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MUNICIPAL SOLID WASTE MANAGEMENT SYSTEM: DECISION SUPPORT THROUGH SYSTEMS ANALYSIS

Thesis submitted to the Universidade Nova de Lisboa, Faculdade de Ciências e Tecnologia for the degree of Doctor of Philosophy in Environmental Engineering.

LISBOA

2010

This thesis has been supported by *Fundação para a Ciência e Tecnologia* (Foundation for Science and Technology), national funds from *Ministério da Ciência, Tecnologia e Ensino Superior* (Ministry of Science, Technology and Higher Education), ref. SFRH/BD/27402/2006.

ACKNOWLEDGES

This work would not have been possible without the help and involving of the people to whom I wish to express my sincere gratitude:

- Prof. Graça Martinho and Prof. Ni-Bin Chang, my advisor and co-advisor, for the continuous guidance, stimulus, trust and support that promoted my scientific growth and made this work possible.

- Eng. João Pedro Rodrigues, Eng. Artur Cabeças, Eng. Ana Martins and Eng. Francisco Bento dos Santos for providing all the information that I needed for my thesis;

- Eng. Eduarda Gomes and Eng. Pedro Rodrigues from Moita municipality, Eng. Paula Pinto from Almada municipality, to city councilman Nuno Canta from Montijo municipality, Eng. Cátia Aleixo from Palmela municipality, Eng. Jorge Didelet from Seixal municipality and Mrs. Natália Valério from Alcochete municipality, for all the attention and care for the data required, and the fastness answering to my requests;

- Eng. Ingrid Falcão, for the explanation about composites recycling and its mass balance;

- Eng. Alexandra Silva, for the explanation about EPS recycling and its mass balance;

- Mr. Carlos Alberto Alves from Extruplás company who have contributed with mass balance from his units;

- Eng. Miguel Nunes, from ERSAR, where its explanations were valuable to unsure quality into my work;

- Mrs. Sara Vicente, which without her mathematical knowledge it would be difficult to process all the information and conduct my thesis;

- Eng. Pedro Simões, for the data and knowing paper/cardboard recycling processes;

- Eng. José Eduardo Costa, for the explanations and data provided for ferrous and non-ferrous metals recycling and virgin production;

- Eng. Peter Müller-Beilschmidt, for its knowledge and technical support during LCA construction;

- A thanking thought is also due to DCEA/FCT/UNL, where I started and have always been working, which made this thesis possible.

- At last but not the least, I would like to thank to my parents and my sister for all the patient and for being so understanding with my absence.

- To João, who in the last leaps of my thesis has been my third arm, I want to thank for the advices, constant encouragement and strong support that he always gave me, helping me in stressful and anxious moments.

- At the end of my acknowledgements and being aware that many people have directly or indirectly contributed to this work, I would like to apologize all those that, for some reason, I have forgotten to thank. Thank you all.

ABSTRACT

The present study intends to show the development of systems analysis model applied to solid waste management system, applied into AMARSUL, a solid waste management system responsible for the management of municipal solid waste produced in Setúbal peninsula, Portugal. The model developed intended to promote sustainable decision making, covering the four columns: technical, environmental, economic and social aspects.

To develop the model an intensive literature review have been conducted. To simplify the discussion, the spectrum of these systems engineering models and system assessment tools was divided into two broadly-based domains associated with fourteen categories although some of them may be intertwined with each other. The first domain comprises systems engineering models including cost-benefit analysis, forecasting analysis, simulation analysis, optimization analysis, and integrated modeling system whereas the second domain introduces system assessment tools including management information systems, scenario development, material flow analysis, life cycle assessment (LCA), risk assessment, environmental impact assessment, strategic environmental assessment, socio-economic assessment, and sustainable assessment.

The literature performed have indicated that sustainable assessment models have been one of the most applied into solid waste management, being methods like LCA and optimization modeling (including multicriteria decision making(MCDM)) also important systems analysis methods. These were the methods (LCA and MCDM) applied to compose the system analysis model for solid waste.

The life cycle assessment have been conducted based on ISO 14040 family of norms; for multicriteria decision making there is no procedure neither guidelines, being applied analytic hierarchy process (AHP) based Fuzzy Interval technique for order performance by similarity to ideal solution (TOPSIS). Multicriteria decision making have included several data from life cycle assessment to construct environmental, social and technical attributes, plus economic criteria obtained from collected data from stakeholders involved in the study.

The results have shown that solutions including anaerobic digestion in mechanical biological treatment plant plus anaerobic digestion of biodegradable municipal waste from source separation, with energetic recovery of refuse derived fuel (RDF) and promoting pays-as-you-throw instrument to promote recycling targets compliance would be the best solutions to implement in AMARSUL system. The direct burning of high calorific fraction instead of

RDF has not been advantageous considering all criteria, however, during LCA, the results were the reversal. Also it refers that aerobic mechanical biological treatment should be closed.

Uncertainty and reliability results performed to LCA could ensure data quality and also procedure taken to perform the analysis. Also fuzzy interval MCDM applied to include uncertainty on the multicriteria decision making has shown as a good tool to verify the results obtained.

The use of multicriteria decision making for support decision making process is vital to guarantee the best solution which, otherwise, could not be found when looking at just one criterion at the time. Such is due when alternative results are the same or too approximate in several criteria, including life cycle assessment results. The inclusion of these results into multicriteria decision making was successful to reach the one best solution.

In the future, the main important developments to be promoted to improve the systems analysis method applied are the development of studies concerning social behavior that could be integrated into the system, but also studies on waste production phase, since its inclusion could bring important conclusions on waste prevention. Other combinations of systems engineering models and system assessment tools could be promoted, to assess their ability compared with the one developed in this study.

SUMÁRIO

O presente trabalho consistiu no desenvolvimento de um modelo de análise de sistemas aplicado à gestão de resíduos, no sistema AMARSUL, um sistema de gestão de resíduos sólidos urbanos responsável pelos resíduos da península de Setúbal, Portugal. O modelo apresentado pretendeu promover o apoio à decisão sustentável, de modo a cobrir os respectivos quatro pilares: técnico, ambiental, económico e social.

De modo a desenvolver o modelo procedeu-se a uma intensiva revisão da literatura sobre modelos de gestão. Para simplificar a discussão, os modelos de análise de sistemas foram divididos em dois grupos – modelos de engenharia de sistemas e ferramentas de avaliação de sistemas, tendo-lhes sido associados catorze categorias, estando algumas delas interligadas entre si. O primeiro grupo compreende as categorias análise custo-benefício, análise de previsão, simulação, optimização e sistemas de modelação integrados; o segundo grupo é referente a ferramentas como sistemas de gestão de informação, desenvolvimento de cenários, análise de fluxos de materiais, análise do ciclo de vida (ACV), análise de risco, estudo de impacte ambiental, avaliação ambiental estratégica, avaliação sócio-económica e avaliação sustentável.

A revisão da literatura mostrou que os métodos de avaliação sustentável têm sido dos mais aplicados na gestão de resíduos, sendo os métodos de análise do ciclo de vida (ACV) e a optimização (que inclui a análise multi-critério (AMC)) de considerável aplicação. Nesta tese, os modelos aplicados para desenvolver o modelo de análise de sistema para a gestão de resíduos sólidos foram a ACV e a AMC.

A análise do ciclo de vida foi desenvolvida através da aplicação da família de normas ISO 14040; para a análise multicritério não existem normas nem orientações, mas a aplicação dos modelos de análise multicritério têm um procedimento definido. Neste caso foram aplicados o processo hierárquico analitico baseado em *technique for order performance by similarity to ideal solution* (TOPSIS) de intervalo de números difusos. A análise multicritério incluiu os resultados da ACV para construir os critérios ambientais, sociais e técnicos, sendo os critérios económicos obtidos utilizando dados pesquisados e fornecidos pelas empresas envolvidas.

Os resultados da aplicação do modelo desenvolvido ao caso de estudo da AMARSUL revelou que, para este sistema, a melhor solução é a que inclui a digestão anaeróbia em tratamento mecânico e biológico e digestão anaeróbia de resíduos urbanos biodegradáveis de recolha selectiva, com a valorização energética de combustível derivado de resíduos (CDR)

em incineradora e a promoção do instrumento *pays-as-you-throw* de modo a permitir o cumprimento das metas de reciclagem de resíduos de embalagens. A queima directa da fracção altamente calorífica sem produçao de CDR não se mostrou a mais vantajosa considerando a totalidade dos critérios, no entanto, durante a ACV, os resultados foram o inverso. Outra recomendação apontada pelo modelo desenvolvido consiste no encerramento da unidade de tratamento mecânico e biológico por via aeróbia.

A análise de incerteza e de sensibilidade, desenvolvidas para a análise do ciclo de vida, podem garantir a qualidade dos dados utilizados e ainda os procedimentos e pressupostos considerados para desenvolver a análise. Igualmente a AMC através de número difusos em intervalo conseguem incluir a incerteza na tomada de decisão, sendo uma correcta ferramenta para obter a melhor solução em situações de grande incerteza.

A aplicação da análise multicritério é vital para garantir a melhor solução que, de outro modo, não seria encontrada quando se observa apenas um critério de cada vez. Tal deve-se ao facto dos resultados obtidos para as diferentes alternativas em estudo serem iguais ou muito aproximados em vários critérios, incluindo os resultados da ACV. Torna-se, por isso, útil a inclusão dos resultados da ACV na AMC para se conseguir a melhor alternativa de gestão dos resíduos.

No futuro, os principais desenvolvimentos a serem desenvolvidos para melhorar os modelos de análise de sistemas aplicados são o desenvolvimento de estudos relativos ao comportamento social, de modo a poderem ser integrados nos modelos, bem como estudos relativos à fase da produção de resíduos, uma vez que a inclusão desta etapa no modelo desenvolvido poderá contribuir para a tomada de decisões que envolvam a prevenção de resíduos. Outras integrações de modelos de engenharia de sistemas e de ferramentas de avaliação de sistemas devem ser promovidas, de modo a avaliar as suas capacidades quando comparadas com a combinação de métodos realizada neste trabalho.

ABBREVIATIONS

ADEME	Agence de l'Environnement et de la Maîtrise de l'Energie
AHP	Analytic hierarchy process
ANP	Analytical network process
ARIMA	Auto-regressive integrated moving average
AWAST	Aid in the management and European comparison of a municipal solid waste treatment for a global and sustainable approach
BANANA	Built absolutely nothing anywhere near anyone
BATNEEC	Best available techniques not entailing excessive cost
BPEO	Best protection environmental option
BMW	Biodegradable Municipal Waste
CBA	Cost-Benefit Analysis
CERA	Comparative Environmental Risk Assessment
DEMATEL	Decision making trial and evaluation laboratory
DMS	Data management system
DP	Dynamic Programming
DPSIR	Driving–forces-pressures-state-impact-responses
DSD	Duales System Deutschland
DSS	Decision Support Systems
DST	Decision support tool
EASEWASTE	Environmental assessment of solid waste systems and technologies
EC	European Community
EDX/EDI	Electronic data exchange
EEA	European Environment Agency
EEC	European Economic Community
EFA	Energy from analysis
EHHRA	Environmental and human health risk assessment
EIA	Environmental Impact Assessment
EIONET	European environment information and observation network
EIS	Environmental impact statement
ELECTRE	elimination and choice translating algorithm
EMS	Environmental Management systems
EPR	Extended Producer Responsibility
ERA	Environmental Risk Assessment
ES	Expert System
ESO	evolutionary simulation-optimization

EU	European Union
EUDIN	European Data Interchange of Waste Notification System
FCVM	Fuzzy contingent valuation method
FIMOMIP	Fuzzy interval multiobjective mixed integer programming
FM	Forecasting Models
GHG	Greenhouse gas
GIGO	Garbage in, garbage out
GIP	Grey Integer Programming
GIS	Geographic Information System
GWP	Global warming potential
IMS	Integrated Modeling System
IOA	Input-Output Analysis
IP	Integer programming
IPPC	Integrated pollution prevention control
ISWM	Integrated Solid Waste Management
IVF	Interval-valued fuzzy
IWM	Integrated Waste Management
LATS	Landfill Allowance Trading System
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	life cycle impact assessment
LP	Linear Programming
LULU	Locally unacceptable land use
MAUT	Multiple attribute utility theory
MBMS	Model base management system
MCDM	Multicriteria decision making
MCEA	Modified cost-effectiveness analysis
MFA	Material Flow Analysis
MGA	modeling-to-generate-alternatives
MILP	Mixed-integer linear programming
MIP	Mixed-Integer Programming
MIMES/WASTE	Model for description and optimization of integrated material flows and energy systems
MIS	Management Information System
MOP	Multi-objective programming
MRFs	Material recovery facilities
MSW	Municipal Solid Waste

NAIADE	Novel approach to imprecise assessment and decision environments
NIMBY	Not in my backyard
NLP	Non-Linear Programming Model
OM	Optimization Models
ORWARE	ORganic WAste Research
PAYT	Pay-As-You-Throw
PET	Polyethylene terephthalate
PIPA	Policy impact potential analysis
PROMETHÉE	Preference ranking organization method for enrichment evaluation
QAS	Quality Assurance System
RA	Environmental and ecological risk assessment
RDF	Refuse derived fuels
ROME	Reasonable maximum exposure
RRPLAN	Resource recovery planning
SA	Sustainability Assessment
SAW	Simple additive weighting
SCOLDSS	Decision support system applied to the operational planning of solid waste collection systems
SD	Scenario Development
SDS	Sustainable Development Strategy
SEA	Strategic Environmental Assessment
SFA	Substance Flow Analysis
SFINX	Substance flow inter-nodal exchange
SLCC	social life cycle costs
SM	Simulation Models
SoEA	Socio-economic assessment
STAN	subSTance flow Analysis
SOP	Single objective programming
SWIFT	Simplex with forcing trials
SWIM	Solid Waste Integrated Model
SWM	Solid Waste Management
TASAR	Tool for Analyzing Separation Actions and Recovery
TOPSIS	Technique for order preference by similarity to an ideal solution
TSP	two-stage stochastic programming
TV	Television
USEPA	United States Environmental Protection Agency
WAMED	waste managements’ efficient decision

WASTED	Waste Analysis Software Tool for Environmental Decisions
WEEE	Waste electrical and electronic equipment
WHP	Waste Hierarchy Principle
WISARD	Waste-Integrated Systems for Assessment of Recovery and Disposal
WRAP	Waste Resources Allocation Program
WRATE	Waste and Resources Assessment Tool for the Environment
XML	Extensible Markup Language

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CHAPTER I – INTRODUCTION

1 INTRODUCTION

1.1 BACKGROUND

Until 1997, municipal solid waste (MSW) in Portugal was based on collection and landfilling of waste in dumpsites. The municipalities were responsible for the collection of waste and each municipality has its own dumpsite. The first Waste Framework Law ("Lei Quadro dos Resíduos, Decreto-Lei n.º 488/85, de 25 de Novembro" (PCM, 1985)) was only published in 1985 (Rodrigues and Martinho, 2007).

With the entrance in European Economic Community in 1986, Portugal have the financial and legislation support to improve environmental protection, including solid waste management (SWM). During the following 10 years, the main advance has been focused on regulation.

According to Pássaro (2003), waste management was only considered a priority in the 1990s. As result, in 1995, 26% of the MSW generated in Portugal have assured the proper treatment, while the remainder was disposed in more than 340 dumps. On the other hand, selected collection for recycling was not very common, being conducted only in metropolitan municipalities, and even then only for paper and glass. For glass, municipalities and glass producers have had the initiative of collect packaging glass, having started in 1983 (Martinho and Rodrigues, 2007). For paper/cardboard, a parallel market existed before starting the selective collection in end 80's decade, provided by Torres Vedras, Tomar, Beja, Oeiras, Faro, Seixal e Almada cities (Martinho and Rodrigues, 2007).

In 1997, the publishing of Strategic Plan for Municipal Solid Waste (PERSU-Plano Estratégico para a Gestão dos Resíduos Urbanos)" (INR, 1999) have promoted the introduction in several waste treatment technologies like composting plants, two incineration plants (Lisbon and Porto metropolitan areas) and sanitary landfills. The closure of uncontrolled dumpsites plus the establishment of recycling targets were also measures included in PERSU. In the case of packaging waste, due to publishing of 94/62/EC (EC, 1994), was created the first manager entity waste Green Dot System ("Sociedade Ponto Verde", in Portuguese). It was created in 1996, having the license one year later (Martinho and Rodrigues, 2007).

From the measures predicted in PERSU, the best established was the closure of dumpsites. In 2002, 100% of MSW generated was disposed appropriately and almost 70% of national territory was covered by selected collection programs of recycling (Pássaro, 2003). Concerning infrastructures, Portugal have three incineration plants, seven composting plants (for MSW and green waste), 37 landfills, 77 transfer stations, 18 sorting units, 133 ecocenters and 13,500 packaging bring systems, named “ecopontos”. According to Magrinho et al. (2006), several MSW management systems were created and came legally into force in order to accomplish the above plan. Such entities were created by public financing, joining several municipalities together in order to facilitate the implementation of the major part of the predicted projects. In 2003, 30 MSW management systems existed in Portugal (not including Azores and Madeira islands) that covered the entire national continental territory. This was the beginning of integrated solid waste management (ISWM) in Portugal.

The ISWM management in Portugal divided the responsibilities through different entities, namely municipalities, MSW management systems, and management entities, like SPV. According to Magrinho et al. (2006), these entities guide their activities according to the legislation and the policies dictated by the Ministry of the Environment, Spatial Planning and Regional Development. MSW management activities are controlled and supervised by the Portuguese Environment Agency (APA – “Agência Portuguesa do Ambiente”).

Besides all the progress accomplished, new challenges has been brought by European Union (EU) to improve MSW management system. Packaging Waste Directive 94/62/EC (EC, 1994) and its alterations in 2004/12/EC Directive (EC, 2004) would impose the recycling targets for materials glass, metals, paper/cardboard, plastic and wood, as is shown in Table 1.1.

Table 1.1 Targets from Landfill and Packaging Waste Directives

Legal reference	Targets
Decreto-Lei n.º 366-A/97, de 20 de Dezembro (changed by Decreto-Lei n.º 162/2000, de 27 de Julho and by Decreto-Lei n.º 92/2006, de 25 de Maio), which transpose to national law Directive n.º 94/62/EC (and related changes)	Portuguese targets to comply for packaging waste in 2011: • total recovery: > 60% • total recycling: 55-80% • glass recycling: > 60% • paper/cardboard recycling: > 60% • plastic recycling: > 22,5% • metals recycling: > 50% • Wood recycling: > 15%
Decreto-Lei n.º 152/2002, de 23 de Maio which transpose to national law Directive n.º 1999/31/EC, concerning landfill of waste	• Portuguese target to comply 2013: biodegradable municipal waste going to landfills must be reduced to 50% of the total amount (by weight) of biodegradable municipal waste produced in 1995; • Portuguese target to comply 2020: biodegradable municipal waste going to landfills must be reduced to 35% of the total amount (by weight) of biodegradable municipal waste produced in 1995

In Landfill Waste Directive 99/31/EC (EC, 1999), biodegradable municipal waste (BMW) deviation targets have been proposed for 2006, 2009 and 2016. Due to the delay on infrastructures construction to treat BMW, the four-year derogation available in the Landfill Waste Directive has been applied. The deviation targets are now defined 2013 and 2020. The targets defined by those Directives have brought forward the application of waste hierarchy.

In this respect, a second version of PERSU (MAOTDR, 2007) was published in 2007 to answer to significant present and near future challenges of reaching Packaging Waste and Landfill Directives targets.

The actual results from PERSU II are the several units of mechanical biological treatment in construction, being the overall of waste treatment technologies existing presented in Table 1.2.

Table 1.2 Waste management infrastructures and equipments in continental Portugal

Infrastructures and equipments	In exploration	Predicted
Organic waste treatment plants (aerobic MBT, BMW composting plants and one	9	15
Incineration plants	2	0
Sanitary landfills	34	8
Transfer stations	81	0
Sorting plants	29	4
Ecocenters	189	1
Packaging waste bring systems (“Ecopontos”)	31,068	0

However, new challenges have been brought into MSW management systems. The New Waste Framework Directive 2008/98/EC (EC, 2008) have contributed with a new definition for waste, by-products and end-of-waste, which have resulted in the need of choosing appropriate technologies that aim to improve the protection of human health and environment, promoting re-use and recycling, enhancing waste prevention programs via organic waste separate collection, and implementing extended producer responsibility (EPR) collectively. Also, key challenges related to long term SWM like climate change and energy, and sustainable consumption and production related to waste prevention programs have been received broad attention by SWM system stakeholders. This new waste management paradigm needs to be included nowadays in waste management systems, since PERSU II have not focused on such regulation.

Besides the new regulation changes presented, MSW technical issues are raising. The waste production growing in the last five years, shown in Fig. 1.1 represent a needed effort to induce measures to promote waste prevention (APA, 2010).

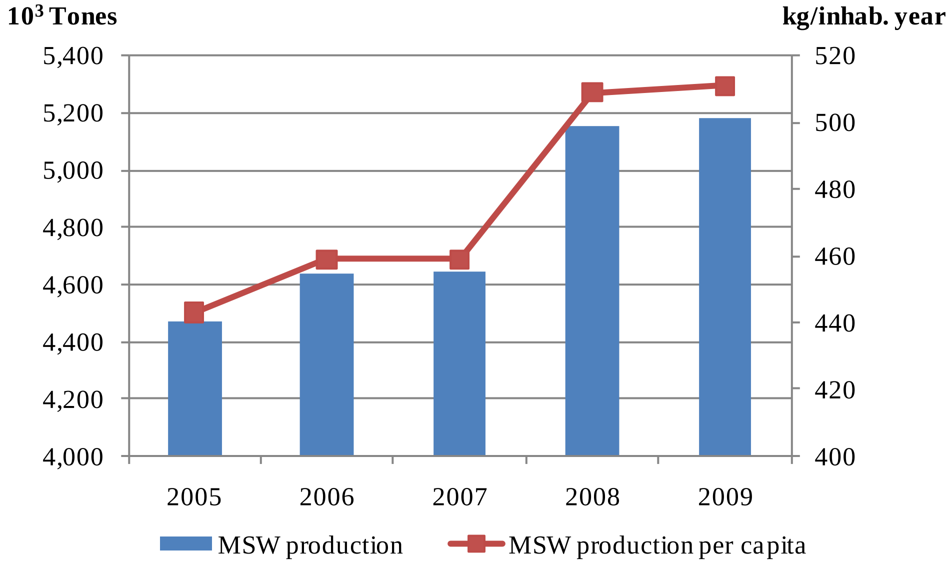

Fig. 1.1 MSW production: total and per capita between 2005 and 2009

The MSW production in Portugal is heterogeneous in Continental Portugal, existing higher productions in major cities (Lisbon and Porto) instead of rural areas (Alentejo mainly), like is present in Fig. 1.2 (APA, 2010).

