evaluación ambiental de procesos productivos. Permitió evaluar de forma adecuada los consumos de energía (mediante el área necesaria para la absorción de CO_2, de forma que incluye a la Huella de Carbono) y el consumo de materiales. Sin embargo, a la hora de analizar los resultados, debe valorarse cuidadosamente el tipo de medida que ofrece este indicador; así, los materiales sintéticos pueden obtener una mejor evaluación ambiental que los de origen natural debido a las necesidades de territorio que estos últimos necesitan para su producción. Su naturaleza agregada y su expresión en unidades fácilmente entendibles hicieron que la HE resultara muy atractiva para ser utilizada en comunicación a nivel corporativo.
- ✓ Durante la aplicación de la HE para evaluar el comportamiento ambiental de una planta de confección se detectaron ciertas debilidades de la metodología, puesto que las emisiones atmosféricos (aparte del CO_2) y los residuos peligrosos no podían ser analizados. En general, la HE no incluye ninguna sustancia que no pueda presentar un ciclo cerrado en la naturaleza. Aunque se propuso su integración con otras herramientas de evaluación ambiental, también se consideró interesante contribuir a la mejora de la metodología de un indicador con reconocido potencial para medir la sostenibilidad.
  - Se desarrolló una metodología para estimar la HE de los residuos tóxicos y peligrosos (también válida para los no peligrosos) considerando un ciclo cerrado modelado mediante un proceso de plasma, fenómeno que ocurre de forma natural en los volcanes y las estrellas. Después de testearlo con el caso de estudio del sector textil, se comprobó que ofrecía resultados en el mismo orden de magnitud que cuando se aplicó la metodología convencional para los residuos no peligrosos.

- La incorporación de otros gases de efecto invernadero además del CO_2 y de emisiones acidificantes reveló que, al excluir estas sustancias de las evaluaciones de HE, se estaba subestimando de forma significativa el impacto ambiental. Por tanto, se encontró inapropiada la filosofía de la metodología tradicional que propone la exclusión de todas aquellas sustancias con bajas capacidades de asimilación.

✓ Aunque se valoró como necesario la aplicación de factores globales (basados en promedios mundiales) en las evaluaciones de HE para poder establecer comparaciones entre estudios llevados a cabo en distintas partes del mundo, también se demostró que, al hacerlo, se perdía precisión en las estimaciones (especialmente cuando se trata de problemas de índole local). La tasa de absorción de CO_2 estimada para Galicia (NO España) puso de manifiesto que los bosques de esta región presentaban una capacidad superior a la media mundial para asimilar estas emisiones (aunque los efectos que origina el CO_2 son de naturaleza global, las directivas europeas dan relevancia a las capacidades regionales para almacenar carbono). El valor obtenido podría ser incluso mayor puesto que en este trabajo no se consideró la absorción en el mantillo y suelo. Por tanto, los valores de HE podrían disminuir como consecuencia de una gestión sostenible de los bosques. Se identificó como necesaria la creación de bases de datos con factores de conversión regionales. Considerar el contexto local puede ser un factor clave para la toma de decisiones a nivel corporativo.

✓ La re-evaluación de la planta de confección incorporando las mejoras metodológicas de la HE, así como combinando el uso de este indicador con el Análisis de Flujos de Materia y Energía (AFME) y el Análisis de Ciclo de Vida (ACV) proporcionó una evaluación mucho más completa y exhaustiva del proceso que cuando se había utilizado sólo la HE. Se pudieron evaluar todos los datos de inventario, así como extraer conclusiones más fiables.

✓ La aplicación de metodologías de AMC resultó útil para la integración de criterios que pueden estar o no en conflicto.