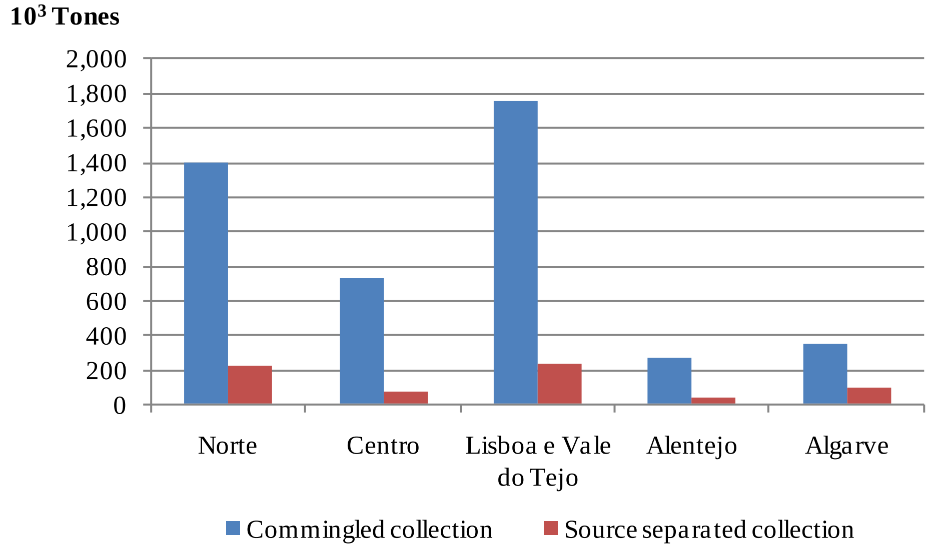

Fig. 1.2 MSW collected commingled and source separated, by region, in 2009

Regarding the waste treatment provided actually, in Fig. 1.3 is visible that landfill has been the main end of waste life, what is not in accordance with the new challenges from New Waste Framework Directive (APA, 2010).

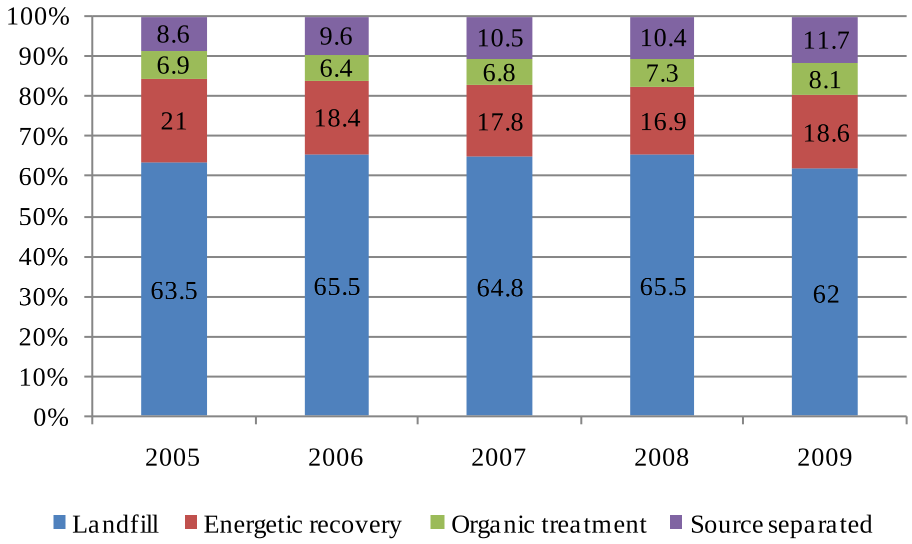

Fig. 1.3 Waste treatment destinies in Portugal

From an environmental point of view, the environmental protection and actual problematic had taken to an increase in efforts to minimize environmental impact from waste management. If in 90's decade the main problem in Portugal was open dumpsites, nowadays the main focus is to extend waste life cycle, which will increase the responsible use of resources, and is to contribute to reduce impact on climate change from sanitary landfills mainly.

Focusing on actual economic aspects, Continental Portuguese MSW management systems have changed considerably to improve economic efficiency of managing waste, to optimize economically the waste management system. The reduction of number of systems from 30 to 25 is a consequence of economic optimization, being the actual MSW management systems presented in Fig. 1.4 (APA, 2009).

Also, according to MAOTDR (2007), Portuguese MSW production is still highly dependent from consumption and economy growing, which is in agreement with a sustainable management of waste. It should also be mentioned the actual problem of waste user charges, where municipalities does not transfer to the citizen the effective costs from MSW management system, internalizing, in several cases, a considerable portion of the charge collected my waste management systems (MAOTDR, 2007). Such is not in accordance with pollution pays principle.

Concerning social aspects, in the beginning of waste management the main focus was to solve a public health issue; nowadays, the actual most relevant items to paid attention are the population acceptance of waste treatment infrastructures and the population participation in source separation collection systems (for packaging waste mainly).

1 - VALORMINHO
2 - RESULIMA
3 - BRAVAL
4 - RESINORTE
5 - Lipor
6 - Valsousa (Ambisousa)
7 - SULDOURO
8 - Resíduos do Nordeste
9 - VALORLIS
10 - ERSUC
11 - AMR do Planalto Beirão (Ecobeirão)
12 - RESIESTRELA
13 - AM da Raia-Pinhal
14 - RESIOESTE
15 - Ecolezíria
16 - Resitejo
17 - Amtres (Tratolixo)
18 - VALORSUL
19 - AMARSUL
20 - Amde (Gesamb)
21 - Amagra (Ambilital)
22 - Amcal
23 - VALNOR
24 - Amalga (Resialentejo)
25 - ALGAR

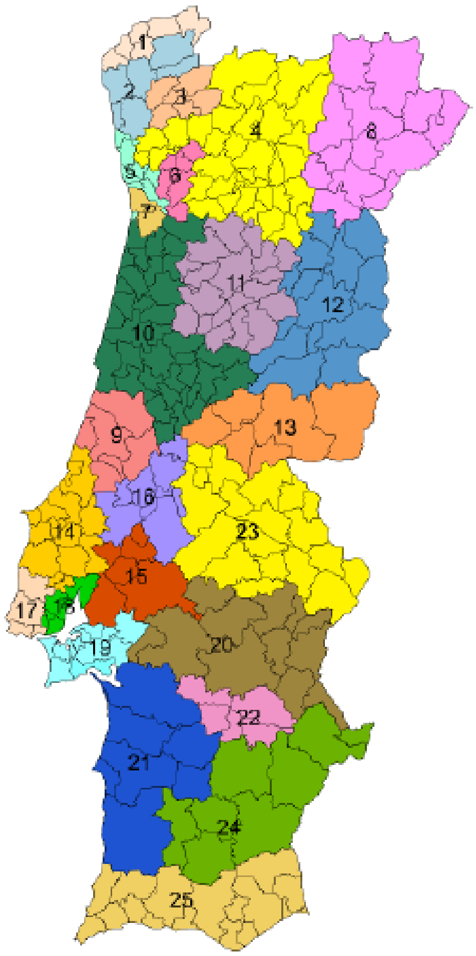

Fig. 1.4 Number of MSW systems existing in Portugal

According to Marques et al. (2005), in Portugal, due to the slow awake of the country to the post April 25 democracy and to the low level of economic development which Portugal had in the 1970’s, only 20 years later the Portuguese population acquired a certain degree of awareness concerning risk perception and environmental problems. In the 1990’s, with the implementation of waste management political strategies, social protests generalized and became common in waste management facilities siting processes. Specifically concerning MSW landfills implementation, in Portugal social conflicts happened in 20 different places during the 1990’s. To increase waste infrastructure facilities acceptance by population is vital the inclusion of their stakeholders in decision making process, concerning the infrastructures to be implemented. Concerning source separation collection systems, the actual public participation is due to population altruism and recycling campaigns, which translates in reduced recycling rates and participation rates at packaging waste recycling. Also, a positive impact of MSW management systems in social aspects is jobs creation.

How is notorious, Portuguese MSW management systems have become complex, with a considerable amount of stakeholders, expensive, social impact susceptible, and with several regulation restrictions and regulation targets to be reached.

Although the definition of the guidelines in PERSU II for waste management, MSW management systems need to find, individually, how to reach the targets and, at the same time, focusing in sustainable management.

As a consequence, all technical and non-technical aspects of a solid waste management system (SWMS) should be analyzed as a whole, since they are interrelated with each other and developments in one area frequently affect practices or activities in another area (UNEP, 2005). This implies the needs for systems analysis that emphasizes "The sum of the value of parts is less than the value of a whole".

Systems analysis techniques could be enlarged to assist in developing long-term MSW management plans (Huang et al., 2001) and to support decisions of short-term waste management operation, which should be helpful for analyzing tradeoffs among various socioeconomic and environmental objectives (Baetz, 1990; Thomas *et al.*, 1990; Huang *et al.*, 2002; Li *et al.*, 2006; Li *et al.*, 2008). These systems analysis techniques play an important role for regionalization assessment of SWM systems (Chang and Davila, 2006) and lead to reach environmentally effective, economically affordable and socially acceptable solutions (Morrissey and Browne, 2004).

There is a need to find improved ways to manage MSW system, to minimize the glitches in SWM, and to gear the insights and findings toward dealing with possible conflicts inherent in different purposes of environmental, social and economic management strategies leading to generate better policy and strategic planning needed to make these systems sustainable. The way analyzed to accomplish the sustainable management is turning to systems analysis.

1.2 GOALS AND SCOPE OF THE STUDY

The main goal of the thesis was the development of a systems analysis method which could improve and support decision making in solid waste management systems. To fulfill the goal, several sub-objectives were defined:

1. To know the role of systems analysis in waste management systems;

2. To assess which systems analysis methods are more adequate to be used in decision making for waste management systems;

3. To assess the application of such methods when used in a case study: AMARSUL waste management system, and to know their ability in decision making in waste management systems.

The choose of AMARSUL as the case study is justified by the fact that all the technologies existing in the present and predicted for the future are common to most waste management systems in Portugal. AMARSUL is the most representative of waste management choices in Portuguese context.

The appliance of systems analysis method life cycle assessment (LCA) and multicriteria decision making (MCDM) to help on MSW management system is justified by:

- LCA is a recommended method by Directive 2008/98/EC to assess waste management alternatives, when is not obvious that waste hierarchy principle is the best sustainable solution;

- LCA is the few systems analysis methods that is capable to bring a life cycle perspective on waste, which is relevant for sustainable purposes, such as resources scarcity and waste prevention;

- LCA is capable of being combined with other methods to increase its features;

- MCDM, being a sub-group of optimization modeling, is a well-defined method;

- MCDM can incorporates different types of information, quantitative, qualitative, of different scales and units;

- Used together with LCA, MCDM can have the property of life cycle approach;

- The use of both methods assures a sustainable method, capable to help on the goal of the thesis.

From a scientific point of view, the outcomes expected from the thesis are:

- Bring more scientific knowledge concerning the combined use of LCA and MCDM methods (analytical hierarchical process (AHP) based fuzzy interval technique for order performance by similarity to ideal solution (TOPSIS)).

From a MSW management point of view, the outcomes expected from the thesis are:

- The methodology developed to help on decision making could be applied at most Portuguese MSW management systems, giving a significant contribution on decision making;
- To know the effects of the inclusion of uncertainty in waste management decision making methodology applied in the thesis, such as implementation of pay-as-you-throw, information diffusion through all stakeholders and data uncertainty.

1.3 GENERAL STRUCTURE OF THE THESIS

The present thesis is divided in six chapters and in two annexes. In Chapter I is provided the introduction to the work developed and presented in the thesis. Following the purposes of the thesis, in Chapter II and III is conducted a literature review with the purpose of answering to know what are systems analysis, their methods, how have been applied in waste management systems and which are the benefits and drawbacks in this specific field. The systems analysis applications were not only search at a general view, but specifically for European countries. The result of those chapters has been the prevalence of LCA and MCDM methods has the most adequate to be applied in AMARSUL and national case. Both methods are presented and explained in Chapters IV and V, respectively. All the data used to conduct LCA and MCDM is presented in Annexes I and II. Finally, in Chapter VI are presented the conclusions of the thesis.

The work developed during the thesis is described in Fig. 1.5.

The Chapters have been fulfilled with articles under publication or submitted, being the articles related to the following:

- Chapter I – Introduction
- Chapter II – Literature review and state of the art of systems analysis applied in solid waste management:
    - Chang, N.-B., Pires, A., Martinho, G. (2010). Empowering systems analysis for solid waste management: challenges, trends and perspectives. Critical Reviews in Environmental Science and Technology. In press.
- Chapter III – Literature of systems analysis in solid waste management at Europe:

  - Pires, A., Chang, N.-B., Martinho, G. (2011). Solid waste management in European countries: a review of systems analysis techniques. Journal of Environmental Management, 92 (4), 1033-1050.

- Chapter IV – Life cycle assessment at solid waste management system in Setúbal Peninsula:

  - Pires, A., Chang, N.-B., Martinho, G. (2010). Reliability-based life cycle assessment for future solid waste management alternatives in Portugal. International Journal of Life Cycle Assessment. Submitted.

- Chapter V – Multicriteria analysis of solid waste management system in Setúbal Peninsula:

  - Pires, A., Chang, N.-B., Martinho, G. (2010). An AHP-based Fuzzy Interval TOPSIS Assessment for Sustainable Expansion of the Solid Waste Management System in Setúbal Peninsula, Portugal. Submitted.

- Chapter VI – Global conclusions.

- Annex I: support information for Chapter IV.

- Annex II: support information for Chapter V.

In Chapter I, an introduction has been elaborated focusing in Portuguese MSW evolution and the justification of applying systems analysis. Besides the scarce information, the relevant information was obtained from scientific articles and conferences proceedings.

In Chapter II a deep literature review concerning systems analysis methods was made, focusing on 14 methods: 1) cost-benefit analysis, 2) forecasting analysis, 3) simulation analysis, 4) optimization analysis, 5) integrated models, 6) management information systems, 7) scenario development, 8) material flow analysis, 9) life-cycle assessment, 10) risk assessment, 11) environmental impact assessment, (12) strategic environmental assessment, 13) socio-economic assessment, and 14) sustainable assessment. Scientific articles have been the basic information used to perform the literature review.

Chapter III was resulted from the previous section, being more focused on European applications. For this purpose the research focus has been scientific articles and conferences proceedings. Also national environmental agencies from European countries have been reached to provide information, however few have answered in a positive way to the request.

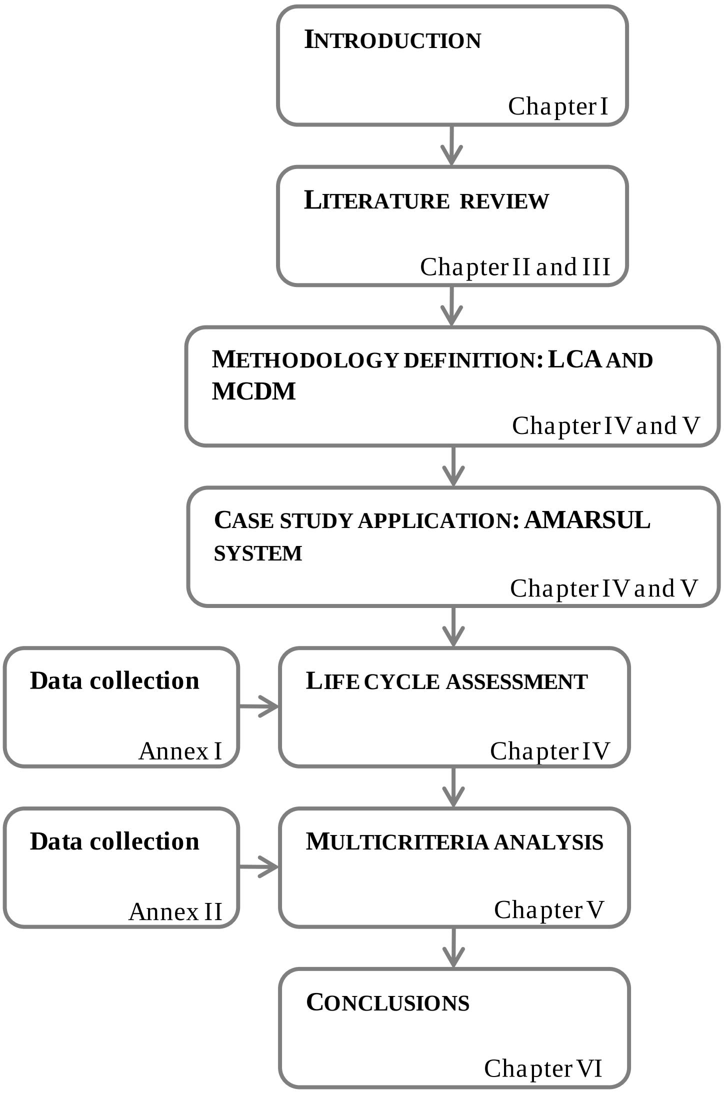

Fig. 1.5 Outline of thesis's structure

Concerning systems analysis methods, the LCA study is presented in Chapter IV. LCA was performed using UMBERTO 5.5 software. The data used are mentioned in Annex I. In Chapter V is showed the MCDM development for the case study. The MCDM method applied is TOPSIS, using AHP as auxiliary method for weight criteria. In Annex II, the information used in both methodologies is referred.

Since the structure of this thesis is composed by independent published or under publishing articles some methodologies and results framing had to be repeated.

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CHAPTER II - LITERATURE REVIEW AND STATE OF THE ART OF SYSTEMS ANALYSIS APPLIED IN SOLID WASTE MANAGEMENT

2 EMPOWERING SYSTEMS ANALYSIS FOR SOLID WASTE MANAGEMENT: CHALLENGES, TRENDS AND PERSPECTIVES

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(accepted for publication in Critical Reviews in Environmental Science and Technology)

2.1 ABSTRACT

Solid waste management is a significant issue for sustainable development that involves the technical, socio-economic, legal, ecological, political and even cultural components. For this reason, systems analysis has been uniquely providing interdisciplinary support for policy analysis and decision making in solid waste management for the last few decades. Considering these challenges and accomplishments retrospectively, this paper presents a thorough literature review and in-depth discussions of systems analysis models that are promising for providing forward-looking, cost-effective, risk-informed, and environmentally benign decisions for sustainable solid waste management. To simplify the discussion, the spectrum of these systems analysis models was divided into three broadly-based domains associated with fourteen categories although some of them may be intertwined with each other. The first domain comprises systems engineering models including cost-benefit analysis, forecasting analysis, simulation analysis, optimization analysis, and integrated modeling system; the second domain introduces systems analysis platforms, such as management information systems/ decision support systems/expert systems; and the third domain introduces system assessment tools including scenario development, material flow analysis, life-cycle assessment, risk assessment, environmental impact assessment, strategic environmental assessment, socio-economic assessment, and sustainable assessment. While some models or tools may be elucidated with respect to multiple managerial purposes, such as integrated modeling system and sustainable assessment, others may cover extended foci of market-based instruments and regulatory requirements, such as socio-economic assessment and environmental impact assessment. We used the term “systems analysis” as placeholder for the disparate strands of research and practice at this intersection between environmental systems engineering and sustainability science. Given that the sustainable management is

necessary at all phases of impact from the interactions among several prescribed paradigms, current and future solid waste management strategies in relation to systems analysis were particularly discussed. Such a critical review paper may aid the prediction of the possible challenges and the preparation of the appropriate tactics in dealing with large-scale complex solid waste management systems under normal operation and special conditions. It should lead the authors to echo some of their real world observational evidence in terms of cost-benefit-risk trade-off in decision analysis for solid waste management.

Keywords: Systems analysis, Simulation and modeling, Solid waste management, Sustainable management

2.2 INTRODUCTION

In systems engineering regimes, a system can be regarded as a set of related components or sub-systems, which interact with each other in various ways. The properties of a system are defined by the whole of the subsystems, their characteristics, and their relationships. The characteristics are related with the boundaries of the system depending on whether they are closed or open systems/sub-systems. In particular, the concept of "system of systems" is a collection of a few dedicated systems or subsystems that pool their resources and capabilities together to connect a more complex, 'meta-system' which offers more functionality and performance than simply the sum of the constituent systems (Maier, 1998). With this definition, a municipal solid waste management (SWM) system fits the complex "system of the systems", in which some of the sub-systems such as landfills, incinerators, anaerobic digestion units, composting facilities and recycling centers are linked with each other through processed waste streams internally providing varying functionality and performance. With implications for design, deployment, operation, and transformation of complex system of systems, the SWM systems may be deemed as an open system, needing materials and energy from outside social and economic channels reflecting impacts in and out of the system boundaries; or they may be deemed as a closed system in which the SWM network can be sustained internally with no transboundary movement. No matter what are the features at system boundaries, interrelated functionality and performance make part of the SWM system exhibit ample interactions between technical and non-technical aspects, both of which may influence the generation and shipping of waste streams to some extent. While technical aspects are tied to the capacity or throughput among related facilities, such non-technical aspects like imposed regulations, seasonal demands and policy instruments bring social, economic, and environmental features together responding to the evolving calls for sustainable development in the system.

Systems analysis has been applied to help many waste management agencies from the end of 1960s to the present with very different approaches. It may thus assist in developing long-term municipal solid waste (MSW) management plans and short-term waste management operational strategies with respect to various socioeconomic and environmental objectives (Baetz, 1990; Thomas et al., 1990; Huang et al., 2001; Huang et al., 2002; Li et al., 2006a; Li et al., 2008a; 2008b). Not only the ways to analyze the SWM systems have altered, but also the SWM systems themselves have been evolving due to technological advancement and structural changes of SWM. In the last few decades, the development of solid waste management technologies has ranged from planning and acting for the ability to maintain environmental quality to collectively meet our needs of waste management with greatest green potential, to enlarge the renewable energy recovery, and to preserve natural ecosystems. It is indeed a good chance for systems analysts to emphasize that "The sum of the value of parts is less than the value of a whole". On this basis, every community can tailor its own unique system to manage various components of the waste streams in an economically and environmentally sound manner (Najm et al., 2002). Clearly, the best waste management strategies from environmental, economic, and social equity points of view are not necessarily consistent. In essence, the cheapest way in economics may not be the most environmentally benign. The most suitable option in terms of social equity may not be the best one to meet the economic principles or environmental goals or both from managerial standpoints. These three aspects of waste management not only constrain, but also complement, each other in the development of a sound integrated solid waste management (ISWM) system by which the economic, environmental and social impacts can be harmonized. It has become necessary for decision makers to take an integrated approach to consider a series of SWM options aiming at capturing the life cycle aspects of waste management practices with respect to economic viability, environmental sustainability, and social equity (Wang et al., 1996). Thus, integrating technical with non-technical aspects for SWM means that all functional units from the product life cycle (i.e., from product manufacturing, to waste generation, to treatment and disposal) have to be emphasized. It aims to link regulation and environmental policy instruments (i.e., command and control, market-based, voluntary agreements, and information instruments) with financial aspects and engineering alternatives. This ultimately results in selecting alternatives with respect to risk adverse, risk averse or risk neutral pathways in the risk management paradigm.

Obviously, the sustainability concept demands even more as the inclusion of a temporal factor is needed in order not to compromise the benefits of future generations with the current consumption habits. As a consequence, the questions of how the SWM systems should be managed from a sustainable perspective, the proper formulation of systems engineering models and/or system assessment tools with multifaceted features connecting all waste, resources, material and energy flows together becomes indispensable in the 21st century. From a systematic point of view, such challenges constrain researchers to deal with the deeper complexity and dynamics that result from higher level connections among all "meta-systems" or "sub-systems" of waste management. It is envisaged that proper implementation of systems analysis should yield a balance between simplifications of the modeling efforts and soundness of capturing the essential features resulting from inherent complication of solid waste management in real world systems.

Systems analysis techniques with a variety of technical and non-technical implications have been applying for handling MSW streams through a range of methodologies in the last few decades (Pires et al., 2011). To ease the discussion, a total of fourteen system engineering models and system assessment tools were formally classified in this field to illuminate the challenges, trends and perspectives. It is worth knowing that the spectrum of these models and assessment tools was classified based on the following two domains with respect to 14 categories although some of them may be intertwined with each other . They are: 1) systems engineering models including cost benefit analysis (CBA), forecasting model (FM), simulation model (SM), optimization model (OM), and integrated modeling system (IMS), as well as 2) systems analysis platforms, including management information system (MIS)/decision support system (DSS)/expert system (ES), and finally system assessment tools such as: scenario development (SD), material flow analysis (MFA), life-cycle assessment or life cycle inventory (LCA or LCI), risk assessment (RA), environmental impact assessment (EIA), strategic environmental assessment (SEA), socio-economic assessment (SoEA), and sustainable assessment (SA).