- Las técnicas de lógica borrosa permitieron la construcción de una herramienta que devolvía un índice de ecodiseño que incorporaba medidas de consumo de energía y recursos (HE), categorías de impacto ambiental (ACV) y daño a la salud humana (ERA). La posibilidad de trabajar con un único índice, aunque sin perder información ni rigor científico, facilitó enormemente el trabajo del equipo de diseño.
- Las metodologías AHP, ELECTRE I and III, PROMETHEE I and II/GAIA mostraron su eficacia para la ordenación de alternativas de tratamiento de Residuos Sólidos Urbanos (RSU). Proporcionaron una guía valiosa para la toma de decisiones, aunque los resultados obtenidos no se deben tomar como soluciones irrefutables.

✓ Se identificó la gestión de residuos como una tarea de gran complejidad. La diversidad de alternativas de tratamiento existentes, tecnologías, requerimientos legales, etc., hace que sea necesario llevar a cabo una evaluación muy exhaustiva y precisa de cara a seleccionar la solución más adecuada de acuerdo con las características específicas del caso de estudio. En esta tesis se comprobó la utilidad del AMC en este ámbito, si bien el inventario de partida no era tan extenso y completo como sería deseable. Por tanto, las investigaciones futuras se deberían enfocar hacia una evaluación más en detalle de las opciones de tratamiento de residuos, para expandir el número de criterios considerados (bajo un enfoque ambiental) y poder extraer conclusiones más rigurosas de la ordenación de alternativas.

Conclusións

A industria encárase fronte a a un grande reto para dirixir os seus procesos e produtos cara un bo comportamento de sostenibilidade, dada a crecente concienciación social sobre aspectos ambientais tanto dos axentes políticos como do público xeral. Como consecuencia, a actitude positiva actual cara a implantación de solucións de prevención e control da contaminación e produción limpa non só é unha resposta ás leis e regulacións emerxentes, senón tamén unha cuestión de responsabilidade corporativa e un factor clave para ser competitivo no mercado. Precísanse indicadores que permitan medir as cargas ambientais, establecer obxectivos, analizar tendencias, levar a cabo comparacións entre fábricas ou para comunicar resultados.

Esta tese supón unha contribución ao desenvolvemento de ferramentas de avaliación ambiental adaptadas a sectores produtivos particulares, co obxectivo de proporcionar medidas que guíen o proceso de decisión para o ecodeseño de procesos e produtos sustentables. Propúxose a combinación e integración de indicadores como a maneira máis eficaz de acadar avaliación globais e exhaustivas. Ao mesmo tempo, se buscouse manter no posible a simplicidade das ferramentas para facilitar a súa aplicación e facela máis atractiva ás empresas.

As dúas seguintes conclusións xerais representan o núcleo común da filosofía que subxace nas conclusións específicas extraídas desta tese:

- ✓ Existe unha grande variedade de metodoloxías e indicadores para levar a cabo avaliacións ambientais. Cada indicador presta especial atención a cargas ambientais particulares (p.ex. esgotamento dos recursos, categorías de impacto como o quecemento global ou a acidificación, danos á saúde humana, etc.). Polo tanto, a súa aplicación pode resultar máis o menos apropiada dependendo das características do caso de estudo ou dos obxectivos establecidos.
- ✓ A información limitada proporcionada por un so indicador pode ser suficiente en determinadas circunstancias; sen embargo, a definición de marcos de traballo integrados faise precisa para acadar avaliacións máis exhaustivas do nivel de esgotamento dos recursos como consecuencia das actividades antropoxénicas. As metodoloxías de Análise Multi-criterio (AMC) poden abordar de forma efectiva a complexidade que supón

manexar a información proporcionada por varios indicadores ao mesmo tempo.