Fig. 2.1 holistically entails the interrelationships among those two domains from which 14 techniques can be connected through such a technology hub in association with two broad-based domains -- systems engineering models and system assessment tools. In the core part, the five systems engineering models can be served as the core technologies where the model-based decision support system can be constructed for separate or collective applications. Yet the rule-based, knowledge-based or graphics-based decision support systems or expert systems can still be formed based on heuristic approaches using the rest of system assessment tools described by the 8 triangles outside. Communication among the 8 triangles can be made possible via the information flows that in turn improve the credibility of five systems engineering models being formulated. The integrated use of these systems analysis techniques play an important role for sustainability assessment of SWM systems (Chang and Davila, 2006) and may lead to reach environmentally benign, cost effective, ecological sound, and socially acceptable solutions (Morrissey and Browne, 2004). The following sections detail such challenges and trends in advancing SWM strategies. They aim to present a comparative analysis while also illuminating future perspectives.

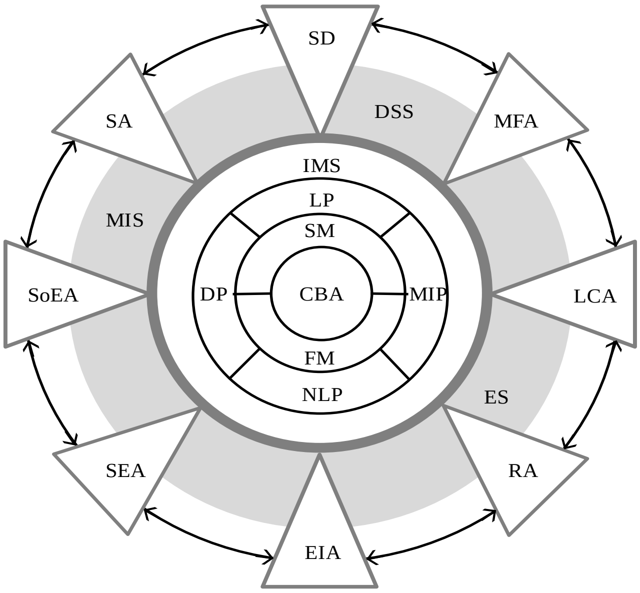

Fig. 2.1 The technology hub for solid waste management systems analysis

2.3 THE EVOLUTION OF SYSTEMS ANALYSIS FOR SOLID WASTE MANAGEMENT

Modeling solid waste management systems is a multidisciplinary activity. Several reviews relating challenges involved within the context of environmental modeling have appeared over the last few decades (Gottinger, 1988; MacDonald, 1996a; AEA Technology, 1998; Berger et al., 1999; Lukasheh et al., 2001; Huang and Chang, 2003; ETCWMF, 2003; Morrissey and Browne, 2004). The historical trends outlined below intend to clarify some of the facts that promoted SWM models and to actualize the information flows in support of these models systemically rather than symptomatically as already provided in the literature.

2.3.1 Systems Analysis in 1970s and Before

At the end of the 1960s and during the 1970s, first generation systems engineering models using linear programming (LP) with a single objective optimization scheme (i.e. cost minimization) were developed to characterize the waste flow pattern simply from transfer stations to landfill sites so as to minimize the total or partial costs in the SWM system (Anderson and Nigam, 1967; Anderson, 1968). At that time, fixed-charge problems of concern included costs, such as site acquisition, that are incurred regardless of the level of activity at a site. Facility capital and operating costs can then be represented for modeling purposes as a fixed cost and a variable cost (linearly dependent on facility capacity) in the integer programming (IP) model. Marks and Liebman (1970) considered the problem of selecting transfer stations in relation to transportation cost, including both fixed-charge and variable costs. Rossman (1971) extended the work of Marks and Liebman (1970, 1971) by adding incinerators to the set of potential facilities in the context of cost minimization. Also Esmaili (1972) developed a transfer station locational model to choose the combinatorial options of processing or disposal facilities, or both, from among a number of alternative facilities that would minimize the overall cost of haulage, processing and disposal of SWM operations over an extended period of time. Greenberg et al. (1976a, b) applied LP techniques to plan a real world waste management system in the US with respect to the cost minimization principles and technical constraints. Later, Clark (1973) discussed some regional planning models for SWM formulated as fixed-charge problems in the context of a mixed-integer programming (MIP) model. Helms and Clark (1974) presented an LP model to aid in selecting alternatives to incinerators and landfills considering fixed-charge and variable costs to be minimized together. Overall, some of the models referred to had applied mixed-integer linear programming (MILP) techniques for solving real world SWM issues, related to single network planning (Anderson, 1968; Fuertes et al., 1974; Helms and Clark, 1974; Kuhner and Harrington, 1975; Jenkins, 1979; Clayton, 1976), and dynamic, multi-period investment for solid waste management regionalization (Marks et al., 1970; Marks and Liebman, 1971). Walker et al. (1974) and Walker (1976) developed the SWIFT (simplex with forcing trials) algorithm for helping decide the number, type, size and location of the disposal sites in a region, which was adopted by the United States Environmental Protection Agency (USEPA). However, the USEPA (1977) itself developed the WRAP (Waste Resources Allocation Program), a model which contains static and dynamic MILP modules. Other types of optimization models, such as the dynamic programming (DP) approach, for solid-waste-management planning were also applied by Rao (1975). On the other hand, Truit et al. (1969) and Liebman et al. (1975) developed optimization models to solve waste vehicle routing as, at the time, very few researchers had concerned themselves with the local scale analysis of vehicle routing.

In concert with these optimization efforts, some independent simulation and forecasting modeling work was carried out for the prediction of waste generation and flow patterns. The first forecasting model was developed for SWM in early 1970s by Niessen and Alsobrook (1972) and Grossman et al. (1974), in which the extended per-capita coefficients were fixed over time and projected to change with time by including the effects of population, income level, and the dwelling unit size via a linear regression model. Clark and Gillean (1974) proposed modeling solid waste generation within the context of an MIS. They applied this idea to solve vehicle routing issues using data from the United State Environmental Protection Agency (USEPA). This was the first attempt to apply computational tools to SWM planning. Within this article, it is enlightening to know that, even this early stage, interdependencies between the various components of SWM were recognized. In addition, one simulation model applied to SWM planning was found in early 1970s (Bodner et al., 1970). The simulation practice therein was developed to determine the optimal routes for refuse collection vehicles. The program yielded exact routes suitable for use by municipalities. Crew size, vehicle capacity, and pickup time may be varied to permit efficient labor and equipment usage. The program computes overtime, incentive time, vehicular capacity utilization, mileage traveled, weight hauled, and productive time. A strategic evaluation of waste management practices was provided by Wilson (1977) to summarize these practices.

2.3.2 Systems Analysis in the 1980s

The 1980s was the decade in which several programs became available for experimentation with a wide range of configurations. The need to make models more realistic in terms of using a hierarchical approach started receiving attention during the 1980s. Such an approach was intended to bring complexity into the models at the system-level. Yet ISWM and waste hierarchy principle (WHP) may result in a dilemma in policy decision making and applications. The spectrum of these optimization models developed in 1980s the include the Standard Operational Procedure (SOP) (Gottinger, 1986, 1988), multiobjective evaluation for disposal planning (Perlack and Willis, 1985), MILP approaches including more types of constraints for SWM planning (Jenkins, 1980, 1982; Hasit and Warner, 1981), pure mixed integer programming (MIP) models (Kirka and Erkip, 1988), and dynamic programming models (Baetz et al., 1989). The optimization models with a single objective for vehicle routing and scheduling was still influential at the operational levels (Chiplunkar et al., 1981; Brodie and Waters, 1988; Shekdar et al. 1987).

Such more sophisticated models were developed to some extent due to a higher accessibility to computers, and the complexity of SWM issues revived some interest in the use of computational tools, especially electronic spreadsheets (MacDonald, 1996b). In fact, computational accessibility allowed the development of specific tools that could be free or proprietary. Among others, specific tools proposed were RRPLAN (Resource Recovery Planning) described by Chapman and Yakowitz (1984), a model which uses LP techniques to size and site facilities, and a cost accounting system to incorporate economies of scale and estimate the effects of decisions. Rushbrook (1987) and Rushbrook and Pugh (1987) described the Harbinger waste management planning using an optimizing model, developed by the Harwell Laboratory, UK and Wilson et al. (1984). It showed the application potential to SWM planning in Hong Kong. The WRAP model was applied by Hasit and Warner (1981). The ROMA model developed by Beture, an engineering and design company, for the City of Paris was also identified as a milestone in this field (Burelle and Monterrat, 1985; Light, 1990). Further, MIMES/WASTE (Model for description and optimization of Integrated Material flows and Energy Systems) developed by Sundberg (1989; 1993) and Sundberg et al. (1994) utilized a non-linear programming model (NLP) considering energy aspects in response to the increased complexity of SWM. At that time, most of those SWM models were still not widely used, because they were difficult for the non-specialists to understand and required a large initial investment of time and/or investment because some of them were proprietary systems (Anex et al., 1996).

Household waste generation multipliers showed a wide variation depending on the source of the survey at that time. Forecasting models for the predictions of waste generation saw progress with a less intensive though promising pace. Khan and Burney (1989) presented a regression model for forecasting solid-waste composition with respect to the consideration of recycling and resource recovery. This model utilized data from 28 international cities for such predictions that uniquely illustrate a strong correlation with actual observed data. Other types of forecasting models were developed by Rufford (1984) and Rhyner and Green (1988) too. Estimates of residential, industrial, and commercial solid waste quantities were computed for some regional SWM systems.

Planning solid waste management systems is not only governed by technical issues. Environmental concerns about system components, such as leachate from landfills and emissions from incineration plants, resulted in obligations to comply with regulations. This arose from the fact that highly specialized and even more expensive pollution control technologies were favored at the same time. Such wariness made those models evolve not only to focus on technical advancement and economic incentives, but also consider environmental quality constraints in the context of optimizations (Chang, 1989). Along this line, CBA in the environmental economics regime started coming into to play for the first time with regards to recycling causing environmental and economic assessment to be integrated cohesively with each other in the end of 1980s, like is the case ofGlenn (1988). Besides, ESs, an artificial intelligence technology, was also interplayed with environmental concerns to help assess possible contamination of aquifers from dumpsites and landfills. Salient examples include DEMOTOX model developed by Ludvigsen and Dupont (1988). Later, the application of ES for SWM was elucidated in greater detail regarding its cost-effectiveness in relation to the choice of waste treatment and disposal alternatives (Thomas et al., 1990).

2.3.3 Systems Analysis in the 1990s

In the 1990s, the improvement of optimization models started with the inclusion of green infrastructures to reflect sustainability goals like recycling centers (Englehardt and Lund, 1990), source separation and curbside recycling programs, and material recovery facilities (MRFs) (Morris, 1991). Optimal scheduling for landfill operation with the recycling effect was also evaluated by Jacobs and Everett (1992). Efforts were also directed toward evaluating and scheduling a given set of recycling measures to help achieve least-cost landfilling with extended lifetime (Lund, 1990). Optimization analyses for siting recycling facilities also became a big concern. Hsieh and Ho (1993) and Lund et al. (1994) discussed the optimization of solid waste disposal and recycling systems by using linear programming techniques for economic optimization. Huhtala (1997) emphasized the use of an optimization model to assess recycling rate at the most economically effective option. Daskalopoulos et al. (1998a) also included net cost and environmental impacts into a model to assess an SWM system. Integrated analyses include studying the locational theory of siting recycling centers (Highfill et al., 1994), transfer stations (Rahman and Kuby, 1995; Chang and Lin, 1997a), MRFs (Lund et al., 1994), optimal allocation of trucks for SWM (Bhat, 1996), waste collection (Kulcar, 1996), and vehicle routing system (Ong et al., 1990). With the aid of Geographic Information System (GIS), Chang et al. (1997c) and Chang and Lin (1997a) performed local scale optimization for collection vehicle routing and scheduling and for siting transfer stations, respectively. Chang and Lin (1997b) further applied GIS to siting transfer stations as an integral part of a bigger regional assessment model for screening and sequencing of dynamic operation among a set of waste management facilities. Besides, efforts in combining the environmental impacts, such as air pollution, leachate impacts, noise control, and traffic congestion, as a set of EIA constraints in a series of economics-oriented locational models were explored using the MILP models (Chang and Wang, 1994, 1996a, 1996b; Chang et al., 1993a, 1996, 1997a, 1997b).

In the 1990s, Multicriteria Decision Making (MCDM) was a discipline aimed at supporting decision makers faced with making numerous and conflicting evaluations and deriving ways to come to a compromised solution in a transparent process. Caruso et al. (1993) developed a location-allocation MCDM model that also reflects environmental issues like resource and environmental impacts on the top of costs. Courcell et al. (1998) formulated a MCDM model to assess economic and environmental performance of municipal multi-material waste collection and sorting programmes applied to nine such programs in European municipalities. Fawcett et al. (1993) and Alidi (1998) applied a goal-programming model to aid in the integrated SWM, using the analytic hierarchy process (AHP) for determining the weights and priorities for a given set of goals. Other types of MCDM models were later developed by Hokkanen and Salminen (1994, 1997), Karagiannidis and Moussiopoulos (1997), and Chung and Poon (1996). The involvement of multiple objectives in decision making within the ISWM process involves various trade-off problems among conflicting objectives (Haastrup et al., 1998). It is tied to the costs, environmental aspect like discharge coefficients, impact factors, and planning objectives, and may affect the simulation and optimization process and generated solutions in modeling stages (Huang et al., 2002; Fiorucci et al., 2003; Costi et al., 2004). In particular, they applied the compromise programming technique to harmonize the potential conflict during siting landfills, incinerators, and transfer stations in a growing metropolitan region. A goal programming model, which is a simplified form of multi-objective programming model, was also applied to assess the compatibility issues between recycling and incineration, considering economic efficiency and environmental protection goals achieved during trade-off (Chang and Wang, 1997b). The nonlinearity embedded in the modeling process was specifically handled by using a nonlinear goal programming model for urban solid waste management (Sudhir et al., 1996; Chang and Chang, 1998a). The use of modeling-to-generate-alternatives (MGA) approach developed by Chang and Li (1997) aimed at generating solid waste management alternatives with specific cost constraints. Specifically, Rubenstein-Montanto and Zandi (1999) applied a genetic-algorithm policy planning for the case of SWM.

System integration also requires concatenating external functions of FMs and/or SMs step by step with OMs providing dynamic information on waste generation and shipping over time. For example, a time series forecasting model (geometric lag model) of solid-waste generation was presented by Chang et al. (1993b) to meet such goals. As reviewed by Beigl et al. (2008), there were overall about 20 forecasting models being developed during 1990s. An effort was made by Lawver et al. (1990) to evaluate integrated SWM systems with a simulation model that is related to the discrete event simulation skill. Similar work was conducted by Anex et al. (1996) producing GIGO (Garbage in, garbage out), a model which might support a large-scale optimization analysis. Tanskanen and Melanen (1999) developed a simulation model – TASAR (Tool for Analysing Separation Actions and Recovery) to study the recovery level reached by different separation strategies in Finland. Baetz (1990) used integrated simulation and optimization models to determine optimal capacity expansion patterns for waste-to-energy and landfill facilities over time. Salient examples of advanced system synthesis include simulation models developed to support LCA and SEA, such as the ORWARE (Organic Waste Research) model (Dalemo et al., 1997; Björklund et al., 1999; 2000). Besides, Powell et al. (1996) and Powell et al. (1999) integrated LCA and MCDM models to examine environmental impacts from alternative waste management scenarios for the city of Bristol, UK. Weitz et al. (1999) put together an economic assessment of SWM.

It was recognized that uncertainty played an important role in decision making. In response to such challenges, systems engineering models for SWM had also evolved from deterministic to probabilistic considerations, from certain to uncertain concerns, and from affirmative to risk-based attitudes compounding the analytical framework at different levels, namely from data to model and to management. There are three types of tools, including probability theory, grey system theory, and fuzzy set theory, which may be helpful in addressing these uncertainties. For example, uncertainties are relevant to the random character governing solid waste generation and the estimation errors in some parameter values (Chang et al., 1997a). Such uncertainties can also be related to technical maintenance: The latter are generally difficult to quantify as exact assessment data (Seo et al., 2003). Besides, the waste-generation rate in a community could vary temporally and spatially (Huang and Chang, 2003). Further, the vagueness of planning objectives and constraints in decision making involves even more uncertainty (Chang et al., 1997a). To fully address the uncertainties in decision making, the fuzzy sets theory and interval (grey or inexact) programming techniques had received wide attention in the field. Extended optimization analyses include the grey linear programming, grey fuzzy linear programming, grey fuzzy dynamic programming, grey integer programming approaches in dealing with a hypothetical solid waste management problem in Canada for identifying the optimal location and capacity of waste treatment facilities (Huang et al., 1992, 1993, 1994, 1995a). Besides, Chang and Wang (1996c; 1997a) applied fuzzy goal programming in dealing with several specific issues for the ISWM in Taiwan.

It is worth mentioning that both DSS and ES continuously received wide attention to promote the ISWM. Extended studies were found useful in this period, especially in combination with information technologies. A relevant exercise can be found in Charnpratheep and Garner (1997), which combined fuzzy set theory and AHP into a raster-based GIS for a preliminary screening of landfill sites in Thailand. Many other cases utilized decision support knowledge for waste processing and economic assessment associated with SWM (Chang and Wang, 1996d; Barlishen and Baetz, 1996; Haastrup et al., 1998; Bhargava and Tettelbach, 1997). USEPA also developed several ES to enhance computer aided design for a leachate collection system, final cover and vegetative cover at landfills. Review articles showed more applications for SWM during this decade (Basri and Stentiford, 1995). Knowledge-based models developed by Boyle (1995) were used to compare components and parameters of the inputs and wastes of different industries and determined the potentials for reuse, recycling or disposal and respective treatments. Haastrup et al. (1998) concentrated on costs, air, water and soil pollution, road congestion, technological reliability, but did not cover noise, employment, health impacts and recycling goals. Su et al. (2007) preferred to focus on the inclusion of public policy impact as part of the solid waste management models.

Methodologies that evolve in the area of systems analysis were extended from system engineering models to system assessment tools in this era. Individual LCA/LCI models were directed to compare recycling routes for particular objects (Song et al., 1999), to assess packaging alternatives (Tillman et al., 1991), and to screen out waste treatment options like landfill vs. recycling and their associated systems (Kirkpatrick, 1993; Powell et al., 1998; Craighill and Powell, 1996; Rieradevall et al., 1997). For instance, White et al. (1995) presented a LCI model specific to a SWM system, where the product system was held constant and the evaluation was done based on the performance of alternatives for solid waste disposal. It was the first LCI dedicated to waste management. Interest in LCA had increased in the late 1990s against a background of comprehensive environmental legislation including Integrated Pollution Prevention Control (IPPC) and Best Available Techniques Not Entailing Excessive Cost (BATNEEC), the growth of the green consumer market and pressure from voluntary green groups. Corporate interest had also been aroused by the introduction of British Standard BS 7750: Environmental Management Systems (EMS), and the EC Eco-Management and Audit Regulation in 1993. The criteria of the EC Eco-labeling scheme are based on the results of partial life cycle studies (Craighill and Powell, 1996).

At this stage, the assessment of policies (including WHP) and directives through CBA (Bruvoll, 1998; Hanley and Slark, 1994; Brisson, 1997; Touche Ross Management, 1994; AEA Technology, 1998; Litvan, 1994) became popular, which included both tangible and intangible cost and benefit terms in association with necessary environmental and ecological assessment. This can be linked with the systems analysis to find pollutant sources and selecting wastes to be allocated to determine possible treatment options through the MFA method (Brunner and Rechberger, 2003). Besides, SEA practices during 1990s were all produced by European countries. The purpose of such a method is to assess national and regional waste management plans (Salhofer et al., 2007). EIA promulgated mainly by many developed countries in the 1990s was the gold standard for specific installations. For example, through Directive 85/337/EEC in Europe, EIA reports became required for new waste management facilities (Barker and Wood, 1999), and countries like Germany and Ireland had to produce EIA for waste disposals. In Portugal, EIA was required to be conducted for all incineration plants (Coutinho et al., 1998). With the aid of the exposure assessment in the context of an EIA, RA was developed in 1990s for municipal solid waste composting as well as incineration in order to examine various issues related to toxic substance emissions (Travis, 1991; Calabrese and Kenyon, 1991), the comparative risks when handling the SWM planning (Chang and Wang, 1996a) and the safety of employees and environment (CWMI, 1999).

2.3.4 Systems Analysis in the 2000s

The challenges of the 1990s had encouraged system engineering models and system assessment tools for SWM to be more realistic and multi-faceted, based on different configurations and purposes. Yet, future models still had to keep in mind their purpose – helping to choose the best technologies and/or management alternatives to make SWM systems more sustainable while including new perspectives, like constraints violation for siting waste management facilities and waste flow allocation (Huang et al., 2002), waste generation step (den Boer et al., 2007), social aspects to environmental and economic assessments (Kijak and Moy, 2004; Contreras et al., 2008), social interactions through game theory under uncertainty (Davila et al., 2005), societal response through minimax regret criteria (Chang and Davila, 2006, 2007), financial management through the government bonds associated with economic development and population growth (Dyson and Chang, 2005; Davila and Chang, 2005), closer linkage between water characteristics and management strategies, and policy (Chang and Davila, 2007; 2008), policy concerns with economic incentives (Chang, 2008), siting landfills by including more stakeholders (Chang et al., 2008), integrated simulaton with forecasting models (Beigl et al., 2008), and stakeholder-driven decision making process with the aid of spatial decision support technology (Chang et al., 2009). Recent efforts also intended to compare different technologies and options to improve existing assessment methodologies, like LCA, via constructing more friendly tools (EPIC and CSR, 2000; Environment Agency of England and Wales, 2000; Kirkeby et al., 2006; Diaz and Warith, 2006), and emphasizing energy recovery goals (Chang and Chang, 2001).

2.3.5 Trend Analysis

The historical narrative presented above clearly indicates that most models at this stage were developed in the United States and Canada. The work done during the 1970s on models for the planning and management of SWM systems dealt with applying and refining various optimization and heuristic techniques to provide a more realistic representation of SWM practices (MacDonald, 1996a). Planners and decision makers gained beneficial use of systems engineering models for achieving the basic short-term and long-term planning of SWM with respect to the cost minimization principle and technical constraints in 1970s. The types of real world SWM issues investigated varied from dynamic, multi-period investment for SWM regionalization to the local scale analysis of vehicle routing. In the mid-1970s, at the onset of the forecasting models, some researchers considered some other endogenous variables, such as the effects of population, income level, and the dwelling unit size, to characterize waste generation. In addition, the USEPA conducted the first attempt to carry out planning and management at system level using a prototype MIS.

Shortcomings from models developed in 1970s were pointed out by Berger et al. (1999). They include having only one time period in most cases, recyclables rarely being taken into account, having only one processing option of each type, and having a single generating source. These limitations have the effect of making them unsuitable for large-scale long-term planning according to Sudhir et al. (1996). Another drawback is that the models developed in 1970s had not promoted the use of WHP for waste management even though the waste hierarchy concept actually originated in the EU 2nd Environmental Framework Program of 1977. However, it appears to be one of the foci of the 1980s.