As conclusións específicas da tese foron:

- ✓ A Pegada Ecolóxica (PE) resultou apropiada para ser aplicada na avaliación ambiental de procesos produtivos. Permitiu avaliar de forma axeitada os consumos de enerxía (mediante a área precisa para a absorción de CO_2, de forma que inclúe á Pegada de Carbono) e o consumo de materiais. Sen embargo, á hora de analizar os resultados, debe valorarse coidadosamente o tipo de medida que ofrece este indicador; así, os materiais sintéticos poden obter unha mellor avaliación ambiental que os de orixe natural debido ao requirimentos de terra que estes últimos precisan para a súa produción. A súa natureza agregada e a súa expresión en unidades facilmente entendibles fixeron que a PE resultara moi atractiva para ser utilizada en comunicación a nivel corporativo.
- ✓ Durante a aplicación da PE para avaliar o comportamento ambiental dunha planta de confección detectáronse certas debilidades da metodoloxía, posto que as emisións atmosféricos (aparte do CO_2) e os residuos perigosos non podían ser analizados. En xeral, a PE non inclúe ningunha sustancia que non poida presentar un ciclo cerrado na natureza. Aínda que se propuxo a súa integración con outras ferramentas de avaliación ambiental, tamén se considerou interesante contribuír á mellora da metodoloxía dun indicador con recoñecido potencial para medir a sostenibilidade.
  - Desenvolveuse unha metodoloxía para estimar a PE dos residuos tóxicos e perigosos (tamén válida para os non perigosos) considerando un ciclo pechado modelado mediante un proceso de plasma, fenómeno que ocorre de forma natural nos volcáns e nas estrelas. Despois de testalo co caso de estudo do sector téxtil, comprobouse que ofrecía resultados na mesma orde de magnitude que cando se aplicou a metodoloxía convencional para os residuos non perigosos.

   - A incorporación de outros gases de efecto invernadoiro ademais do CO_2 e das emisións acidificantes revelou que, ao excluír estas sustancias das avaliacións da PE, estábase subestimando de forma significativa o impacto ambiental. Polo tanto, atopouse non apropiada a filosofía da metodoloxía tradicional que propón a exclusión de todas aquelas substancias con baixas capacidades de asimilación.

- ✓ Aínda que se valorou como necesaria a aplicación de factores globais (baseados en medias mundiais) nas avaliacións de PE para poder establecer comparacións entre estudos levados a cabo en distintas partes do mundo, tamén se demostrou que, ao facelo, perdíase precisión nas estimacións (especialmente cando se trata de problemas de índole local). A taxa de absorción de CO_2 estimada para Galicia (NO España) puxo de manifesto que os bosques desta rexión presentaban unha capacidade superior á media mundial para asimilar estas emisións (aínda que os efectos que orixina o CO_2 son de natureza global, as directivas europeas dan relevancia ás capacidades rexionais para almacenar carbono). O valor obtido podería ser incluso maior dado que neste traballo non se considerou a absorción no mantillo e o solo. Polo tanto, os valores de PE poderían diminuír como consecuencia dunha xestión sustentable dos bosques. Identificouse como necesaria a creación de bases de datos con factores de conversión rexionais. Considerar o contexto local pode ser un factor clave para a toma de decisións a nivel corporativo.
- ✓ A re-avaliación da planta de confección incorporando as melloras metodolóxicas da PE, así como combinando o uso deste indicador coa Análise de Fluxos de Materia e Enerxía (AFME) e a Análise de Ciclo de Vida (ACV) proporcionou unha avaliación moito máis completa e exhaustiva do proceso que cando se tiña empregado só a PE. Puidéronse avaliar todos os datos de inventario, así como extraer conclusións máis fiables.
- ✓ A aplicación de metodoloxías de AMC resultou útil para a integración de criterios que poden estar ou non en conflito.
   - As técnicas de lóxica borrosa permitiron a construción dunha ferramenta que devolvía un índice de ecodiseño que incorporaba medidas de consumo de enerxía e recursos (PE), categorías de

impacto ambiental (ACV) e dano á saúde humana (ERA). A posibilidade de traballar cun único índice, pero sen perder información nin rigor científico, facilitou enormemente o traballo do equipo de deseño.

- As metodoloxías AHP, ELECTRE I and III, PROMETHEE I and II/GAIA amosaron a súa eficacia para a ordenación de alternativas de tratamento de Residuos Sólidos Urbanos (RSU). Proporcionaron unha guía valiosa para a toma de decisións, aínda que os resultados obtidos non se deben tomar como solucións irrefutables.