What actually motivated the elaboration of such models in 1980s? Firstly, it is necessary to understand the context that had forced/promoted them. Increasing waste generation, difficulties in labor-management relations, rising costs, and uncertainty in technology evolution were problems pointed out by Clark and Gilleann (1974). According to Gottinger (1988), at this time, increased attention was given to the efficiency and effectiveness of waste management operations and economies of scale effects of large waste treatment facilities. The concept of ISWM and AHP emerged in 1980s as two strands of research to promote system thinking at the same time. Furthermore, the availability of suitable mathematical optimization models emerged gradually with more powerful computational resources and comparative modeling skills that provided an extra impetus for the construction of such models. Cost-benefit concerns appeared simultaneously in an optimization model with respect to more types of constraints, such as environmental and recycling. Forecasting models for the predictions of waste generation in 1980s were moved on with less intensive yet promising pace by including the characteristics of waste composition. Further, the inclusion of MIS and ES gradually drove the application from system modeling to system assessment using artificial intelligence technology.

In 1990s, decision makers turned to rely much more on the both ISWM and WHP with respect to different occasions and regions. The former oftentimes ranked a few treatment options in order of preference with regard to scientific or technical evidence and the latter pursued a community-based approach trying to bring in all stakeholders for problem solving. Technology evolution associated with technical challenges in the beginning of the 1990s covered composting plants (odor control), landfills (leachate control and gas recovery), and incineration facilities (toxic gaseous emissions), thereby making decision makers consider a variety of waste management alternatives at an enhanced level of sophistication. Environmental emissions control of waste treatment and disposal facilities also became mandatory requirements, and decision making for siting new waste incineration and disposal facilities became the core argument for authorities to deal with, resulting in big societal impacts. Representative social movements resulted in several syndromes like NIMBY (Not In My BackYard), BANANA (Build Absolutely Nothing Anywhere Near Anyone), and LULU (Locally Unacceptable Land Use). After experiencing scarcity of resources, population growth and environmental deterioration of natural resources, the emergence of the sustainable development concept in Brundtland report of 1987 (WCED, 1987) started to be considered in dealing with SWM issues. With the increased complexity in modeling domain during this period, improved modeling skills in the area of simulation, optimization, forecasting and control were confirmed during the 1990s. Obviously, the promotion of ISWM strategies for meeting the sustainability goals under given complexity would inherently seek for a balance among options of incineration, composting and recycling so as to maximize the social welfare and minimize the public health impacts simultaneously, subject to an increased waste generation and limited land and resources availability. Along this way, many of the ISWM strategies were developed in the late 1990s as an indispensable tool to possibly reach a sustainable management goal for waste minimization, cleaner production, and resources conservation and recovery. This made system synthesis and system integration become a norm in the field.

With the increased complexity in the modeling domain during this period, improved modeling skills in the area of simulation, optimization, forecasting and control were confirmed during the 1990s. Consciousness and consensus were reached to some levels in 1990s in regard to how to integrate waste management options based on technical, economic and environmental factors, which led to promote the value of systems analysis for SWM. Within this time period, these dynamic optimization criteria cover the simultaneous interactions among the effects of waste generation, source reduction and curbside recycling, collection and transfer, processing and transformations, site selection, waste disposal, tipping fee evaluation, and environmental impacts like air pollution and leachate impacts. One IMS normally requires concatenating several external functions of FM and/or SM step by step with OM to form a powerful functional structure leading to perform more sophisticated practices. With the evolvement described above, a lot of uncertainties in decision making are related to the different nature of the complexities in, triggering reformulation of optimization models in the 1990s (Huang et al., 2002). System synthesis to illustrate the source of uncertainties via the use of fuzzy set theory, grey systems theory, and probability theory, was booming. While the ISWM became a common acronym almost mandatory by many industrialized countries at the government level, such a new concept to manage those SWM systems deepened the insights needed for conceptual modeling and resulted in profound impacts on methodological footings in systems analysis. This observational evidence was confirmed by the booming of a plethora of waste management strategies under uncertainty with high levels of complexity and subjectivity, which actually opened a new field of evolution of systems engineering models. GIS and DSS started being integrated with each other to help decision making with challenges on a long-term basis. On the other hand, observational evidence confirmed that improvements were extended from system engineering models to system assessment tools, including LCA or LCI, EIA, SEA, MFA, and RA. Such endeavor intended to supply decision makers to obtain the necessary information about how environmental impacts related to SWM systems can be better understood and managed as a whole via systems analysis. The strength at this stage rests upon the capability of integrating a variety of SM, FM with OM to achieve the multifaceted assessment needs whereas system assessment tools may provide background information to narrow down the options. Yet the lack of full system integration and/or system synthesis between ISWM strategies and system assessment tools limits the all-inclusive exploration for SWM.

From a retrospective point of view, the need to comply with regulatory aspects was already imposed during the 1990s, though it still received wide attention in system engineering models and system assessment tools in the early 2000s. However, regulations have been changing concerning not only environmental emissions but also targets in relation to the waste management hierarchy (recycling, recovery and incineration), waste diversion from landfill targets, and market-based instruments. But new approaches, like SD, MFA, SoEA, and SA, have been evolving since 2000 to elaborate versatile waste management plans. Heightened sophistication in SWM strategies can be seen in DSS (Fiorucci et al., 2003; Costi et al., 2004), LCA (Bovea and Powell, 2006), OM (Ljungreen, 2000; Chang and Davila, 2006), and the inlcusion of market-based instruments through models and tools (Nilsson et al., 2005; Bjorklund and Finnveden, 2007). Resulting from the advancement of cyberinfrastructure, the new century has also brought more powerful computers and more distributed data storage capacity to support systems analysis for SWM. Given that the Internet-based information technologies are more spread than ever, the increased applications includes the web-based GIS along with electronic data exchange throughput (Chang et al., 2001).

2.4 UNDERSTANDING THE INDIVIDUAL FEATURES OF SYSTEM ENGINEERING MODELS

To further elucidate the essence and uniqueness of systems analysis, it would be very insightful if those systems engineering models for SWM may be reviewed and discussed individually in greater detail. From a technical point of view, five modeling techniques can be classified as: 1) CBA, 2) FM, 3) SM, 4) OM, and 5) IMS. These form the basis of the review of different types of analytical tools for system assessment in the next section.

2.4.1 Cost-Benefit Analysis (CBA)

Cost-benefit analysis is a modeling technique for decision-makers to assess the positive and negative economic effects of a project or policy in which all relevant impacts are measured in both physical and monetary values. The theoretical foundation of CBA is economic welfare theory expressed through the linkages of “the willingness-to-pay” for a benefit and “the willingness to accept” for a cost. Within such a context, benefits are defined as increases in human well-being (utility), and costs are defined as reductions in human well-being. In many applications of SWM, it is necessary to estimate the monetary value of environmental and ecological impacts (i.e., indirect benefits and costs) which do not have a direct price estimable via the market mechanism so that the non-market value of natural resources can be taken into account in decision analysis for SWM (Boardman et al., 2001). Those goods with no market value are often referred to as 'public goods'. However, the value of these public goods has to be derived in some unique ways, such as through observed behavior, surveys, or estimated shadow prices (Boardman et al., 2001). The idea of decision making behind CBA is that a project should be carried out if the sum of direct and indirect benefits exceed sum of the direct and indirect costs (EEA, 2003). Economic impacts in this regard were assessed through the quantification of costs (capital, operational and expansion from different waste unit operations, tax/fees) and revenues (energy production, materials like recyclables and compost). Oftentimes, the value of all costs and benefits involved may be expressed as an assessment metrics in a case-based scenario of SWM for justification as a pure CBA or as an integral part of the FM, SM and OM. For this reason, as one of the objective functions, CBA is always deemed an integral part of systems engineering models. However, this should not prohibit CBA to be deemed as an independent system assessment tool. From policy standpoint, this metrics with having all CBA, FM, SM, and OM components cohesively integrated can be used in the ex-ante evaluation for the selection of an investment project (EEA, 2007a. 2007b). Yet, this metrics can also be used in the ex-post evaluation to measure the economic impact of an intervention when its effects may go beyond the simple financial effects for both the private and public investors in major infrastructure projects, especially in the transportation and environmental sectors (EEA, 2007a).

Some countries have developed guidelines such as the Nordic guideline for CBA in solid waste management specifically for waste management (Nordic Council, 2007). The methodology can be generally described by the following five steps: 1) objective definition and scope, 2) inventory, 3) monetary valuation, 4) discounting and 5) evaluation. Objective definition and scope is needed to precisely identify the problem to analyze which alternatives are to be assessed, functional units, system boundaries, and time horizon. Inventory is the step to be used for listing economic effects, effects from treatment of waste (reuse to final disposal of waste), time consumption and space in households, and environmental effects. Monetary valuation should be carried out to estimate direct and indirect economic costs and benefits in a project properly discounted to the present value based on the choice of a discount rate. Evaluation is the last step, resulting in the final result of the assessment in terms of net present value. The applications of CBA to aid in decision making of SWM systems may be deemed essential regardless of whether or not other types of models, such as forecasting, simulation, and optimization models, need to be applied. Fuzzy set theory may be combined with CBA, such as the fuzzy contingent valuation method (FCVM), to address the uncertainties and vagueness possibly associated with cost or benefit terms in the CBA (Chen et al., 2005). However, indirect costs and benefits terms that are related to the "intergeneration externality" were rarely considered in previous ISWM analyses. At the practical level, analysis of the cost effectiveness may be applied to assess the impact of the waste hierarchy principle (Schall, 1992). Assessment of possible options in municipal solid waste collection systems was made possible based on CBA (Tin et al., 1995). Another case study for waste paper disposal was carried out to investigate the economic and environmental consequences in relation to source reduction (via a tax), recycling, incineration, and landfill (Bruvoll, 1998).

Several studies of CBA applications in waste management can be found in (Pickin, 2008). However, most are applied to assess policies and not SWM systems. Cases in which CBA have been applied to assess SWM systems can be the case of Lavee (2010), where a full cost-benefit analysis of a deposit refund program for beverage containers in Israel was developed, considering the system elements: storage, collection, treatment technology and disposal. Dewees and Hare (1998) analyzed different packaging waste reduction programs and the costs and benefits of several policy options. Rabl et al. (2008) used CBA to evaluate the impacts and damage costs due to pollution from waste treatments, considering the system (disposal, treatment technologies and transport distance, and considered negligible waste collection and material recovery). Pearce and Turner (1993) analyzed waste management policies, in detail three typical policy instruments: packaging taxes, deposit-refunds and marketable permits. Also Palmer et al. (1997) studied three price-based policies for solid waste reduction: deposit refund, advance disposal fees and recycling subsidies, although it was not possible to know if the system perspective approach was considered. Ibenholt and Lindhjem (2003) have assessed recycling of liquid boards containers, and Vigsø (2004) investigated and quantified the social costs and benefits of collecting single use drink containers via a Danish deposit system.

2.4.2 Forecasting Models

Both planning and design of SWM systems require accurate prediction of solid waste generation (Dyson and Chang, 2005). Obtaining data related to solid waste generation is a difficult quest. At the onset of a SWM system, it is necessary to characterize the waste streams quantitatively and qualitatively and construct a management information system to accumulate the information flows over time. Even so, data from historic records normally is not available and data is often highly uncertain mainly because of its vague nature and disparate records in measurements. To capture the trend in waste generation, forecasting models have been developing since the 70s for solid waste management, based on methods like system dynamics, regression analysis, multiple regression analysis, correlation analysis, grey fuzzy dynamic modeling, time series analysis and material flow analysis methods. Decision and policy makers in SWM systems or governmental institutions often prefer to apply forecasting models to avoid missing links in long-term ISWM planning.

Single and multiple regression analyses are the most common forecasting methods for estimating solid waste generation. These models are designed to describe and evaluate the relationships between a given variable (e.g., waste generation) and one or more relevant variables for making good predictions of the future trend of waste generation. When applied, they predict the outcome of a given factor (dependent or explained variable) based on the interactions with other related drivers (independent or explanatory variables). Factors that influence solid waste generation are normally related to population (Grossman et al., 1974, Saeed et al., 2009, Jiang et al., 2009), income level (Grossman et al., 1974; Beigl et al., 2005), dwelling unit size (Grossman et al., 1974), total consumer expenditure and gross domestic product (Daskalopoulos et al., 1998b), production measures, household size, age structure, health indicators (Beigl et al., 2005), and per capita retail and tipping fees for waste disposal (Hockett et al., 1995). These modes therefore help understand which variables are best related to solid waste generation. Factors identified as relevant often include household size, tenure and type of accommodation, home heating arrangements, employment status, social class, education level attained by head of a household, and age profile of residents (Abu-Qdais et al., 1997; Dennison et al., 1996a, b; Benítez et al., 2008). Bach et al. (2004) have identified the number of overnight stays per person, indices of purchasing power, parameters describing the employment structure and family structure of municipalities as significant factors to predict collectable waste paper. Chang and Davila (2008) have applied a multiple regression model to predict the lower heating value of MSW. Liu et al. (2006) integrated a regression model and MFA to predict the amount of five scrap electronic appliances in urban households and to analyze the flow after the end of their product life time so as to aid in planning the collection system and processing facilities needed for management of such e-waste in the near future. To further account for temporal impacts and identify the historic trend, time-series regression analyses - extensions of regression models, were used to forecast solid waste generation. The most common model applied in this area is the Auto-Regressive

Integrated Moving Average (ARIMA), developed by Box-Jenkins in 1976. Its applicability has been proved effective to forecast short-term solid waste generation contributing to collection vehicle scheduling (Katsamaki et al., 1998; Navarro-Esbrí et al., 2002), assessment of how recycling programs affect solid waste generation (Chang et al., 1997c), and even an assessment of lifestyle changes on household waste generation (Howard et al., 2006). Some econometric forecasting methods could also be helpful to predict waste generation, but these methods need a socioeconomic and environmental database to support the essential practices (Chen and Chang, 2000).

More advanced forecasting models are those that are able to aggregate different dynamic features from solid waste generation and their interactive interrelationships – dynamic systems. The system dynamics modeling approach attempts to quantify qualitative aspects without altering the accuracy of the original statement and provides a much more explicit basis for communication. According to Dyson and Chang (2005), to build a system dynamics model, one should identify a problem and develop a dynamic hypothesis explaining the causal loops of the waste management problem. The model formulation is normally designed to test one or several unique scenarios with regard to alternative policies in the SWM systems. Simulation runs in regard to the state of the system (e.g., waste generation) in a system dynamics model is entirely governed by the passage of time. When the initial conditions of the state variable(s) can be assigned, the model may start to produce the related consequences of the flow information. Thus more recently, some system dynamics models were implemented to predict solid waste generation, being justified by the complex socio-economic nature embedded in solid waste generation. These models are capable of dealing with assumptions about system structures in a stringent fashion, and particularly of monitoring the effects of changes in subsystems and their relations representing systematic feedbacks rendering the modeling structure communicable (Karavezyris et al., 2002). At a practical level, Karavezyris et al. (2002) considered environmental behaviors, waste treatment price, quantities of waste collected, treated and recycled, and regulation in the context of a systems dynamic model. Dyson and Chang (2005) illustrated how the selected driving forces can change the trend of solid waste generation in terms of total income per service center, people per household, historical amount generated, income per house and population. The model predicted waste generation in a fast growing city based on those influential technical and socio-economic factors to support MRF planning. When system dynamics models suffer from the impact of uncertainty in estimations, random variables may be defined to replace constant parameter values. Otherwise, the grey fuzzy dynamic model can be a legitimate substitute for predicting solid waste generation while keeping all socio-economic and technical aspects as the influential factors (Chen and Chang, 2000). Recently, Beigl et al. (2008) presented a review concerning forecasting models applied to support SWM systems. Thøgersen (1996) used single regression analysis to assess relations between MSW production and consumption styles and Gay et al. (1993) have applied input-output analysis to estimate county and city-level solid waste composition and generation.

2.4.3 Simulation Models

Simulation modeling is defined as the use of digital computers to trace lengthy chains of continuous or discrete events based on the cause-and-effect relations describing the operations in complex systems and helping investigate the dynamic behavior of the system (Wang et al., 1996). When applied to handle SWM issues, the interactions between selected variables, each of which can affect and be simultaneously affected by the others, can hardly be amenable to purely mental evaluation or ordinary mathematical treatment (Wang et al., 1996). Making an analogy to Driving Forces-Pressures-State-Impacts-Responses (DPSIR) framework terms, simulation models predict the state and sometimes the impact of determined pressure (Wang et al., 1996). Such efforts may help to predict the consequences of some sources of environmental impacts with or without involving the time domain.

In SWM systems, it is possible to use the same perspective for investigating the behavioral patterns of the system of systems with changing inputs when choosing parameters to understand how the object that is being simulated, behaves (state). The purpose of such simulation models in this field can be logistic simulation, single and multi-machine processes, simulation of the environmental fate and transport of waste constituents, and simulation of costs and schedules for waste management project or program (Miller et al., 2003). Such simulation models can test the SWM systems at low cost. With such a tool at hand, it is possible to allow the exploration of complex systems in many different ways (Wang et al., 1996). The more variables (e.g. locations of facilities, size and type of collection trucks, type of recyclable materials to be collected) that users can specify, the more dimensions the model can investigate when simulating a complex system. These computer-based models can then simulate the dynamic evolution of a real or proposed system and could be formulated via a spreadsheet-based, discrete-event, transaction-based approach to modeling specific changes to the system in the context of system dynamics studies (Miller et al., 2003). Within this context, spreadsheet-based models are the most used, Microsoft Excel being the predominant software package. These models typically use columns in the spreadsheet to represent the system's state variable at a point in time. They consist of entities (units of traffic), resources (elements that service entities) and control elements (elements that determine the states of the entities and resources). Therefore, the applications of such models can make SWM systems process waste streams more easily understandable (i.e., in other words, how the waste life cycle works) and can show, through trying different changes in simulation, whether there is a need for improving the SWM systems.

Table 2.1 summarizes most of the interesting simulation models focusing on decision making of SWM with a system perspective. As presented in Table 2.1, some models, such as SWIM (Solid Waste Integrated Model) (Wang *et al.*, 1996), GIGO (Lawver *et al.*, 1990; Anex *et al.*, 1996), AWAST (Aid in the management and European comparison of a municipal solid waste treatment for a global and sustainable approach) (Villeneuve *et al.*, 2009), EcoSolver IP-SSK (Krivtsov *et al.*, 2004), TASAR (Tanskanen and Melanen, 1999) belongs to the type of simulation models designed specifically for assessing the functionality of SWM systems.

Table 2.1 A summary of simulation models applied for solid waste management

Reference	Scope	Methodology
Tanskanen and Melanen, 1999	TASAR model to study recovery levels reached by different separation strategies.	Spreadsheet-based – Microsoft Excel (static and linear)
Wang *et al.*, 1996	SWIM interactive computer package to provide a structure for systems analysis of SWM problems.	Spreadsheet-based – Microsoft Excel
Ycel and van Daalen, 2008	Simulation model to understand Dutch waste management through replicating historical trends, exploring the underlying mechanisms, hence contribution to the level of comprehension.	Systems dynamics
Kum *et al.*, 2004	A decision support tool for financial planning in community-based SWM systems	System dynamics through Vensim
Krivtsov *et al.*, 2004	Simulation model (EcoSolver IP-SSK) to assess the glass waste management strategies in SWM systems	System dynamics
Wäger *et al.*, 2001	Simulation model to assess the plastic waste management strategies in SWM systems	System dynamics from PowerSim
Villeneuve *et al.*, 2009	Simulation model of SWM systems to provide decision-makers with quantitative information that can be brought into public debate.	AWAST – spreadsheet-based
Tucker and Fletcher, 2000	Simulation model addressing community-based composting behavior based on contributions of individual households, each actively managing the organic fraction of their own domestic waste.	Discrete-event

In the context of system assessment, the applications of simulation tools for decision making can be further classified into two different types of models. One type encompasses the environmental assessment models, like LCA and MFA models, and the other type assesses only the functionality of SWM systems. The LCA and MFA models will be discussed independently in separate subsections below as part of the assessment tools. To consider the uncertainties embedded in the systems of interest, Monte Carlo simulation technique is usually the tool of choice.

2.4.4 Optimization Models

Optimization models are the core of the systems engineering modeling approach. Single objective programming (SOP) models aim to search for the optimal solution associated with a well-defined SWM problem in which there is a single objective and several technical and managerial constraints in the context of MCDM (Edwards-Jones *et al.*, 2000). These models were often applied to help solve cost minimization issues and were normally formulated by deterministic methods, including linear programming, non-linear programming, dynamic programming and mixed-integer programming models. Along these lines, these optimization models are capable of optimizing economic issues like the minimization of total costs or to maximize the total benefits to help the vehicle routing (Liebman *et al.*, 1975), to decide what type of SWM system should be designed and the location of landfill facilities, incinerators, transfer stations (Anderson and Nigam, 1967; Anderson, 1968; Esmaili, 1972; Helms and Clark, 1974; Marks and Liebman, 1970, 1971; Gottinger, 1986, 1988; Kirka and Erkip, 1988). Environmental issues such as materials and energy management (Caruso *et al.*, 1993; Chang and Chang, 1998a; Hokkanen and Salminen, 1997), greenhouse gas emissions, acidification compounds, and other pollutant emissions (Hokkanen and Salminen, 1997; Seo *et al.*, 2003; Nasiri and Huang, 2008) were of significance. Others were social issues like public acceptance, employment (Hokkanen and Salminen, 1997), labor issues (Chang *et al.*, 1997b) and public consensus and participation (Hung *et al.*, 2007). To harmonize both aspects, Berger *et al.* (1999) developed an optimization model to help decision-makers arrive at a long-term planning of SWM activities by which both environmental and social issues were brought into the optimization context.

Approaches applied to improve results might concern uncertainty associated with either data or the waste management decision making itself. The methods that were applied for addressing the uncertainty impacts mainly consist of fuzzy set theory, grey system theory, and probabilistic theory. Some of these techniques are used alone or in combination with others. Stochastic programming requires large data sets for the identification of the probabilistic distributions, and its application is helpful to effectively reflect the probability distributions of a single right-hand side value in a constraint of optimization models (Huang *et al.*, 2001; Li *et al.*, 2006c). Combination of several right-hand-side values makes the algorithm numerically intangible. Fuzzy sets theory that refers to the absence of sharp boundaries in the information was applied to support decision analysis of SWM systems in the context of various optimization models. A subjective continuous membership function is usually used for the description of this kind of vague information (Chang *et al.*, 1997a). It enables one to deal with uncertainties connected with vague linguistic expressions in decision making when probabilistic data are not available. Grey systems theory applied to support optimization analysis in SWM systems is capable of dealing with several uncertain parameter values while at the same time addressing the vagueness of its intrinsic characteristics in the information during parameter estimation (Chang *et al.*, 1997a). Such parameters are most likely expressed as interval numbers linked with the environmental or economic factors in objective functions and constraints. It was applied to handle a variety of uncertainty concerns associated along with costs minimization in different SWM systems with respect to construction and expansion planning of waste management facility and waste flow allocation planning (Huang *et al.*, 1992, 1994, 1995a, b). Huang *et al.* (1993) first conducted cost minimization using a grey fuzzy integer programming model. Huang *et al.* (2001) pointed out that integrated methods with various combinations of the three uncertainty theories above can produce answers concerning types, times and sites for SWM practices with improvements in uncertainty, data availability and computational requirements. Such integration enables us to handle uncertainty of different sources at the same time (Zou *et al.*, 2000). With such a philosophy, the interval-parameter fuzzy-robust programming model was developed and applied to a SWM system to minimize the total system cost through optimal waste flow allocation (Nie *et al.*, 2007).