✓ Identificouse a xestión de residuos como unha tarefa de grande complexidade. A diversidade de alternativas de tratamento existentes, tecnoloxías, requirimentos legais, etc., fai que sexa necesario levar a cabo unha avaliación moi exhaustiva e precisa de cara a seleccionar a solución máis axeitada dacordo coas características específicas do caso de estudo. Nesta tese comprobouse a utilidade da AMC neste ámbito, se ben o inventario de partida non era tan extenso e completo como sería desexable. Polo tanto, investigacións futuras deberían encamiñarse cara unha avaliación máis en detalle das opcións de tratamento de residuos, para expandir o número de criterios considerados (baixo un enfoque ambiental) e poder extraer conclusións máis rigorosas da ordenación de alternativas.

Appendix

Abbreviations and notation

Abbreviations

AHP	Analytical Hierarchy Process
ANP	Analytic Network Process
AP	Acidification Potential
BATs	Best Available Techniques
BBP	Benzyl butyl phthalate
BSI	British Standard Institute
CCN	Carrying Capacity Network
CF	Carbon Footprint
CFC	Chlorofluorocarbon
CML	Institute of Environmental Sciences, Leiden University
COD	Chemical Oxygen Demand
CP	Composting Plant (LIPOR)
CR	Cancer Risk
CSR	Corporate Social Responsibility
DAI	Dissipation Area Index
DB	Dichlorobenzene
DBP	Dibutyl phthalate
DEFRA	British Department for Environment, Food and Rural Affairs
DEHP	Di(2-ethylhexyl) phthalate
DINP	Di-isononyl phthalate
DMI	Direct Material Input
DNOP	Di-n-octyl phthalate
DPO	Domestic Process Output
DPSIR	Drivers-Pressures-State-Impacts-Responses
EEA	European Environmental Agency
EEI	Environmental Emission Index
EF	Ecological Footprint
EFTA	European Free Trade Association
EI	EcoIndicator
ELECTRE	Elimination and Choice Expressing Reality
EMC	Environmentally Weighted Material Consumption
EMFA	Energy and Material Flow Analysis
EP	Eutrophication Potential
ER	Ecological Rucksack
ERA	Environmental Risk Assessment
ERP	Energy Recovery Plant (LIPOR)

ES	Environmental Space
EU	European Union
EUROSTAT	Statistical Office of the European Communities
EUSES	European Union System for the Evaluation of Substances
FAET	Fresh water Aquatic EcoToxicity Potential
FU	Functional Unit
FWQ	Fuzzy Water Quality
GAIA	Geometrical Analysis for Interactive Aid
GDP	Gross Domestic Product
GFN	Global Footprint Network
GHG	Greenhouse gases
GRI	Global Reporting Initiative
GWP	Global Warming Potential
HANPP	Human Appropriation of Net Primary Production
HQ	Hazard Quotient
HTP	Human Toxicity Potential
IA	Integrated Assessment
ICAC	International Cotton Advisory Committee
IDEA	Spanish Institute for Energy Diversification and Saving
ifeu	Institute for Energy and Environmental Research Heidelberg Ltd.
IFN	Spanish National Forest Inventory
IFSRA	Integrated Fuzzy-Stochastic Risk Assessment
ifu	Institute for Environmental Informatics Hamburg GmbH
IGE	Galician Statistics Institute
INE	National Statistics Institute
IPCC	Intergovernmental Panel on Climate Change
IPP	Integrated Product Policy
IPPC	Integrated Pollution Prevention and Control
ISO	International Organization for Standardization
IWMS	Integrated Waste Management System
JRC	Joint Research Center
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
LEAC	Land and Ecosystem Accounts
LIPOR	Intermunicipal Waste Management of Greater Porto (Portugal)
LPG	Liquefied petroleum gas
MACBETH	Measuring Attractiveness by a Categorical Based Evaluation TecHnique
MADM	Multiple-Attribute Decision-Making
MAETP	Marine Aquatic EcoToxicity Potential