Facing the need to include multiple objectives, such as the need for minimization of total cost and maximization of recycling efforts at the same time, multi-objective programming (MOP) models were often formulated and applied. These deterministic MOP solution procedures may search for the compromised or satisfactory solution via a variety of methods. They include, but are not limited to the AHP, TOPSIS (Technique for Order Preference by Similarity to an Ideal Solution), ELECTRE (Elimination and Choice Translating Algorithm), PROMETHÉE (Preference Ranking Organization Method for Enrichment Evaluation), and NAIADE (Novel Approach to Imprecise Assessment and Decision Environments), to aid in SWM decision making (Caruso *et al.*, 1993; Hokkanen and Salminen, 1997; Chang and Lu, 1997; Chang *et al.*, 2009). When uncertainty becomes a major concern, probability theory, fuzzy set theory, and grey systems theory may also be applied to supplement the model formulation of MOP. For example, Chang and Wang (1997a) applied the fuzzy set theory to help in SWM system assessment with several objective functions that intended to minimize costs, traffic impacts, noise impacts and air pollution impacts simultaneously. Findings indicate that the use of fuzzy sets allows a better comparison between these objectives reflecting economic, environmental and social nature. Further, the fuzzy sets and grey systems theories can be combined with each other in a MCDM formulation with respect to environmental performance indicators in the sense that uncertainties related to fuzzy goals and inexact or grey parameter values may be properly integrated as a valid part of the MOP models (Chang and Wang, 1997a; Chang *et al.*, 1997a).

In decision sciences, however, an important issue that can affect the consequences of a SWM system is related to the violation of policies pre-defined by authorities, such as capacity expansion limits within a defined time period, or capacity limitations from installations which would have economic consequences. This unique type of uncertainty in decision making was brought into optimization context through the use of a two-stage stochastic programming (TSP) model – an extension of the stochastic programming model. In the TSP, a decision is first undertaken before values of random variables are known; then, after the random events have happened and their values are known, a second-stage decision can be made in order to minimize 'penalties' that may appear due to any infeasibility (Loucks *et al.*, 1981; Birge and Louvenaux, 1988, 1997; Ruszczynski, 1993). Many models for handling the SWM issues were formulated as two-stage stochastic programming models with the aim of including economic penalties based on the degree of violation of policy targets. The remaining features that characterize such models are related to combinations of methods for addressing uncertainty. To the greatest extent, the fuzzy interval two-stage stochastic mixed-integer linear programming model is capable of including uncertainties described in terms of probability density functions, fuzzy membership functions and discrete intervals (Li *et al.*, 2006b). It allows some penalty violation under a range of significance levels, being capable of facilitating dynamic analysis of decisions for expansion capacity planning for multi-region, multi-facility, multi-period and multi-option (Li *et al.*, 2006b), resulting in alternative solutions with respect to environmental, socio-economic and system reliability conditions. If nonlinearity exists at the planning stage, fuzzy two-stage stochastic quadratic programming model is capable of reflecting uncertainties expressed as probability distribution, interval values, and fuzzy-membership functions, respectively, providing better general satisfaction after tackling the nonlinearity in the context of optimization, and generating more robust solutions (Li *et al.*, 2006b). Overall, Table 2.2 summarizes the spectrum of application addressing ramifications of uncertainty analyses.

Table 2.2 A summary of optimization models applied for solid waste management

Reference	Scope	Methodology
Li *et al.*, 2007	Uncertainty analysis through optimization of waste flow and/or capacity expansion	Inexact two-stage chance-constrained LP
Li *et al.*, 2006a		Interval fuzzy two-stage stochastic mixed integer LP
Nie *et al.*, 2007		Interval-parameter fuzzy-robust programming
He *et al.*, 2009a,2009b		Inexact mixed integer bi-infinite programming
Huang *et al.* 1995a, 1995b		Grey integer programming (GIP)
Huang *et al.* 1995c		Grey fuzzy integer programming
Huang *et al.*, 2001		Interval-parameter fuzzy stochastic LP
Li *et al.*, 2006b		Interval-parameter two-stage stochastic mixed integer programming
Huang *et al.*, 1994		Grey dynamic programming
Zou *et al.*, 2000		Independent variable controlled grey fuzzy LP
Li *et al.*, 2008a		Two-stage fuzzy robust integer programming
Chang and Wang (1996a)		Grey fuzzy multiobjective mixed integer programming (MOMIP)
Li *et al.*, 2009b		Inexact fuzzy-stochastic constraint-softened programming
Li *et al.*, 2009a		Interval-fuzzy two-stage chance constrained integer programming
Guo and Huang, 2009		Inexact fuzzy chance-constrained two-stage mixed-integer LP
Guo *et al.*, 2009		Interval-parameter fuzzy-stochastic semi-infinite mixed-integer LP
Jing *et al.*, 2009		Interval-parameter two-stage chance- constraint mixed integer LP
Xu *et al.*, 2010		Stochastic robust interval LP
Zhang *et al.*, 2009		Hybrid interval-parameter possibilistic programming
Xu *et al.*, 2009		Stochastic robust chance-constrained programming
Cai *et al.*, 2009		Interval-valued fuzzy robust programming
Li and Huang, 2009b		Interval-parameter robust optimization
Li and Huang, 2009[a]		Inexact minimax regret integer programming
Liu *et al.* 2009		Dual-interval parameter LP
Wu *et al.*, 2006	Uncertainty analysis through optimization of waste flow allocation, considering effects of economies of scale	Interval non-linear programming
Chang *et al.*, 1997[a]	Location-allocation model considering economic costs and environmental issues	Fuzzy interval multiobjective mixed integer programming approach
Chang and Lu, 1997		Fuzzy global criterion approach
Cheng *et al.*, 2003	Waste landfill siting and waste flow allocation	Inexact MILP
Cheng *et al.*, 2009	Uncertainty analysis to	Random-boundary-interval LP

Reference	Scope	Methodology
	determine optimized waste allocation	
Grunow and Gobbi, 2009	WEEE management system	Mixed integer programming
Badran and El-Haggar, 2006	Optimized municipal SWM system concerning collection stations location	Mixed integer programming
Mitropoulos *et al.*, 2009	To determine the number, sizes and locations of the SWM facilities	Mixed integer programming
Seo *et al.*, 2003	Uncertainty analysis helps select the preferred solid waste management system	Fuzzy and AHP
Tseng, 2009	To evaluate different MSW management solutions	ANP (analytical network process) and DEMATEL (decision making trial and evaluation laboratory)
Huang *et al.*, 2007	Include public for sustainable decision making	AHP with consensus analysis model
Nasiri and Huang, 2008	Environmental performance assessment of waste recycling programs	Fuzzy multiple attribute decision analysis
Huang *et al.*, 2002	Violation analysis constraint through optimization of waste flow allocation and/or capacity expansion	Interval-parameter fuzzy integer programming
Li and Huang, 2007		Fuzzy two-stage quadratic programming
Li *et al.*, 2008b		Two-stage programming
Li and Huang, 2006		Inexact two-stage mixed integer programming
Maqsood *et al.*, 2004		Inexact two-stage MILP
Otegbeye *et al.*, 2009	To assess recycling system form SWM	LP
Berger *et al.*, 1999	To help regional decision-makers in the long-term planning of SWM activities	Mixed-integer LP

2.4.5 Integrated Modeling System

The Integrated modeling system class consists of different types of models which, by their nature, present different features, scales and complexity. From an environmental point of view, they may significantly help address the forcing of human-induced impacts, identify the responses in the environmental systems, and assess consequences due to such disturbances in our society (Huang and Chang, 2003). From the perspective of MSW management, the use of IMS can be helpful to understand the driving forces that are responsible for the SWM system behavior and the consequences of that outside the systems. Models used in the context of IMS therefore may cover the integration or coupling of simulation, forecasting, and optimization analyses. This is, however with a higher uncertainty, most of the time, since data from SWM systems are often of low quality, the methods that were employed to address various types of uncertainties by themselves, exhibit a higher variation over time in the context of integrated modeling analysis. Hence, scaling and consistency among integrated modeling components have to be harmonized.

By looking at a particular IMS, it becomes more visible as to what kind of skills and solutions from models can be simultaneously applied. For instance, integrated simulation and optimization (or vice-versa) modeling systems, as presented in Table 2.3, refers to a group of solution techniques that intends to solve complex interactive problems among these systems engineering models by possessing interactive, uncertain and/or highly non-linear features in which precise formulations of the investigated systems do not often exist (Fu, 1994). The IMS, considering some or all of the objective functions and constraints, can be both stochastic and implicit functions of decision variables that can only be evaluated efficiently through computer simulation (Yeomans and Huang, 2003). Finding solutions to such problems tends to be difficult and generally necessitates the combination of simulation with an optimization-based search technique. The underlying rationale for the optimization component is to efficiently guide the exploration strategy through the solution space using only a limited number of simulation experiments (Lacksonen, 2001). On the other hand, optimization-simulation models intend to solve the concern that optimized models only produce one solution without regard to those near optimum solutions, whereas those near optimum solutions can be easily identified through a simulation module. In this case, it is advisible to construct an interactive loop, since solutions from the simulation model may be reintroduced again into the optimization model before reaching the “stop criteria”. A salient case where such IMS was applied for SWM includes Baetz (1990), in which the procedure can be justified by the stochastic nature of solid waste data. Besides, Karavezyris *et al.* (2002) applied simulation models to support forecasting of waste generation, since it is assumed that a dynamic process may be suitable to delineate the simulation events. Simulation software used in such forecasting models can be Vensim®, Stella®, and Monte Carlo simulation. Another type of integration used to support decision making in SWM systems includes the integration of forecasting and optimization models in a two-tiered framework to link a dynamic forecasting model such as grey fuzzy dynamic model (Chan and Chang 2000) with an optimization model (Chang and Chang, 1998; Chang and Wang, 1996b, 1997a; Chang *et al.*, 1997a) or link a systems dynamic model (Dyson and Chang, 2005) with an optimization model under uncertainty (Chang *et al.*, 2005; Chang and Davila, 2007). A more recent case of IMS is one that relates an evolutionary algorithm with the integrated model, called evolutionary simulation-optimization (ESO) (Yeomans, 2007). The inclusion of ESO into an IMS allows search procedures exploration to be performed by the optimization component through the solution space that could be directed by the evolutionary algorithm (Yeomans, 2007).

Table 2.3 A summary of basic integrated modeling systems applied for solid waste management

Reference	Scope	Methodology
Yeomans and Huang, 2003	Planning design phase for the expansion of a waste management system	Evolutionary simulation-optimization model
Baetz, 1990	To determine the optimal capacity expansion patterns for waste disposal and waste-to energy facilities over time	Optimization/simulation model
Huang *et al.*, 2005	Policy planning model for large-scale municipal solid waste problems planning with significant uncertainty	Evolutionary simulation-optimization (with grey implication) model
Yeomans, 2007	Planning a MSW system	Evolutionary simulation-optimization (with grey implication)
Dyson and Chang, 2005; Davila and Chang, 2005	Dynamic forecasting of solid waste generation, simulation of the SWM network flow pattern and optimization for SWM model under uncertainty	Forecasting/optimization model
Sufian and Bala, 2007	System dynamics to predict solid waste generation, collection capacity and electricity generation from solid waste	Simulation/forecasting
Karavezyris *et al.*, 2002	Dynamic forecasting of solid waste generation	Simulation/forecasting
Anex *et al.*, 1996	An event-based simulation for operation of a SWM system	Simulation/forecasting
Tucker *et al.*, 1998	Used spreadsheet-based and time series to predict waste recycling targets from social behavior – voluntary	Simulation/forecasting
Huhtala, 1997	Optimal recycling rate for MSW management	Optimization/simulation
Chang and Davila, 2006	Landfill site selection	Optimization/forecasting

In decision analysis of complex systems, such terms as “multiple criteria”, “multiple objectives,” or “multiple attributes” are often used to describe decision situations. These terms are used interchangeably in different occasions. Basically, multiple criteria decision-making (MCDM) has seemed to emerge as the common nomenclature for all decision analysis models and approaches in dealing with both multi-objective decision-making (MODM) and multi-attribute decision-making (MADM) (Chang, 2010). Hence, research on multi-criteria problems can be broken down into two broad categories: MODM and MADM. The former problems refer to making decisions in the presence of multiple, usually conflicting, objectives and can be defined as finding a feasible alternative that yields the most preferred or satisfactory set of values for the objective functions. Both MODM and MADM may be used as core models as an integral part of the IMS to accommodate not only the complexity in decision making but also the incorporation of different system assessment tools, such as LCA, EIA, MFA, and EFA. To ease the applications, as shown in Table 2.4 illustrating some extended regimes, these IMSs may be constructed by embracing the advancements of system assessment tools via various types of system analysis platforms, such as GIS, MIS, DSS, and ES, and carry out the requirements introduced from multiple disciplines.

Table 2.4 A summary of extended integrated modelling systems applied for solid waste management – IMS with systems assessment tools

Reference	Scope	Methodology
Azapagic and Clift, 1998	Linear programming as a tool in life cycle assessment	LCA and LP
Contreras, et al., 2008	To analyze treatment plans for MSW	LCA and MCDM
Skordilis, 2004	A systems engineering model for the strategic planning of an integrated solid waste management	LCA and MCDM
Chang et al., 2008	Landfill siting	GIS and MCDM
Şener et al., 2006	Landfill site selection	GIS and MCDM
Chiueh et al., 2008	Spatial methodology was developed for distribution of a compensatory fund based on environmental impact	GIS and MCDM
Kontos et al., 2005	To optimize landfill siting	GIS and MCDM
Shmelev and Powell, 2006	Assessment of regional solid waste management system model	GIS, LCI and MODM
Chang et al., 2009	System-based approach to help fair fund distribution for incineration	GIS, MCDM and EIA
Chang et al., 1997c	Model for scheduling and collection vehicle routing for solid waste management system	GIS and MODM
Chang and Lin, 1997b	Model to assess regional waste management, using GIS as a preliminary screening tool to identify potential transfer stations location	GIS and optimization
Chang and Lin, 1997a	Optimal model to locate transfers station	GIS and optimization
Su et al., 2007	To solve insufficiencies in policy impact analysis used for decision-making	PIPA and MCDM
Sundberg and Wene, 1994	MIMES/Waste in Sweden Model – developed to provide and assist decision about systems with nested materials and energy flows	MFA, EFA (energy flow analysis) and optimization model
Döberl et al., 2002	To evaluate the goal defined in the Austrian waste management Act	MFA, CBA and MCEA
Wenig et al., 2005	A cost-effective planning tool for local governments and regional waste management systems to assist in managing risk associated with landfills	RA and GIS
El Hanandeh and El-Zein, 2009	An integrated, stochastic multi-criteria decision-making tool developed to analyze the carbon credit potential of a MSW	LCA and MCDM

Reference	Scope	Methodology
	management systems different strategies	
Lahdelma et al., 2002	To locate a waste treatment facility	EIA and MCDM
Lu et al., 2009	Dynamic optimization for solid waste management in association with greenhouse gas emission control	LCA and LP under uncertainty

2.5 TYPES OF SYSTEM ANALYSIS PLATFORMS

2.5.1 Management Information Systems

The MIS can be defined as an organized combination of people, hardware, software, communications networks and data resources that collect, transform, and disseminate information in an organization (Kumar and Mittal, 2004; Whitten and Bentley, 2008). To be specific, a MIS can be defined as an information system that provides management-oriented reporting based on transaction processing and operation of the organization (Whitten and Bentley, 2008). It can be implemented for varying sizes of SWM systems to deal with a particular problem such as waste collection. First, an organization in a SWM system can be a waste management unit/company that manages part of the waste streams, but can also be the local, regional, national and international waste management agencies, responsible for assessing and controlling wastes and resources management – in a word, mainly policy makers. Second, waste management companies may need to make plans and decisions (i.e. a top-down approach) for their company to be economically viable.

Recently, electronic data exchange (EDX/EDI) and GIS applications in SWM systems have become available in the MIS regime to account for both temporal and spatial variations. EDX/EDIs are one type of MISs, which uses a common language to exchange/interchange data electronically. The most widely applied language is XML (Extensible Markup language). A typical case of application of EDI for SWM is the EIONET (EIONET, 2009). Such a MIS is composed of directory, repository, registry and parameter services (EIONET, 2007a-2007l). The processes incurred in a MIS are collection of data, storing of data, processing of data and transmission of information (Kumar and Mittal, 2004). As it is possible to see, EDX/EDI exchange information and data between actors and agents and there is an important part of the data/information relative to geographic items. The MIS tool typically used to deal with data/information of this nature is GIS to handle spatial information. The input data can be provided by other MIS elements (like GIS) or simple online databases. Also the construction of waste exchange platforms (the Waste Exchange, 1999; BSC, 1990; OCETA, 1997; Massachusetts Materials Exchange, 2004; CIWM, 2003; CIWMB, 1995; Denmark Waste

Exchange, 2008; IHK Recyclingborse, 2008; IWEN, 2008; Jean-Gerard, 2008) where owners and buyers can exchange information and commercialize waste are also an example of EDX applied to SWM.

GIS is an information technology which stores, analyses and displays both spatial and non-spatial data, and may be extended to act as a decision support system involving the integration of spatially referenced data in problem-solving environment (EEA, 2003). It is normally composed of a data input subsystem, a data storage and retrieval subsystem, a data manipulation system and analysis subsystem, a data reporting subsystem and a subsystem responsible for the graphical user interface interacting with the user and the programming language within the GIS environment (EPA, 2002). The type of data or information that can be traded through GIS are documents, online queries for information collection, data definition and tools storage, visualization tools for data, workflow mechanisms and enhanced email support (EPA, 2002). GIS is effective in handling complicated spatial information that is essential for many environmental studies, as well as providing platforms for integrating various models, systems, and interfaces (Lovejoy, 1997; Huang *et al.*, 1999). Such data streams can be related to waste generators, disposal facilities and workplaces, presenting waste types, quantities, composition and analysis realized.

2.5.2 Decision Support Systems and Expert Systems

Supporting decision making requires understanding of the various processes involved to enable computer-based system support to be designed and increase their efficiency (Lukasheh *et al.*, 2001). The DSSs are computer-based information systems which have been designed to affect and improve the process of decision making. They underline the ideas that collectively use data and models to solve unstructured problems (Sprague and Carlson, 1982). DSS may consist of three parts: 1) an interactive graphic display capacity for managing the interface between the decision makers and the system; 2) a data management system (DMS); and 3) a model base management system (MBMS), which aggregates different models, such as optimization models, forecasting models, and simulation models. The DSS components described each have their own mode of interaction, with a higher information change. DMS is capable of supplying information to MBMS and after completion this information can be returned to the DMS to be stored. But data can be changed and updated by users through the interactive graphic display. While the DMS may be the same as a MIS, the interactive graphic display capacity may be configured in a GIS environment. Applying and developing DSS for SWM can be justified by the need to solve unstructured, semi-structured and structured problems. Such model makes possible the construction and evaluation of arguments both for and against competing courses of action (MacDonald, 1996b).

Other types of DSS models might include a fourth part, related to a knowledge-based system, with the intention of helping to estimate input parameters and helping to interpret modeling results (Lukasheh *et al.*, 2001). Such knowledge-based systems may be called expert systems (ESs). An ES is a computer program which is designed to imitate the advice of a human expert. It aims to draw conclusions from information where there is not a precise, unambiguous answer (AEA Technology, 1998). Thus, an ES consists of three components: 1) a knowledge base; 2) an inference engine which applies built-in rules (often rather rough rules of thumb) to the knowledge base to draw conclusions; and 3) a user interface, which enables the user to ask questions and understand the answers. The way that such components work together has been described by Lukasheh *et al.* (2001). ES categories can be divided in rule-based, knowledge-based, neural networks, fuzzy, object-oriented, case-based reasoning, system architecture development, intelligent agent, modeling, ontology and database methodology (Liao, 2005). Once an interaction with the ES is initiated, the inference engine searches for matching patterns. This mechanism compares information supplied by the user with the knowledge contained in the knowledge base and deduces whatever conclusion may logically follow (Lukasheh *et al.*, 2001). During this interaction, a working memory holds all information supplied by the user or deduced by the system's inference mechanism, while working on the knowledge base (Lukasheh *et al.*, 2001). A case-based ES in SWM can be developed through the acquisition of relevant data and information providing the planner with technical information that may not be readily available. For example, an ES database was used to characterize a waste stream, and estimate implications concerning transport, processing and disposing of materials and waste (MacDonald, 1996b). Table 2.5 summarizes the applications of EDI/EDX and GIS, DSS, and ES for SWM.

Table 2.5 A summary of MIS, DSS, and ES applied for solid waste management

Reference	Scope	Methodology
Chiueh and Yu, 2006	An integrated framework of solid waste management information system, with applications in public or private sectors and partial contribution of sustainable development indicators	MIS
Hřebíček *et al*, 2003	Slovak waste information system to support data collection related to waste management	MIS
Chang *et al.*, 2001	Internet-based management information system for scrap vehicle management	MIS
Şener *et al.*, 2006	Landfill site selection	GIS
Chang *et al.*, 1997c	Scheduling and vehicle routing in a collection waste system	GIS
Ghose *et al.*, 2006	GIS optimal routing model to determine the minimum cost/distance efficient collection paths for transporting solid waste to landfills	GIS/DSS
Karadimas and Loumos, 2008	GIS based model developed to establish a waste collection system considering waste generation parameters	GIS
Chang and Wang, 1996d	Solid waste management system planning	DSS
Haastrup *et al.*, 1998	Model for evaluating policies for service organization of the collection and for identifying areas suitable for locating waste treatment and disposal plants	DSS
Bhargava and Tettelbach, 1997	Model to help recycling system on World Wide Web	DSS
Simonetto and Borenstein, 2007	SCOLDSS – a DSS applied to the operational planning of solid waste collection systems	DSS
MacDonald, 1996b	Model to assist in improving solid waste the decision-making process	DSS and ES
Barlishen and Baetz, 1996	Model for planning a MSW management system	DSS and ES
Wey, 2005	Model to support waste incineration siting problems	DSS and ES
Basri and Stentiford, 1995	Guidelines for ES application to solid waste management	ES
McCauley-Bell, and Reinhart, 1997	Methodology for MSW composition studies	ES
Rubenstein-Montanto and Zandi,1999; Rubenstein-Montano, 2000	Solid waste management policy planning	ES

2.6 TYPES OF ANALYTICAL TOOLS FOR SYSTEM ASSESSMENT

The classification of analytical tools for system assessment includes: 1) SD, 2) MFA, 3) LCA, 4) RA, 5) EIA, 6) SEA, 7) SoEA, and 8) SA. They are complementary in many real world applications. A summary of all the contemporary assessment tools for various process assessments would be very helpful for model synthesis and integration when dealing with a variety of SWM systems in different countries.

2.6.1 Scenario Development

Scenarios are hypothetical sequences of events constructed for the purpose of focusing attention on causal processes and decision points (Kahn and Wiener, 1967). A more recent definition of scenario refers it as archetypal descriptions of alternative images of the future, created from mental maps or models that reflect different perspectives on past, present and future developments (EEA, 2000). Such definitions emphasize the future image concept and associated events, expected and unexpected ones, but also bring out the notion that scenarios are not predictions or projections. Scenario development therein is thus a system analysis tool to make visions of future SWM conditions in order to assess some prescribed problems that might happen in the future. Such a methodology is able to show how alternative policy decisions may reach specific goal and purpose given the resources availability and limitations. Scenario development (or scenario building) can be divided in two steps: the scenario design step, where driving forces, events and trends are established to construct the scenario; and scenario calculation, where models are used to finish the scenario, bringing more information to characterize it. Such a methodology utilizes the synergy of all types of systems engineering models collectively or separately to address the scenario. Different subdivisions of scenarios exist such as forecasting v.s. backcasting, descriptive v.s. normative, exploratory v.s. anticipatory, exploratory v.s. predictive v.s. normative, baseline v.s. policy, and quantitative v.s. qualitative.

Forecasting vs. backcasting, descriptive v.s. normative, exploratory v.s. anticipatory are all defined considering the way that the future is created and handled. Forecasting starts at the current situation with or without the expected/desired policy efforts. Backcasting starts at a desired future situation and offers a number of different strategies to reach this situation (EEA, 2000). Descriptive v.s. normative scenarios may also be defined through the driving forces that are brought into the scenario. Descriptive scenarios sketch an ordered set of possible events irrespective of their desirability or undesirability, while normative scenarios take values and interests into account (EEA, 2000). The exploratory vs. predictive v.s. normative approach is a more recent classification proposed by Börjeson et al. (2006) with sub-classes. Predictive scenarios aim to predict what is going to happen, which means that they include forecast and what-if scenarios types. Explorative scenarios try to explore the future from a variety of perspectives, including what-if scenarios but only those with long-term horizon and with profound changes. They can be divided into external and strategic sub-classes. In this context, normative scenarios are scenarios that take their starting point into account as one or several well-defined targets; here the backcasting scenario is one kind of normative, transforming scenario. Baseline and policy scenarios are classifications referred to by EEA (2001) where baseline scenarios are also known as reference or benchmark or non-intervention scenarios, representing the future state of society and the environment in which environmental policies either do not exist or do not have a discernable influence on society or environment. These scenarios also answer to “what-if” scenarios, i.e., predictive scenarios if they do not consider any kind of event. Policy scenarios are designed for future effects of environmental protection policies, being known as pollution control, mitigation or intervention scenarios — also defined as forecasting scenarios. All of these definitions overlap to some extent without well defined boundaries mainly due to their complexity. The exceptions are quantitative and qualitative scenarios, given that the qualitative scenarios are the ones that describe possible futures in the form of words or visual symbols, and quantitative scenarios require numerical information to delineate the future (EEA, 2001).

Applications of SD can be found in several contemporary environmental issues, such as global environmental change (Alcamo and Kreileman, 1996; Leemans et al., 1996; Alcamo et al., 1996, 2000), world water management (Cosgrove and Rijsberman, 2000), global greenhouse gas emissions (Nakicenovic et al., 2000), and future lifestyle trends and forecasting based on lifestyle scenarios in relation to the future of waste composition (Fell and Fletcher, 2007). But many systems analyses for SWM call for combined scenarios to address a particular concern. For example, the applications of SD technique for SWM focused on scenario definition and forward evaluation through techniques like SEA, LCA and optimization models. The perspective that SD is a much needed technique is its ability to explore events (events in this case are policies and decisions taken) that might occur associated with SWM on a temporal scale. Such events can be external or internal to the frontiers of the SWM system. To illuminate, the cases in Table 2.6 from which some SD practices have been applied to handle systems analysis for SWM are characterized by the typology of exploratory versus anticipatory, baseline versus policy, and quantitative versus qualitative. When applying this approach to deal with SWM, SD is not often characterized or described explicitly, which means that is not completely obvious how authors built them. In Table 2.6, however, it seems that the explorative scenarios are relatively popular. This probably due to the fact that the explorative scenarios allow more speculative scenarios than their anticipatory counterparts. However, an anticipatory approach could be more interesting for improving SWM perspective, since more demanding scenarios could be constructed (i.e., higher recycling targets, zero landfill), helping decision makers to prevent negative impacts from happing and prepare for meeting the sustainability goals.

Table 2.6 A summary of scenario development applied for solid waste management

Reference	Scope	Exploratory vs. anticipatory	Baseline vs. policy	Quantitative vs. qualitative
Dornburg and Faaij, 2006	To identify optimal biomass and waste treatment strategies for the Netherlands in order to save primary energy efficiently with regard to energy and costs	Anticipatory	2 baseline and 9 policy	Combined
Ljunggren, 2000	Strategic planning of Swedish solid waste management system	Exploratory	1 baseline and 2 policy	Combined
Nilsson *et al.*, 2005, Björklund and Finnveden, 2007	Swedish waste incineration tax proposal, including life cycle environmental impacts	Both	1 baseline and 3 policy	Combined
Salhofer *et al.*, 2007	Waste management plan of Province of Salzburg assessed by SEA methodology	Exploratory	1 baseline and 8 policy	Quantitative
Li and Huang, 2006	Scenarios applied to assess different waste management policies determined by an inexact two-stage mixed integer linear programming method	Exploratory	1 baseline and 2 policy	Quantitative
Chang *et al.*, 2005	To address the optimal site selection and capacity planning of MRFs in conjunction with an optimal shipping strategy of solid waste streams in a multi-district urban region	Exploratory	1 baseline and 4 policy	Quantitative
Ulli-Beer *et al.* 2007	To enhance the understanding of different pricing systems of solid waste management: effect on recycling behavior of citizens, effect on the budget goals, and operational logic of different economic measures	Both	4 Policy	Quantitative
Skordilis, 2004	Strategic planning of an integrated solid waste management at local level	Exploratory	2 policy	Quantitative
Chang and Lin, 1997a	Scenarios applied to assess optimal siting of transfer stations	Exploratory	1 baseline and 2 policy	Combined
Villeneuve *et al.*, 2009	Applied to assess a collection basin from a waste management system	Exploratory	1 baseline and 4 policy	Combined

However, many assessment/quantification models applied for scenario assessment were formulated by a slightly unique way. In these studies, a scenario term is used concerning alternatives that are assessed by determined model, not being related to events consequences or driving forces that have defined such scenario, or even to a future time frame. For example, scenario term is currently applied in LCA models where the only purpose is to assess different waste management options. Some of the developed model was applied to assess waste management scenarios by a retrospective approach (Thorneloe et al., 2007).

2.6.2 Material Flow Analysis

According to Brunner and Rechberger (2003), material flow analysis (or accounting) is a systematic assessment of the flows and stocks of materials within a system defined in a space and time. It connects the sources, the pathways, and the intermediate and final sinks of a material management (Brunner and Rechberger, 2003). Because of the law of the conservation of matter, the results of a MFA can be controlled by a simple material balance comparing all inputs, stocks, and outputs of a process management (Brunner and Rechberger, 2003). MFA have somewhat left the traditional SWM boundary, focusing on product consumption patterns, waste generation, recycling, recovery, and reuse. It is this distinct characteristic of MFA that makes the method attractive as a decision-support tool in resource - waste -, and environmental management (Brunner and Rechberger, 2003). MFA can also be designed to understand the material flow that occurs during different phases of the product life relating it to temporal aspects so as to predict when it will become waste and in which phase of its waste life it will be standing. There exist three methods to make MFA practical (Brunner and Rechberger, 2003). The first method is directly designed for addressing waste composition (sampling and waste characterization, including chemical analysis. The second one focuses on market product analysis, which requires information related to goods production and destination during their consumption. The third method is related to indirect analysis linking waste treatment with waste composition. The advantage of the third method is that the outputs of the process are less heterogeneous than waste inputs. In general, process-based MFA is primarily used to analyze specific questions of resources and waste management and industry-based MFA focusses more on the environmental impact of economic development by analyzing total material throughput in a system (Porter *et al.*, 2005). 10 out of the 15 waste studies were considered to be usable in policymaking in some way (Wiedmann *et al.*, 2006).

Some MFA applications for SWM are summarized in Table 2.7. Selecting the example of plastics waste, the data used to forecast waste production were consumption of plastics products, quantity of plastics used and residence time (Patel *et al.*, 1998). In the case study developed by Liu *et al.* (2006), on the other hand, the data considered were waste possession, obsolete ratio, population, sales and number of households linking anthropogenic metabolism, meaning that it works based on economic principles, in the nexus of industrial ecology, economic planning, and waste management. These types of practices lay down the foundations of life-cycle assessments, eco-balancing, environmental impact statements, and waste management collectively. A number of models have been developed and dedicated to substances and products analysis tools – integrated SFA, MFA and LCA (Boelens & Olsthoorn, 1998). Extended studies, such as SFINX (van der Voet *et al.*, 1995a; 1995b), FLUX (Huijbregts, 2000), Gabi (PE International, 2006), DYNFLOW (Elshkaki, 2000) and UMBERTO (IFU, 2006), developed decision support tools based on the concept of MFA for waste management applied in a growing economic system (Chanchampee and Rotter, 2007). For example, the SFINX (Substance Flow InterNodal exchange) computer program is a tool to assist in substance flow analyses (van der Voet *et al.*, 1995a; 1995b).

Table 2.7 A summary of material flow analysis applied to municipal solid waste

Reference	Scope	Methodology
van der Voet *et al.*, 1995a,1995b	Substance flow through economy and environment	MFA
Tasaki *et al.*, 2004	Waste TV pollutants assessed to inform decision in Japan	Time-series MFA
Lang *et al.*, 2006a, 2006b	Analysis for recycling schemes of biowaste in Canton of Zurich	MFA
Streicher-Porte *et al.*, 2005	WEEE management system in Delhi, India	MFA
Tian *et al.*, 2007	Method developed to select the optimal treatment and disposal technology of SWM	Elemental stream analysis
Wiedmann *et al.*, 2006	Policy analysis of waste materials	MFA
ifu, 2006	Material and energy flow calculation	MFA and LCA
Frakgou *et al.*, 2009	To develop a indicator of self-sufficiency MSW management	MFA
Eckelman and Chertow, 2009	Analysis of material flows to obtain long-term strategies for diminishing waste generation on an island	MFA
Mastellone *et al.*, 2009	Quantification and assessment of a MSW management system scenario	SFA
Sokka *et al.*, 2004	Municipal waste system in 1952-1999 assessed through N and P	MFA

The FLUX model focuses on the emission impacts (Huijbregts, 2000). DYNFLOW is a modeling system that features both MFA and LCA together (Elshkaki, 2000), Gabi (PE International, 2006), and UMBERTO is designed for material and energy flow calculation. In addition to industrial applications (ifu, 2006), MFA has also been used to quantify and compare the environmental impacts caused by several modern household waste management strategies in the federal state of Baden-Württemberg, Germany (Escalante *et al.*, 2007).

2.6.3 Life-cycle Assessment

The LCA is a process to: 1) evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying the energy and materials used, wastes and emissions released to the environment; 2) assess the impact of those energy and material uses and releases to the environment; and 3) identify and evaluate opportunities that lead to environmental improvements (EEA, 2003). Environmental impacts in relation to air, water, and land pollution via resource exploitation and pollutant emissions across the product's life cycle are of concern. According to the International Organization for Standardization (ISO 14040, 2006), LCA addresses the environmental aspects and potential environmental impacts (e.g. use of resources and the environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave). This can be done through four phases: goal and scope definition, inventory analysis, impact assessment and interpretation. Goal and scope definition intends to define the purpose, specifications and limits to be considered in the assessment. Inventory analysis phase is responsible for the collection of data of the unit processes within the system and relating it with a functional unit. Impact assessment intends to make inventory information more understandable through its translation into environmental impact categories. Final interpretation allows evaluating results obtained and comparing them with the initially defined goal (ISO 14040, 2006).

Besides its original application to products, LCA was also applied to processes and systems, including SWM. LCA started to be applied for SWM because there was an acute to understand how to deal with solid waste with less environmental impacts. Nowadays, the applications of LCA as part of the systems analysis for SWM mainly concern the best management practices or decisions required when considering the least environmental damage. The decisions mainly concern technology screening related to the lifecycle steps, namely collection, treatment, and final disposal. Thus, a LCA can result in a comparative assessment of environmental impacts from different scenarios defined and analyzed.

Scenarios used in an LCA are related to alternatives that are going to be analyzed, which might be associated with the scenarios from SD described above. As a consequence, an LCA offers a system map that sets the stage for a holistic approach, thereby comparing such system maps with different options. No matter whether they are prepared for different products or SWM systems, environmental improvements can be made possible (McDougall *et al.*, 2001). Several LCA models were tailored specifically for SWM systems, including IWM, WASTED (Diaz and Warith, 2006), WISARD (Ecobilan, 2004; Buttol *et al.*, 2007), and EASEWASTE (Christensen *et al.*, 2007). Specifically, the IWM versions 1 and 2 for SWM systems provide LCI. The models enabled decision makers and waste managers to use an LCA for their specific waste management configurations without in-depth knowledge of the methodology and allowed them to learn how changes in the system affect the environmental impacts through scenario analysis (Winkler and Bilitewski, 2007). However, generic models, like UMBERTO (ifu, 2006), Gabi (PE International, 2006), and SimaPro (Bovea and Powell, 2006), were applied to SWM, with heightened potentials concerning waste management specificities. In particular, the German LCA-software GaBi developed by the Department of Life Cycle Engineering of the Chair of Building Physics at the University of Stuttgart in cooperation with PE International GmbH (PE International, 2006) is a tool to address the need for sustainability data administration and evaluation on the organization, facility, process or product life cycle level. The advantage of later type assessment models is that we can build a complete model with all of the relevant processes and mass flow involved and it is not necessary to deal with a pure SWM system (Winkler and Bilitewski, 2007). On the other hand, specific LCA models for ISWM systems are easier to use and more intuitive. A few cases are shown in Table 2.8 which cover LCA, LCI and lifecycle impact assessment (LCIA).

Table 2.8 A summary of life cycle assessment applied for solid waste management

Reference	Scope	Methodology
Banar *et al.*, 2009	LCA to assess solid waste management options	LCA – SimaPro
Bovea and Powell, 2006	Assessment of alternatives for solid waste management system to reach targets required by European Directives in the Community of Valencia, (Spain)	LCA – SimaPro
Bovea *et al.*, 2007	To assess the integration of transfer stations within a waste management system considering environmental factors	LCA – SimaPro
Buttol *et al.*, 2007	WISARD model application to show decision-makers at political level the benefits obtainable with the use of LCA	LCA – WISARD
Diaz and Warith, 2006	WASTED model development	LCA – WASTED
Emery *et al.*, 2007	WISARD model application to assess waste disposal scenarios	LCA – WISARD

Reference	Scope	Methodology
Kirkeby, *et al.*, 2006	EASEWASTE model development	LCA – EASEWASTE
Kirkpatrick, 1993	Assessment of waste hierarchy applied to mixed waste paper and HDPE	LCA
Liamsanguan and Gheewala, 2008a,b	To assess several scenarios related to solid waste management, only considering global warming potential (GWP) impact	LCA
White *et al.*, 1995	IWM-1 model for solid waste system	LCI – IWM
Özeler *et al.*, 2006	IWM-1 model applied to assess SWMS in Ankara	LCA – IWM
Powell *et al.*, 1998	SWM strategies assessment	LCA
Rieradevall *et al.*, 1997	Landfill environmental impacts assessment	LCA
Song *et al.*, 1999	PET recycle routes assessment	LCA
Xará *et al.*, 2005	SWM assessment	LCA
Luoranen *et al.*, 2009	To assess different energy recovery options in a waste management system (MSW)	LCA
Rigamonti *et al.*, 2009a;2009b	Analysis of material and energy recovery within an integrated MSW management system concerning recovery of source-separated collection and energy recovery from residual waste	LCA – SimaPro
Wittmaier *et al.*, 2009	Analysis of thermal treatment and energy recovery from waste management system	LCA – GaBi
Tan and Khoo, 2006	To evaluate various waste management options in Singapore	LCA – SimaPro
Zhao *et al.*, 2009	Evaluates the current and possible patterns of MSW management with regard to GHG emissions	LCA
de Feo and Malvano, 2009	Assessment of several MSW management systems	LCA – Wisard
Batool and Chuadhry, 2009	Study of alternative scenarios to manage MSW	LCA
Morris, 2005	Evaluate the environmental burdens associated with collection and management of MSW	LCA
Beigl and Salhofer, 2009	To compare different waste management systems	LCA
Miliūtė and Staniskis, 2009	LCA to optimize a MSW management system	LCA
Schmidt *et al.*, 2007	To assess waste hierarchy applied to waste paper	LCA – SimaPro
Bergsdal *et al.*, 205	Environmental assessment of two waste incineration strategies	LCA

2.6.4 Risk Assessment

Risk Assessment is a broad term covering many different types of assessments (Finnveden *et al.*, 2007). In the context of system analysis, risk assessment (or environmental risk assessment, or risk analysis) can be defined as a method which assesses the possible damage to a system. Such assessment starts at a risk source, either an operational unit, an infrastructure, and follows the consequent chain of accidents that might happen, ending up assessing possible damage caused to the population and environment. In this case, risk assessment from a system perspective is to relate environmental and human health risk to accidents quantitatively. Required RA was included in the SEVESO directive and other directives in the European Economic Community (EEC) related to dangerous substances and environmental accidents although harmful impacts from waste management are not their major concern. In comparison with other system analytical methods, environmental risk assessment (ERA) or comparative environmental risk assessment (CERA) evaluates and assesses transversal system to waste management flow. An ERA consists of the identification and evaluation of ecological risk and public health risk effects from a dangerous source and environmental receptor related to exposure factors and their attendant consequences during a specified period of time.

Infrastructures like landfill, incineration, anaerobic digestion, and composting plants can have important emissions to the environments, and explosions and fires during operations. For waste management, ERA methodologies may be applied to landfills, incineration and composting plants for exposure assessment. With the increase of different type of infrastructure and treatment options, it is important to understand if these will result in an increased risk of accidents and environmental releases of specific substances associated with differing scenarios. These feedbacks would be helpful in some modeling analysis for siting and sequencing those waste management facilities. From a strategic point of view, an ERA can be applied to assess the location of those facilities such as landfills, making up part of the EIA (when projects are obligated to that). This “expert-challenging” role requires that the risk assessment process be open to public involvement and influence (Petts, 2000). In this regard, ERA can also be an integrative tool due to the possibility of bringing public participation into the waste management system. During an EIA, RA can be requested by the public, especially in cases of incineration location decisions. The public is capable of testing the credibility of the management procedure through a form of quality assurance (Petts, 1997). On some occasions, RAs are wanted even though plants will operate according to emission limits that follow the precautionary principle and where local conditions can influence design (Petts, 2000). Besides, RA can also be applied to those products or byproducts resulting from waste treatment, such as recyclables, refuse derived fuels (RDF) and compost. In the cases of compost, for instance, several quality assurance systems were created all over Europe in such way as to control compost quality considering risk relevant to environmental and public health.

Like other methods, an ERA also has specific software tools especially for designing waste management infrastructures, like landfills – GASSIM, LANDSIM and other risk assessment models like EHHRA-GIS (Environmental and Human Health Risk Assessment) combining risk assessment and GIS, ROME (ReasOnable Maximum Exposure) for human health risk assessment developed by the Italian Environmental Protection Agency. One case in which such software tools were applied to landfill site selection is Nakaishi *et al.* (2005); Bote *et al.* (2003) assessed the gas risk from landfills. Cangialosi *et al.* (2008) and Snary (2002) have assessed the health risk from a municipal solid waste incineration plant. Harrop and Pollard (1998) quantified the risks applied to incineration environmental impact.

2.6.5 Environmental Impact Assessment

An EIA is a procedure that aims to ensure that the decision-making process concerning the proposed activities may have a significant influence on the environment (Tukker, 2000). To proceed to an EIA it is necessary to perform a systematic process that examines the environmental consequences of development actions in terms of physical, biological, cultural, economic and social factors (Lenzen *et al.*, 2003). There are a set of steps to be followed (Glasson *et al.*, 2005): project screening, scoping, consideration of alternatives, description of the project/development action, description of environmental baseline, identification of the main impacts, prediction of impacts, evaluation and assessment of significance, identification of mitigating measures, public consultation, Environmental Impact Statement (EIS) presentation, EIS review, decision-making of the project, post-decision monitoring, audit of predictions, and mitigation measures. In fact, this methodology requires public participation but also approval from authorities, being a process that gives relative assurance of the consistency of the method to all stakeholders. It can also promote compensatory measures in cases where environmental impacts cannot be eliminated (Lenzen *et al.*, 2003). The assessment provided can be made through scenario/alternatives comparison finally. The results from this method can be an important contribution to solve controversial issues from the target project such as siting issues originated from the NIMBY effect, technical issues in justifying the choice of technology for emission reduction, and even the non-approval of the project. A good example can be found in Barker and Wood (1999). Saarikoski (2000) have applied EIA to assess waste management strategy at regional level.

2.6.6 Strategic Environmental Assessment

SEA can be simply defined as the environmental assessment of a strategic action such as a policy, a plan or a program (Thérivel and Partidário, 1999). Such an environmental assessment is a formalized, systematic and comprehensive process of evaluating the environmental effects of a policy, plan or program and its alternatives, including the preparation of a written report on the findings of the evaluation, and using the findings in publicly accountable decision-making (Thérivel *et al.*, 1992). According to Nilsson *et al.* (2005), SEA steps can be explained in the following way: scoping intends to handle what to include in the SEA, the temporal and spatial boundaries, the institutional context and decision scope, and delimitations in terms of issue coverage and stakeholder participation. Yet such a procedure is not rigid, and can be rearranged to fit specific cases. Decision analyses may then generate the decision alternatives for analysis in close deliberation with the decision-makers, often through applying a scenario analysis. In some cases, it is possible to introduce 'sustainability alternatives' as part of the package.

The applications of a SEA for SWM were emphasized by some countries as described in EU Directive 2001/42/EC, to which it is obligated for the promotion and elaboration of a SEA for SWM plans. The need to assess plans and programs is because it has become clear that such decisions have important environmental impacts that should be considered when they are being produced instead of being conducted, when potentially less damaging possibilities can be performed. Therefore, SWM systems are also obligated to be assessed through a SEA. Besides plans, a SEA was even applied to assess economic instruments, like the incineration tax. Most of these applications did apply the SEA along with other methods, like GIS, LCA, RA, CBA, MCDM, and SD together. This is due to the nature of a SEA procedure that needs to have more specific and quantifiable information to provide the assessment. More detail can be found out in the Dutch Ten Year Program on Waste management 1992 and 2002 (Verheem, 1999). Also in Federico *et al.* (2009) a practical problem of analyzing an integrated provincial solid waste management system can be found.

2.6.7 Socio-economic Assessment

Social impacts include non-technical indicators and criteria such as employment, public health, willingness to pay, odors, noise, traffic vehicles, and public participation. Socio-economic assessments are practices that apply integrated market-based and/or policy/regulation requirements for SWM such as Waste-to-Energy (WTE) taxation. The way that such system engineering models and assessment tools, like LCA, IMS, MFA, and SD, can perform largely fits in this mission. In the case of optimization analysis in the context of a full-cost accounting approach, the inclusion of these socio-economic factors into the models can be done through the use of financial objectives and/or constraints. For example, such applications include but are not limited to CBA-based linear programming (Chang *et al.*, 1997a; Chang *et al.*, 1996), CBA-based integer programming (Chang *et al.*, 2005), CBA-based fuzzy goal programming (Chang and Wang, 1997a), fuzzy contingent valuation method for fair fund distribution (Chang *et al.*, 2009), GIP-based game theory for landfill space pricing (Davila *et al.*, 2005), optimal control of landfill space consumption (Chang and Schuler, 1991), and CBA-based MCDM (Karagiannidis and Moussiopoulos, 1997; Rousis *et al.*, 2008). They can also be linked with regulations in a wealth of SWM issues that expand the nature of these assessments such as DSS (Fiorucci *et al.*, 2003; Costi *et al.*, 2004), multiobjective programming (Minciardi *et al.*, 2008), as well as the quality assurance requirements system products, like RDF. Table 2.9 summarizes the recent trend in this regard.