MAUT	Multi-Attribute Utility Theory
MCA	Multi-Criteria Analysis
MFA	Material Flow Analysis
MIPS	Material Input Per unit Service
MODM	Multiple-Objective Decision-Making
MSW	Municipal Solid Waste
NAIADE	Novel Approach to Imprecise Assessment and Decision Environments
NFA	National Footprint Accounts
ODP	Ozone layer Depletion Potential
OECD	Organization for Economic Co-operation and Development
OFMSW	Organic Fraction of Municipal Solid Waste
ORNL	Oak Ridge National Laboratory
POCP	Photochemical Oxidation Potential
PROMETHEE	Preference Ranking Organization Method for Enrichment Evaluation
PTB	Physical Trade Balance
RAIS	Risk Assessment Information System
REACH	Registration, Evaluation, Authorization and Restriction of Chemical
SAFE	Sustainability Assessment by Fuzzy Evaluation
SEN	Spanish Electricity Network
SFA	Substance Flow Analysis
SP	Sorting Plant (LIPOR)
SPI	Sustainable Process Index
TBL	Triple Bottom Line
TETP	Terrestrial EcoToxicity Potential
UN	United Nations
US EPA	United States Environmental Protection Agency
USES	Uniform System for the Evaluation of Substances
WCED	World Commission on Environment and Development
WF	Water Footprint
WHO	World Health Organization

Notation

Chapter 2

Ecological Footprint

A_{ik}	Area required for the component *i* belonging to the main category *k*
A_{ik}	Single elements included in the main category *k*
CF	Counter Footprint
EF	Ecological footprint of the process due to the annual performance
EF_r	Relative Ecological Footprint
EP_i	Energy productivity for element *i*
EV_i	Embodied energy for element *i*
F_j	Equivalence factor for land type *j*
gha	Global hectares
gm^2	Global m^2
NEF	Net Ecological Footprint
NP_i	Natural productivity for element *i*
P_{yr}	Annual production rate

Life Cycle Assessment

A_s	Amount of emission *s* released
C_{nt}	Normalized value for the category *t*
C_{st}	Contribution of the emission *s* to the category *t*
C_t	Global value of the category *t*
N_t	Normalization factor for the category *t*
W_{st}	Characterization factor for the emission *s* within the category *t*

Environmental Risk Assessment

CR	Cancer Risk
Dose	Exposure dose to a chemical
HI	Hazard Index
HQ	Hazard Quotient
HQ_p	Hazard Quotient for a pathway *p*
RfD	Reference Dose
SF	Slope Factor

Multi-criteria analysis (general)

a_i	Alternative
A	Set of alternatives
F	Set of criteria
g_j	Criteria
$g_j(a_i)$	Evaluation of criterion g_j for alternative a_i
J	Set of criteria indices
k	Number of criteria
N	Number of alternatives
w_j	Weight
W	Weight vector modeling the preferences of the decision maker

AHP

CI	Consistency index
CR	Consistency ratio
M	Number of elements at a given level of the hierarchy
P	Intensity of preference
RI	Randomly generate index
T	Matrix dimension
λ_{max}	Maximum eigenvalue

ELECTREE

a, b	Alternatives
B_l	Ordered classes belonging to Z_1
C_{ab}	Global concordance index for a pair of alternatives (a,b)
c_j	Concordance index
dc_j	Discordance index
E	Number of classes belonging to Z_2
G_e	Ordered classes belonging to Z_2
L	Number of classes belonging to Z_1
p_j	Preference thresholds
q_j	Indifference thresholds
S	Binary outranking relations
v_j	Veto threshold
Z	Partial pre-order of alternatives
Z_1/Z_2	Descending/ascending pre-order of alternatives
$\rho(a,b)$	Credibility index