Table 2.9 Socio-economic assessment applied to solid waste management

Reference	Scope	Methodology	Instruments
Jacobs and Everett, 1992	Optimize SWM	Linear programming	Landfill tipping fees
Karagiannidis and Moussiopoulos, 1997	Multi-criteria application to large SWM system	MCDM (ELECTRE method)	Landfill tipping fees
Ljunggren, 2000	MIMES model	Linear programming	Landfill tipping fees
Chang and Davila, 2006	Optimization model for routing and possible landfill/incinerator construction	Grey minimax regret integer programming	Landfill tipping fees
Chang and Davila, 2007	Model to improve SWM strategies	Minimax regret optimization programming	Landfill tipping fees
Chang *et al.*, 1997b	Optimization model for SWM system	non-linear and integer programming	Landfill tipping fees
Chang and Wang, 1997a	Planning SWM	Fuzzy goal programming	Tipping fees, air pollution emission limits, noise limit values
Chang *et al.*, 1997d	Optimization model for assessment management strategies for SWM	FIMOMIP	
Daskalopoulos *et al.*, 1998[a]	Theoretical model for the management of MSW streams	Linear programming	Air emissions from incinerator
Chang *et al.*, 1996	Location/allocation model for SWM	Mixed integer programming with framework of dynamic optimization	Landfill tipping fees, noise limit value
Chang and Wang, 1996c	Optimization model to facilitate evaluation of compatibility issue between waste recycling and incineration	Fuzzy goal programming	Tipping fees
Najm *et al.*, 2002	Optimization model to evaluate alternatives and obtain optimized	Linear programming with dynamic modeling	Fees (optional)

Reference	Scope	Methodology	Instruments
	combinations of technologies		
Fiorucci *et al.*, 2003	Model to help in decision making in SWM	DSS model	Normative constraints like waste recycling level minimum, RDF quality minimum
Minciardi *et al.*, 2008	Model to support the decision on the optimal flow of SW sent to different processes	Non-linear multiobjective programming	Regulation requirements: minimum recycling rate, environmental constraints related to air emissions
Costi *et al.*, 2004	Model to help decision makers	DSS with non-linear optimization programming	
Nilsson *et al.*, 2005	Analytical framework to evaluate policy proposals, in this case WTE taxation	LCA and other models (site-dependent pathway and a qualitative pathway)	WTE taxation
Björklund and Finnveden, 2007	Assessment model related to economic and environmental impacts for the introduction of a weight-based tax in waste incineration	LCA and SEA	WTE taxation
Louis and Shih, 2007	Model to support decision for recycling system to assess the feasibility of implement a inventory warehouse for recyclables	NL dynamic programming	Landfill tipping fees, some material specific
Lu *et al.*, 2009	Waste model	Dynamic optimization programming	Regulation requirements concerning air pollutants emission
Moutavtchi *et al.*, 2008	WAMED – Methodology to assess environmental and economical issues for SWM	Full-cost accounting and CBA	Fees received by waste management companies from polluters, fees collected for waste removal and treatment, remediation of depleted resources and polluted environment
Lang *et al.*, 2006	Model developed to assess and identify parameters and mechanisms involved biowaste separation	MFA	Fees for garbage-bags/curbside collection, fees for biowaste collection
Ulli-Beer *et al.*, 2007	To assess main SWM driving forces	Dynamic system	Garbage bag charge, prepaid tax
Rousis *et al.*, 2008	To assess WEEE management system	MCDM	Harmonization with existing institutional/legislative frame. Application of priorities of legislation
Komilis, 2007	Model to optimize SWM system	Mixed linear and integer optimization	Tipping fees
Courcelle *et al.*, 1998	To assess the economic and environmental performance of MSW collection and sorting	MCDM	Pollution, noise and residues

Reference	Scope	Methodology	Instruments
	programmes		

2.6.8 Sustainable Assessment

Sustainable assessment refers to the integration of different methodologies in such a way that is geared toward obtaining an analysis, an evaluation or a planning that approaches several management aspects in which the sustainability implications may be emphasized and illuminated. Such models are different to an IMS or others in terms of the sustainability concerns. For example, the development of such a SA scheme may be motivated by taking the energy production and material recycling into account when modeling the SWM systems allowing the system planning/evaluating/analysis to become more sustainable. In particular, the UK's Waste and Resources Action Programme works with local authorities, business and households to prevent waste, increase recycling and develop markets for recycled and sustainable products that is a big database in support of SD (WRAP, 2009).

A LCA combined with other types of system assessment methods, like a MFA, allows the assessment of systems to consider new perspectives, such as sustainability implications. For example, MFA and substance flow analysis (SFA) were used together in the ORWARE model, helping to understand where substances are being concentrated. It is important when necessary to control output quality more than the assessment of environmental impacts since SFA can bring the flow of concentration or dissolution of harmful substances to a LCA when they leave the system. Proper arrangement of a LCA with MFA and energy analysis methods was made possible by Cherubini *et al.* (2008), where an SWM system was analyzed with a new perspective, (zero landfill emissions) , making environmental impact and energy balance much easier to understand and where and how the material and energy are being wasted. Other types of integrated models applicable in this regime are MCDM and policy impact potential analysis (PIPA) method, which is designed to include the policy aspect in addition to the common aspects of technical, economical, environmental and social ones being brought through MCDM (Su *et al.*, 2007). Note that a SEA is a procedure method, which needs quantifiable arguments to be used in the assessment of plans, programs and policies. LCA was used to assess environmental impacts as an integral part of SEA alternatives. The combination of SEA and LCA related to the models that are more focused on environmental assessment has different orientations. Besides, using this type of integration it was found difficult to get the public and non-expert elements of a SEA process connected to the LCA results. Bringing environmental and economic assessment together for SWM was also performed by a LCA with both aspects optimized (Solano *et al.*, 2002a,b; Harrison *et al.*, 2001) and assessed with respect to environmentally economic options (Viotti *et al.*, 2005). Under the umbrella of MCDM, a LCA brings different aspects besides economic, technical, social and environmental concerns, such as global warming potential and public health impact. Forecasting models were also combined with a LCA for making a best bet of waste to be generated. An LCA-ISWM model accounting for temporal effect falls into this category (den Boer *et al.*, 2007). In addition, an MFA can be combined with a CBA, for the optimization analysis of SWM systems (like Markal and MIMES/Waste for Sweden models). GIS combined with a LCI, an EIA, and an optimization model can represent a typical ramification in systems analysis. One salient example is landfill siting issues considering social, economic, and technical aspects simultaneously with such integration described above (Chang *et al.*, 2008; 2009). Table 2.10 summarizes all of the latest developments on this front.

Table 2.10 Sustainable assessment applied for solid waste management

Reference	Scope	Methodology
Kijak and Moy, 2004	Decision support tool to assess scenarios for ISWM	LCA and MAUT
Solano et al., 2002a,b	Model for MSW to obtain the best? bet solution to manage waste considering economic and environmental items	LCA/LCI and optimization model
Harrison, 2001	Model that considers environmental emissions and costs	LCA/LCI and optimization model
Thorneloe et al., 2007, Weitz et al., 1999	MSW-DST: a decision support tool for MSW systems	LCI and FCA
Reich, 2005	Combination of LCA and LCC to assess municipal waste management systems	LCA and LCC
Dahlbo et al., 2007	Model to study newspaper waste management options	LCIA and SLCC
den Boer et al., 2007	Model LCA-ISWM	LCA and forecasting
Salhofer et al., 2007	Assessment of waste management plan for Austria	SEA, LCA and forecasting
Tukker, 2000	LCA included at EIA procedure	LCA and EIA
Cherubini et al., 2008	Models combined to study the possibility to reduce the quantity of waste going to landfill – zero emissions principle	LCA, MFA, emergy accounting and embodied energy analysis
Dalemo et al., 1997, Björklund et al., 1999, 2000, Eriksson et al., 2002, 2005	ORWARE model development	LCA, MFA and simulation
Hischier et al., 2005	Assessment of the two take-back recycling systems	MFA and LCA
Batool and Ch, 2009	Analysis of waste management system	LCA and GIS
Purcell and Magette, 2009	Forecasting of biodegradable municipal waste generation concerning quantity and distribution within a diverse ‘landscape’ of residential areas and commercial establishments	Forecasting and GIS

More cases of integration can be found including the combination with LCA, MFA, energy accounting and embodied energy analysis, the integration of LCA with Multiple Attribute Utility Theory (MAUT), the integration of LCI with full-cost accounting (FCA), and combining ecological and economic assessment of options through the integration between LCIA and economic analysis of social life cycle costs (SLCC).

2.6.9 Comparative Analysis and Future Perspectives

Understanding how and which are the frontiers of the systems analysis is essential to select appropriate systems engineering models and assessment tools capable of achieving the management goals of SWM. The models and assessment tools shown so far have been used for analyzing a variety of SWM systems all over the world. They can be applied by considering the type of waste, time, spatial coverage, and aspects from technical, environmental, ecological, energy, economic, and social viewpoints. To broaden the SWM context, in several cases, the coverage was extended to other types of municipal waste, like sewage sludge (in case of ORWARE sustainable assessment model) and industrial waste (MIMES/WASTE model). Some models may focus on specific waste streams, such as packaging waste (CBA, SM), WEEE and specific WEEE, such as TV and refrigerators (FM, SoEA), and waste materials like paper waste (LCA, SA), plastic waste (MFA, SM), hazardous household waste (SD), and organic waste (SD, SA). Overall, the trend analysis clearly indicates that emphasis has been placed upon the systems engineering models to system assessment tools during the 1990s. However, during the 2000s, system assessment tools have gained prominence, mainly due to LCA and SA models. Fig. 2.2 and Fig. 2.3 confirm this observation. Both chorological summaries reveal a fast growth of the applications of those systems engineering models and system assessment tools in late 1990s and early 2000s.

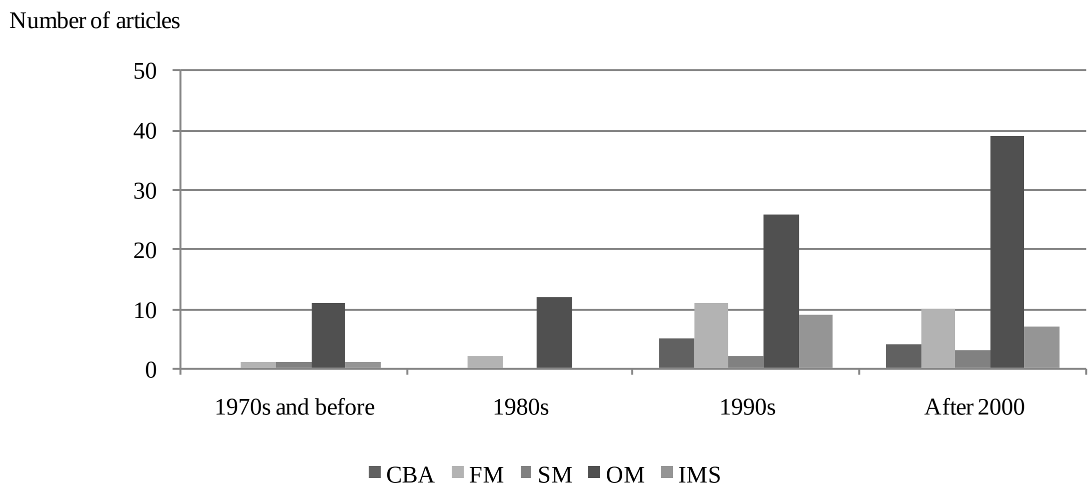

Fig. 2.2 Trends in the number of publications concerning systems engineering models in the past four decades

Varying temporal scales addressed by different models or tools result in significant traits across differing applications. MFA used to be applied to reflect environmental effects from one year to 1,000 years. A similar application was carried out using SA models from 20 years to 500 years period. Yet, handling the shorter time frame of 1 to 15 or 20 years can be seen in most forecasting models of solid waste generation. There are some cases in which the time scales are ignored completely, like EIA and MIS.

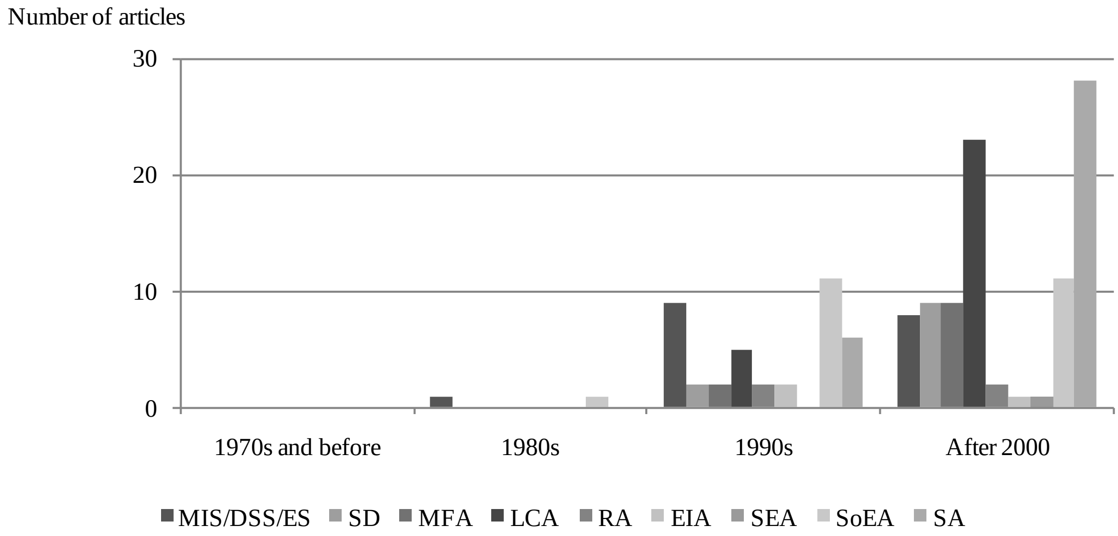

Fig. 2.3 Trends in the number of publications concerning systems assessment tools in the past four decades

In RA, however, the time scales are fully dependent on the type of risk to be assessed. On the other hand, spatial scales are considered in a more homogenous way among the developed models presented. Local scales referring to municipalities and districts, regional scales referring to metropolitan complexes, and national and international scale can be used solely for policy analysis in differing models. IMS, MFA, SD and SEA are models or tools without considering local scale due to different features. An MFA has, at least, a regional scale; otherwise, it would not be able to cover all the elements needed for the assessment in the regional SWM system. MFA models were even applied at the international level for scenario development. LCA models may deal with environmental impacts based on differing scales from global to regional. These environmental impacts are related to global warming, euthrophication, acidification, climate change, tropospheric ozone, and ecological toxicity. SEA models may also consider new elements like production of residue and use of space. Y*et al*l of the system assessment tools stand alone as a single tool to tackle a particular type of issue that could be useful with reference to a specific policy and decision-making.

From a socioeconomic point of view, many CBA, SM, OM and IMS have considered capital and operational costs and associated benefits. Forecasting models may include some socio-economic factors, such as household family size, occupation, education level, employment level, and average income features associated with population. The social aspects considered are the most heterogeneous one at least in dealing with the sustainability issue. To be specific, the intergeneration externalities are very difficult to address by a general CBA framework. It might be implicitly considered through the public participation, such as EIA, SEA and MCDM. Human health effects, which could be also considered as an environmental aspect, can be considered by LCA, MFA and RA. Overall, the publication statistics across systems engineering models and system assessment tools can be summarized chronologically by Figures 1 and 2 below. The summary reveals that more systems assessment tools appeared in the last decade whereas most systems engineering models were developed in 1990s.

Many models developed in 1990s have severe drawbacks as pointed out by Morrissey and Browne (2004). These include: 1) Most models considered economic and environmental aspects, but very few of them considered social aspects. For a SWM to be sustainable, it needs to be environmentally effective, economically affordable and socially acceptable. Y*et al*most none of the models being developed before the year 2000 considered the intergenerational effects. 2) None of them considered the complete waste management cycle, from the prevention of waste through to final disposal. Most were only concerned with refining the actual MCDM technique itself or to compare the environmental aspects among waste management options (recycling, incineration, and disposal). 3) Another identified weakness of the previous models before the year 2000 is that no model considers the involvement of all relevant stakeholders, including the government agencies, the local communities, the industrial experts, waste generators, and the formal and informal sector service providers.

Shmelev and Powell (2006) also reviewed these previous waste management models and concluded that most models do not have a holistic view over the SWM system, they tend to focus on a single problem and they are not very useful to decision makers. What is missing in a sound modeling technique for solving regional SWM problems is an all-inclusive approach, which inevitably has a large number of possible solutions. Consequently, these drawbacks could limit the power of systems analysis, and new driving forces appeared to be necessary within the 21st century. These influences led to the consideration of climate change effects, energy crisis, and the scarcity of resources as new waste management targets imposed on nations, complying with more demanding environmental regulations and emphasis on green technology, all of which may be needed in future SWM planning. Gaps in knowledge as to how to make the true system integration and system synthesis between system engineering models and system assessment tools are obvious. With such an assessment of research needs, it is expected that sound system integration and system synthesis with an all-inclusive approach will be carried out in the next decade.

2.7 CONCLUSIONS

The development of systems analysis models or tools for SWM systems over the last few decades was fully reviewed in this paper. Fourteen main categories of models and tools were clearly classified and discussed, including CBA, FM, SM, OM, IMS, MIS/DSS/ES, SD, MFA, LCA, RA, EIA, SEA, SoEA and SA. Overall, the system engineering models and tools developed in the early stages are SM, OM, FM, and CBA, followed by IMS, where market-based instruments and regulatory requirements were gradually considered in the decision making. With the later emphasis on the concept of sustainability later, these tools, such as MFA, LCA, RA, SD, SEA, SoEA and EIA with the specific applications of MIS/DSS/ES, collectively or separately, promote the sustainable planning and management of SWM. Nevertheless, the models or tools described have individual limitations and none of them has considered the complete vision of the whole waste management cycle, from prevention of waste through to final disposal, except the LCA. While improving these decision-making techniques, we suffer from being time and data consuming with respect to a varying boundaries set for different models and/or tools at differing technical, environmental, economic and social aspects. Ideal solution procedures normally yield a balance between simplifications of the analysis and the soundness of capturing the essential features resulting in additional complications in systems analysis for SWM. Future systems analysis requires conducting interdisciplinary and policy-relevant research relating to SWM systems, with the emphasis on enhancing the sustainability of systems challenged by rapid changes of societal environment and/or extreme events of global climate change. IMS will be applied on different scales in combination with different assessment tools, such as LCA, more often. Gaps in knowledge as to how to create true system integration and system synthesis between system engineering models and system assessment tools are obvious. With such an assessment of research needs, it is expected that sound system integration and system synthesis with an all-inclusive approach will succeed in the next decade. All of the efforts will certainly allow risk-informed, forward-looking, cost-effective, and environmentally benign decision making to be developed.

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CHAPTER III. LITERATURE OF SYSTEMS ANALYSIS IN SOLID WASTE MANAGEMENT IN EUROPE

3 SOLID WASTE MANAGEMENT IN EUROPEAN COUNTRIES: A REVIEW OF SYSTEMS ANALYSIS TECHNIQUES

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(accepted after revision in Journal of Environmental Management)

3.1 ABSTRACT

In the past few decades, solid waste management systems in Europe have involved complex and multi-faceted trade-offs among a plethora of technological alternatives, economic instruments, and regulatory frameworks. These changes resulted in various environmental, economic, social, and regulatory impacts in waste management practices which not only complicate regional policy analysis, but also reshape the paradigm of global sustainable development. Systems analysis, a discipline that harmonizes these integrated solid waste management strategies, has been uniquely providing interdisciplinary support for decision making in this area. Systems engineering models and system assessment tools, both of which enrich the analytical framework of waste management, were designed specifically to handle particular types of problems. Though how to smooth out the barriers toward achieving appropriate systems synthesis and integration of these models and tools to aid in the solid waste management schemes prevalent in European countries still remains somewhat uncertain. This paper conducts a thorough literature review of models and tools illuminating possible overlapped boundaries in waste management practices in European countries and encompassing the pros and cons of waste management practices in each member state of the European Union. Whereas the Southern European Union countries need to develop further measures to implement more integrated solid waste management and reach EU directives, the Central EU countries need models and tools with which to rationalize their technological choices and management strategies. Nevertheless, considering systems analysis models and tools in a synergistic way would certainly provide opportunities to develop better solid waste

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management strategies leading to conformity with current standards and foster future perspectives for both the waste management industry and government agencies in European Union.

Keyword: Solid waste management, Systems analysis, Integrated solid waste management, Sustainability

3.2 INTRODUCTION

In the 21st century, the sustainable management of municipal solid waste (MSW) will become necessary at all phases of impact from planning to design, to operation, and to decommissioning. As a consequence, the spectrum of new and existing waste treatment technologies and managerial strategies has also spanned from maintaining environmental quality at present to meeting sustainability goals in the future. Such an orderly evolution allows both waste management industries and government agencies to meet common needs of waste management with greatest green potential, to recycle materials out of waste streams, to enlarge the renewable energy supply, to seek for more socially acceptable options, and to preserve biodiversity and natural ecosystems simultaneously. To achieve such goals, all technical and non-technical aspects of a solid waste management (SWM) system should be analyzed as a whole, since they are interrelated with one another and developments in one area frequently affect practices or activities in another area (UNEP, 2005).

Systems analysis techniques have been applied to handle MSW streams through a range of integrative methodologies in the last few decades. A total of five system engineering models and nine system assessment tools were formally classified in this field to illuminate the challenges, trends and perspectives (Chang *et al.*, 2010). It is worth knowing that the spectrum of these models and assessment tools was classified based on the following two domains although some of them may be intertwined with each other (Chang *et al.*, 2010). They are: 1) systems engineering models including cost benefit analysis (CBA), forecasting models (FM), simulation models (SM), optimization models (OM), and integrated modeling system (IMS), as well as 2) system assessment tools including management information system (MIS)/decision support system (DSS)/expert system (ES), scenario development (SD), material flow analysis (MFA), life-cycle assessment or life cycle inventory (LCA or LCI), risk assessment (RA), environmental impact assessment (EIA), strategic environmental assessment (SEA), socio-economic assessment (SoEA), and sustainable assessment (SA). Fig. 3.1 holistically illustrates the interrelationships among these two domains from which fourteen technologies can be connected through such a technology hub in association with these two broad-based domains (Chang *et al.*, 2010). In the core part, the five systems engineering models can be seen as the core technologies in which the cost benefit analysis may be used as a common platform in support of decision making. Integrated modeling systems may flexibly concatenate various optimization models including linear programming (LP), mixed integer programming (MIP), nonlinear programming (NLP), and dynamic programming (DP) models to address the system concerns in which the SM and FM can support the essential background in concert with CBA in the context of systems analysis. With such a core structure, the model-based DSSs can be constructed for separate or collective applications. Yet rule-based, knowledge-based or graphics-based DSSs or ESs can still be formed based on heuristic approaches. All of these core efforts may be enhanced by the rest of system assessment tools described by the eight outer triangles. Communication among the eight triangles canalizes the information flows that in turn improve the credibility of the five systems engineering models being formulated through MIS, DSS, and even ES. Overall, Fig 3.1 Fig. 3.1 from Chang *et al.* (2010) leads to a sound realization of the structure between systems engineering models and systems assessment tools from which a systems analysis should be well balanced for generating environmentally benign, cost effective, ecologically sound, and socially acceptable solutions (Morrissey and Browne, 2004; Chang and Davila, 2007).