PROMETHEE

I^{I}	Indifference in partial outranking of PROMETHEE I
I^{II}	Incomparability in partial outranking of PROMETHEE II
P	Preference function
p	Threshold of strict preference
P^{I}	Preference in partial outranking of PROMETHEE I
P^{II}	Indifference in partial outranking of PROMETHEE II
q	Threshold of indifference
R^{I}	Incomparability in partial outranking of PROMETHEE I
s	Intermediate value between q and p
$\pi(a,b)$	Expresses with which degree a is preferred to b over all criteria
Φ	Net outranking flow in PROMETHEE II
ϕ^{-}	Negative outranking flow
ϕ^{+}	Positive outranking flow

Chapter 4

C	Carbon content
CF_{slag}	Counter Footprint associated to slag production
E_A	Electricity consumed by the auxiliary operational units
E_C	Electricity generated in the combined cycle
$EF_{carbon\ emissions}$	Area required for absorption of CO_2 emissions
$EF_{electricity}$	Ecological Footprint from net electricity balance
EF_{wastes}	Ecological Footprint of wastes
E_N	Net external electricity demand
E_P	Electricity consumed by the plasma torch

Chapter 5

A_{Acid}	Area for the absorption of acidifying emissions
A_{GHG}	Area for the absorption of GHG emissions
B	Annual energy consumption
ba	Basal area
BEF	Biomass expansion factor
C	Carbon content in wood
C_{clay}	Carbon content in clay
d	Diameter at breast height
$Efac$	Emission factor
h	Height of tree

N	Number of trees
P	Bricks annual production
r_C	Carbon absorption rate
r_{CO2}	Carbon dioxide absorption rate
S	Wooded area
t	Time
V_{cc}	Bark volume
W_t	Total biomass

Chapter 7

AdhF	Adherence factor
BW	Mean body weight
CS	Concentration of contaminant in shoes
CT	Contact time
CW	Concentration in water
DAF	Dermal absorption factor
Derms	Estimated daily dose of exposure due to dermal contact
DerRefD	Dermal reference dose
DerSF	Dermal slope factor
Dose	Dose of exposure to a contaminant due to ingestion
FEcoDI	Fuzzy Ecodesign Index
GIAB	Gastrointestinal absorption factor
OralRefD	Oral reference dose
OralSF	Oral slope factor
SAfeet	Mean surface area for feet
WIF	Water intake factor

Outputs of the Thesis

Journal papers

Herva, M., Franco, A., Fdez-Carrasco, E., Roca, E., 2008. The ecological footprint of production processes as indicator of sustainability (in Spanish). Ingeniería Química 460, 180-186.

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Herva, M., Franco, A., Roca, E., Ferreiro, S., Álvarez, A., 2007. An approach for the application of the ecological footprint as environmental indicator in the textile sector. International Ecological Footprint Conference. New Developments in Ecological Footprint Methodology, Policy & Practice. Cardiff (United Kingdom), 8-10 May. CD with proceedings. ISBN: 978-1-904393-90-0.

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Book chapters

Roca, E., Herva, M., 2009. Biofuels: are we in the way towards energetic sustainability? In: Biofuels: challenges and opportunities, pp. 267-280 (in Spanish). ISBN: 978-84-613-4672-1. Santiago de Compostela (Spain).

Technological transfer to industry

- Member of the team in charge of developing a simplified tool, integrating Ecological Footprint and LCA according to ISO 14040 standards, for environmental evaluation in the textile sector. This technological transfer took place in the framework of the research project PGIDIT 05TAM023E. Client: INDITEX.
- Member of the team in charge of developing a tool for the ecodesign of footwear based on the integration of different environmental and sustainability indicators. This technological transfer took place in the framework of the research project PGIDIT 07MDS050E. Client: INDITEX.
- Member of the research team that collaborated in the development of the Manual for the design, construction, maintenance and management of the eco-efficient store. This technological transfer took place in the framework of the research project PGIDIT 08MDS030E. Client: INDITEX.