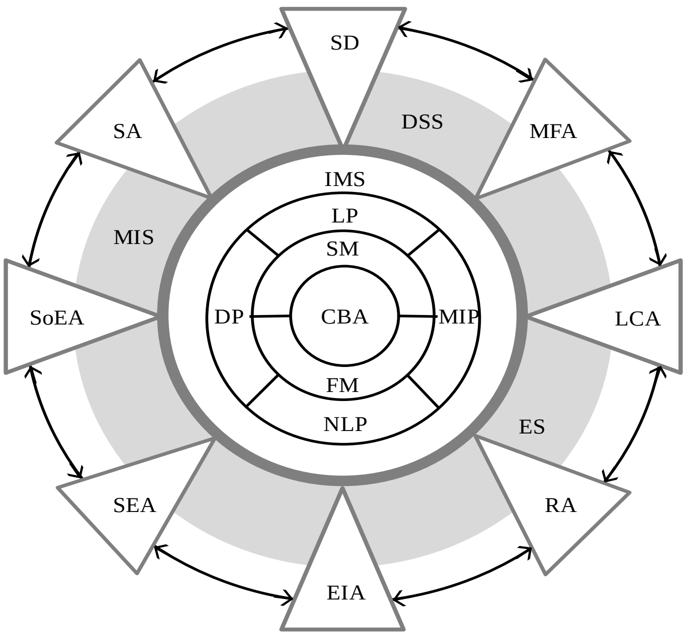
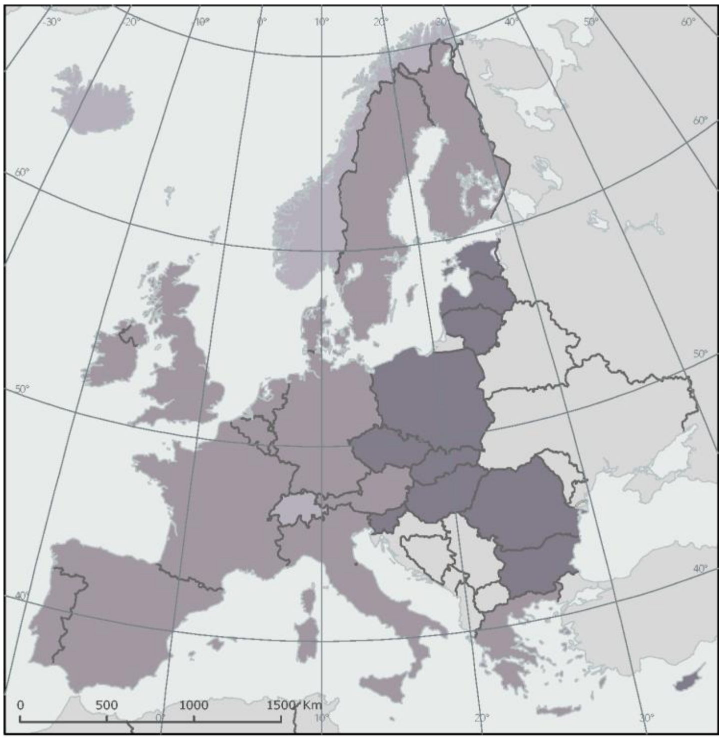


Fig. 3.1The technology hub for solid waste management

With such a tool, every community can tailor its own unique system to manage various components of the waste streams in a flexible manner (Najm *et al.*, 2002). Yet how to smooth out the barriers toward achieving appropriate systems synthesis and integration of the five systems engineering models and the nine system assessment tools to aid in solid waste management practices in European countries remains somewhat uncertain. It is the aim of this paper to present a thorough literature review and a critical analysis in sequence so as to answer the following key questions: 1) what achievements have been reached so far? ,2) what are the gaps in knowledge of waste management that we need to achieve in the context of sustainable development in the long run? and 3) what are the research needs and future directions in systems analysis for SWM in European countries. At a practical level, discussions of this paper were limited to 15 European Union (EU) member states, facing the same driving forces with similar waste legislation to manage MSW systems .The EU is an economic and political union of 27 member states which are located primarily in Europe.) Norway and Switzerland (non-EU ,embers) were also included to understand how other countries within the region with similar waste management legislation applied the techniques of systems analysis to manage their waste management issues. Fig. 3.2 illustrates the overall study boundaries (adapted from EAA, 2007b).



EU-15	New Member States	European Free Trade Association	Other European Countries
Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Nethertlands, Portugal, Spain, Sweden and UK	Bulgaria, Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Romania, Slovakia, Slovenia	Iceland, Liechtenstein, Norway and Switzerland	

Fig. 3.2 Groups of countries within the European Union

3.3 CURRENT WASTE MANAGEMENT PRINCIPLES IN THE EU

After the commitments made at the Earth Summit in Rio de Janeiro (1992), the European Council in 2001 adopted the first EU Sustainable Development Strategy (SDS). The overall aim of the renewed EU SDS is to support and promote actions enabling the EU to achieve continuous improvement of quality of life for both current and future generations. This is expected to be achieved through the creation of sustainable communities capable of managing resources efficiently, tapping the innovation potential of the economy, ensuring prosperity, environmental protection and social cohesion. These changes will bring about a sense of urgency in SWM. While short-term action is required for tackling operational issues in SWM, maintaining a long term perspective of SWM also needs to be set out. The most recent legislation published by European Commission (EC) is the New Waste Directive 2008/98/EC (EU, 2008), which reflects EU SDS and brings new challenges to SWM systems. New definitions for waste, by-products and end-of-waste, result in the need for choosing appropriate technologies that aim at improving the protection of human health and environment, promoting re-use and recycling, enhancing waste prevention programs via biowaste separate collection, and implementing extended producer responsibility (EPR) collectively. In addition, key challenges related to long term waste management are climate change and energy use, linking SWM systems with the reduction of greenhouse gas (GHG) emissions and the enhancement of energy recovery. Sustainable shipping of waste streams is thus an important issue in SWM too. Sustainable consumption and production, related to waste prevention programs have received wide attention in the nexus of resources conservation, recovery, and reuse. Social factors, including population growth and migration, become essential for the accurate forecasting of waste generation and estimation of the proper capacity of the SWM facilities. Public health, which used to be considered by LCA impact categories must be included through the application of a quality assurance system (QAS) for product control. All of them compound the structure of current SWM systems and deepen the need for systems analysis within the EU member states.

Proper consideration of the impacts of climate changes and resources scarcity has been mandatory in environmental management including SWM in Europe. Scarcity of resources has motivated new strategies at European level to promote life-cycle thinking in waste management policies, and consequently, the problems of MSW management are tied with how to integrate economically feasible and environmentally sustainable practices holistically. Challenges arise with respect to interfaces between optimal planning and sizing of solid waste management facilities and optimal scheduling of waste flows and throughputs while evaluating new system components and taking into consideration environmental and social costs, such as municipal taxes, user charges, capital opportunity costs and government grants and subsidies. These socioeconomic strategies, which were implemented only by a handful of industrialized countries in the world, might be extended to reduce waste generation and, simultaneously, de-link waste generation from economic growth.

An improved knowledge base influences the advancement of waste collection and shipping, resources use, and disposal alternatives via substitution of more systematic modeling practices. The Thematic Strategy on the Prevention and Recycling of Waste (EU, 2005) is an example of such a policy change. Improving the existing legislation, with simplification and modernization effects on waste definition, end-of-waste criteria, recycling, recovery and disposal activities, is one of the guidelines which is crucial to continue into the next decade. In addition, climate changes have also forced new measures to be implemented at the EU level. They include promoting GHG emissions reduction through biowaste diversion from landfills, improving energy efficiency at waste treatment and disposal facilities, promoting organic fertilizers (compost) in soils as an alternative to mineral fertilizers, enhancing quality in waste management outputs (like recycled materials) to reduce resource consumption, and raising materials' utility. Some social aspects in MSW management have also been made mandatory by EU regulations, like the SEA Directive, related to public participation with respect to the drawing up of certain plans and programs relevant to the environmental directive (EU, 2001).

To ultimately improve urban sustainability and offer the level of service required by the population, the ability to increase the reliability of green infrastructure systems with waste management functionalities, particularly through the proper interfaces between the partnerships of private and public sectors could be even more critical. Ultimately, not only mitigatory solutions related to climate change should comprise a part of the MSW management strategies but also challenges in adaptation are also made necessary for SWM, which are mainly related to waste treatment technologies. On the other hand, waste collection systems should be designed and operated so as to be capable of improving public health protection. For example, higher temperatures after climate change that may result in more biowaste degradation, thereby generating odor control problems which require further attention. More sophisticated societal measures, such as voluntary agreements on encouraging responsibility among producers and consumers, which might become mandatory to reach an integrated solid waste management (ISWM), compound the decision making when arriving at consensus and involvement between stakeholders and decision makers during participation processes in different EU member states. This situation triggers an acute need to thoroughly review all the existing "state-of-the-art" systems engineering approaches and system assessment tools in order to reach such ISWM goals, particularly in EU member states with differing levels of economic development. These goals are necessary to facilitate a unique integration for tackling complexity with respect to multi-objectives, risk, and uncertainty characteristics in decision making as a whole.

3.4 SYSTEMS ANALYSIS TECHNIQUES

In systems engineering regimes, a system can be a set of related components or sub-systems, which interact with each other in some way. The properties of a system are defined by the whole of the subsystems, their characteristics, and their relationships. The characteristics are related to the boundaries of the system depending on whether they are closed or open systems/sub-systems. With this definition, a MSW management system fits the concept in which the technical aspects like landfill, incineration, anaerobic digestion, composting and collection are sub-systems linked with one another through processed waste streams internally and municipalities through managed truck fleets externally. The sub-systems make up part of the SWM system that has interactions between technical and non-technical aspects, both of which may influence the generation and shipping of waste to some extent.

Considering SWM systems, several systems analysis techniques have been applied to help decision making. These can be divided into two main groups as we mentioned above: systems engineering models and systems assessment tools. Their contribution to SWM systems will be summarized in the following sub-sections in concert with the concept.

3.4.1 Systems engineering models

Complexity in SWM system arises from siting facilities, selecting technologies, and comparing management options. To tackle the synergistic interfaces, systems engineering models can be helpful for promoting analysis based on cost-benefit analysis (CBA), optimization models (OM), simulation models (SM), forecasting models (SM) and integrated modeling systems (IMS). Table 3.1. presents the contribution of systems engineering models to SWM system analysis over the past few decades. It offers a systematic overview showing how the landscape of systems engineering models was conceptualized (complexes, structures, functionalities) and the relationship to other components in connection with Fig. 3.1 (adapted from Chang *et al.* (2010).

Table 3.1 The contribution of systems engineering models to SWM

Types of systems engineering models	Description	Contribution to SWM system
Cost-benefit analysis	To assess positive and negative economic and physical effects independently or support simulation and optimization models for systems analysis	Well-defined cost-benefit models may translate environmental aspects into economic terms. However, the intergeneration externalities are very difficult to address.
Optimization model	To reach the best solution among numerous alternatives, considering one or several objectives.	Models have solved the following issues: • single network planning (Anderson and Nigam, 1967; Anderson, 1968; Fuertes *et al.*, 1974; Helms and Clark, 1974; Kuhner and Harrigton, 1975; Jenkins, 1979; Clayton, 1976, Rao, 1975) • dynamic, multi-period investment (Marks, *et al.*, 1970; Marks and Liebman, 1971) • size and site facilities (Chapman and Yakowitz, 1984; Li and Huang, 2006a; Nie *et al.*, 2007; Li and Huang 2009a; Li *et al.*, 2007; Li *et al.*, 2006a, 2008a,b; Huang *et al.*,2001, 2002; Xu *et al.*, 2009; Li and Huang, 2006b, 2009b) • manage infrastructures like landfill (Davila *et al.*, 2005) Models developed: WRAP (USEPA, 1977)
Simulation model	To trace the lengthy chains of continuous or discrete events based on cause-and-effect relations describing the operations in complex systems and helping investigate the dynamic behavior of the system (Wang *et al.*, 1996).	Models developed for SWM systems: SWIM (Wang *et al.*, 1996), GIGO (Lawver *et al.*, 1990; Anex *et al.*, 1996), AWAST (Villeneuve *et al.*, 2008), EcoSolver IP-SSK (Krivtsiv *et al.*, 2004), TASAR (Tanskanen and Melanen, 1999)
Forecasting model	To characterize waste streams quantitatively and qualitatively and construct a management information system to accumulate information over time. To predict waste generation, time-series regression analysis (Katsamaki *et al.*, 1998 and Navarro-Ésbri *et al.*, 2002), system dynamics models (Dyson and Chang, 2005), and other regression models have been applied (Grossman *et al.* 1974).	Models have related variables like population (Grossman *et al.* 1974), income level (Grossman *et al.* 1974; Beigl *et al.*, 2005), dwelling unit size (Grossman *et al.* 1974), total consumer expenditure and gross domestic product (Daskalopoulos *et al.*, 1998), production measures, household size, age structure, health indicators (Beigl *et al.*, 2005), per capita retail and tipping fees for waste disposal (Hockett *et al.*, 1995) to waste generation, total income per service center, people per household, historical amount generated, income per house and population (Dyson and Chang, 2005).
Integrated modeling systems	To improve synergistic connections among different models, concatenating their total functionalities.	IMS have provided: • dynamic information of waste generation and waste shipping (Chang *et al.*, 1993b) • optimal capacity expansion patterns for waste-to-energy and landfill facilities over time (Baetz, 1990) Models developed: ORWARE (Dalemo *et al.*, 1997; Björklund *et al.*, 1999)

3.4.2 Systems assessment tools

Most of the time, after systems have been created and implemented, it is necessary to evaluate their performance and consider how improvements could be made, especially in answer to the increasing challenges promoted by regulation. Models that can help decision makers towards such goals are systems assessment tools. Such tools can be MIS, DSS, ES, SD, MFA, LCA, RA, EIA, SEA, SoEA and SA. Table 3.2 presents the contribution of systems assessment tools to SWM with significant ramifications on how the relationships between system assessment tools and systems engineering models are described when connected with Fig. 3.1 (Chang *et al.*, 2010). As a consequence, the appropriate use of system assessment tools has such a large effect on the overall optimization especially in the context of IMS because the outputs from these tools are normally used as the primary inputs in models reflecting socioeconomic, climate change, and managerial considerations.

Table 3.2 The contribution of systems assessment tools to SWM

Systems assessment tools	Description	Contribution to SWM systems
Management information system, decision support system and expert systems	Consists of different methods applied to exchange and manage information; used to help in decision making	MIS/DSS/ES have been applied: • to provide information storage and transmission through countries (EIONET, 2009) • to yield specific decision support (Chang and Wang, 1996d; Barlishen and Baetz, 1996; Haastrup *et al.*, 1998; Bhargava and Tettelbach, 1997; AEA Technology, 1998) • to relate waste stream characterization with implications on shipping, processing and disposal of waste streams (MacDonald, 1996b)
Scenario development	To create hypothetical sequences of events constructed for the purpose of focusing attention on causal processes and decision points (Kahn and Wiener, 1967)	Has the ability to explore events (events in this case are policies and decisions taken) that might occur associated with SWM on a temporal scale. Such events can be inside or outside the SWM system. Fell and Fletcher (2007) have contributed with scenario developments for future lifestyle trends and forecasting based on lifestyle scenarios for waste composition
Material flow analysis	Consists of a systematic assessment of the flows and stocks of materials within a system defined in space and time (Brunner and Rechberger, 2003)	Software developed in MFA: SFINX (van der Voet, 1995a,b), FLUX (Huijbregts, 2000), STAN (TU Vienna, 2009), DYNFLOW (Elshkaki, 2000), GaBi (PE International, 2006) and Umberto (IFU, 2006)
Life cycle assessment	Consists of a process to evaluate environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used, wastes and emissions released to the environment, to assess impact of those energy and	Models developed for SWM systems: IWM (White *et al.*, 1995; McDougall *et al.*, 2001), WASTED (Diaz and Warith, 2006), WISARD/WRATE (Ecobilan, 2004; Buttol *et al.*, 2007), EASEWASTE (Christensen *et al.*, 2007)

Systems assessment tools	Description	Contribution to SWM systems
	material uses and releases and to identify and evaluate opportunities that lead to environmental improvements (EEA, 2003)	
Risk assessment	To relate environmental and human health risk to accidents quantitatively, through a statistical evaluation	Help in the evaluation of transversal SWM systems
Environmental impact assessment	A procedure that aims to ensure that the decision-making process concerning activities that may have a significant influence on the environment takes into account the environmental aspects related to the decision (Tukker, 2009)	EIA associated to a specific project attempts to solve controversial issues from the target project such as siting issues originated from the NIMBY effect, technical issues to justifying the choice of technology for emission reduction, and even the rejection of the project (Chang *et al.*, 2009b). In Europe, EIA is mandatory for landfills and incineration plants with regard to capacity limits through EU Directive 85/337/EEC (EU, 1985), as amended by EU Directive 97/11/EC (EU, 1997). A good example can be found in Barker and Wood (1999)
Strategic environmental assessment	Consists of the environmental assessment of a strategic action as a policy, a plan or a program (Thérivel and Partidário, 2009)	Its applicability is emphasized by EU Directive 2001/42/EC (EU, 2001), to which it is obligated for the promotion and elaboration of an SEA for SWM plans. More details can be found out in Dutch Ten Year Program on Waste management 1992 and 2002 (Verheem, 1999)
Socio-economic assessment	Consists of computer-based practices that apply integrated market-based and/or policy/regulation requirements for SWM	Has allowed the inclusion of user-charges, landfill disposal fees, recycling credits, product charges, deposit-refund schemes, and producer responsibility schemes into the decision making in SWM systems, promoting a more sustainable management of waste. For such purposes, several methodologies have been applied: CBA-based LP (Chang *et al.*, 1997a; Chang *et al.*, 1996), CBA-based MIP (Chang *et al.*, 2005), CBA-based fuzzy goal programming (Chang and Wang, 1997a), fuzzy contingent valuation (Chang *et al.*, 2009b), minimax regret optimization (Chang and Davila, 2007), GIP-based game theory (Davila *et al.*, 2005), CBA-based MCDM (Karagiannidis and Moussiopoulos, 1997; Rousis *et al.*, 2008), optimal control of landfill space (Chang and Schuler, 1991) inexact fuzzy-stochastic constraint (Li *et al.*, 2009), IOA (Brahms and Schwitters, 1985; Franklin Associates, 1999; Gay *et al.*, 1993; Hekkert *et al.*, 2000; Joosten *et al.*, 2000; Patel *et al.*, 1998; Nakamura, 1999; Pimenteira *et al.*, 2005)
Sustainable assessment	Refers to the integration of different methodologies in such a way that obtaining an analysis, an evaluation or a planning that approaches several management aspects in which sustainability implications may be emphasized and illuminated	SWM systems assessed to reach sustainable management, focusing on different aspects. Models developed: LCA-IWM (den Boer *et al.*, 2007) and MSW-DST (Thorneloe *et al.*, 2007, Weitz *et al.*, 1999). Several methods have been combined to reach sustainability: Cherubini *et al.* (2008) have combined LCA with MFA and energy analysis methods, Nakamura and Kondo (2002) used IOA and LCA to construct a waste input-output model, Huppes *et al.* (2006) and Tukker *et al.* (2009) have combined both methods to obtain IOA with environmental extensions

Systems assessment tools	Description	Contribution to SWM systems
		for different sections (including waste management sectors). A Geographical Information System (GIS) combined with LCI, EIA and optimization model have been promoted by Chang *et al.* (2008, 2009a) for landfill siting

3.5 SYSTEMS ANALYSIS USED FOR SOLID WASTE MANAGEMENT IN EUROPEAN COUNTRIES

3.5.1 Methodology

Several types of SWM systems in European countries can be identified and classified. The characterization of these systems in the EU and its member states was mainly performed by the authors in 2008 and 2009 based on the databases developed by European Topic Centre on Sustainable Consumption and Production. In the case of Belgium and Spain, for example, the inquiry was made through the regional entities of Flanders and Catalonia, respectively. To understand which are the research needs and future directions in regard to the systems analysis techniques for SWM in European countries, a comparative analysis was also conducted in this paper for the distinction of relevant applications.

3.5.2 Waste management systems in European countries

From a life cycle point of view, an all-inclusive MSW management system includes all essential operational units from collection, to shipping, to treatment, to recycling, and to disposal. Yet the current European regulations promoting the hierarchy of waste management inevitably involve a wealth of waste management practices tied to policies, institutional settings, financial mechanisms, technology selection, and stakeholder participation. For instance, the landfill directive promoted biodegradable municipal waste (BMW) management systems, which focus on building separate collection systems provided by local authorities through specific bins leading to separate, mandatory BMW treatment systems (Austria, Netherlands). Some of the EU member states applied economic instruments including Pay-As-You-Throw (PAYT) and an organic waste tax to create economic incentives for residents to divert BMW from regular waste streams normally being collected by municipalities to specific collection avenues. They recognized that the diversion costs that each waste disposal authority would face would differ according to the particular circumstances (EIONET, 2007a). For example, both BMW system and Landfill Allowance Trading System (LATS) in the United Kingdom (UK) were launched to provide local authorities with the flexibility to manage waste streams more effectively. The LATS system revolves around transferable allowances which enable the greatest amount of waste diversion to occur in areas where it is cheapest, and most practicable to do so.

The Packaging waste directive has also promoted similar incentives by using the “EPR system” and the “deposit refund system” to ensure the maximum reuse and recycling. The most well known EPR system is the packaging waste Duales System Deutschland (DSD) (or Green Dot system) that was firstly applied in Germany in the 1990s and later on all over Europe (Buclet, 2002). The basic idea of the DSD is to establish a privately organized channel assuring that all primary packaging can be collected from the consumers will then undergoes a material-specific recycling process through the consumers and service providers. This is done through the so called “Green Dot” which is a label on packaging material used to identify the product belonging to the dual system during the consumption phase (Klepper and Michaelis, 1994). As for these deposit refund systems, Dansk Retursystem is one of the oldest ones, and has been in use since 1984; it is applicable for refillable, non-refillable, reusable and disposal, and ready-to-drink beverages and mineral water bottles (Pro-Europe, 2009). However, in Denmark there is no producer-responsibility scheme, namely no separate management system for packaging waste (Danish EPA, 1999; Pro-Europe, 2009), making the costs for handling packaging waste uncertain, and resulting in a higher budget for waste management in local authorities (EEA, 2005).

The remaining part of waste streams usually called residual household waste or mixed municipal waste still need to be cleaned up by municipalities and local authorities. The Danish Waste Model is a representative residual/mixed waste system as it is based on a joint venture to form a coherent whole (Danish EPA, 2001). According to the Danish EPA (2001), the structure of the SWM systems is characterized by the following principles: 1) the system includes all types of waste (e.g. household, industrial and hazardous waste); 2) the responsibility for the SWM lies solely with the local authorities (council), which are responsible for establishing capacity for waste management and for providing information on how to dispose of the waste produced within the local council, irrespective of whether this waste originates at households or trade and industry (EIONET, 2007b); 3) the duty to assign waste treatment and disposal facilities lies with the local authorities, and waste generators are bound to those who use them; 4) financing of the system rests on the polluter-pays principle (PPP); and 5) waste collection and waste treatment rest on the principle of source separation (EIONET, 2007b). With these principles, the systems boundaries are quite well defined, such as packaging waste collection created by local authorities was managed by an external system. Considering the main waste streams in MSW – residual waste, BMW and packaging waste – a review on SWM systems in European countries may be summarized in Table 3.3.

Table 3.3 Waste management systems in European countries

Country	Residual/mixed waste system	BMW system	Packaging system
Austria	Yes, with ban of landfill regulation for BMW	Yes, being mandatory with penalties	• Alstoff Recycling Austria system (Green Dot Dystem) – EPR • Bonus Holsystem (commercial packaging waste) – EPR • Öko-Box for beverage carton containers • Pet2Pet for polyethylene terephthalate (PET) bottle • Deposit refund system for beverage containers but only mandatory for refillable plastic beverage containers • ARO system for wastepaper • AGR system for glass • ArgeV for lightweight fraction
Belgium	Yes, with cash tax, residual waste and/or environmental tax and organizations like Fost Plus	Yes, with PAYT	• Fost Plus (Green Dot System) – EPR
Denmark	Yes, with collection fee based on polluter-pays principle	Yes, but is voluntary and only for garden waste	• Dansk Retursystem – deposit refund system • Packaging glass system: recycling schemes • Wastepaper and waste cardboard system: recycling schemes are mandatory • Packaging tax
Finland	Yes, with waste charge: sorted materials pays less	Yes, promoted by information instruments	• Suomen Palautuspakkaus Oy (Palpa) – deposit-refund system for packaging waste, including beverages • Newspaper, copy paper and other paper products and packaging waste – EPR • Beverage containers – packaging tax (exemption or lower tax rates only if package is part of a returnable deposit scheme)
France	Yes, with fees	Exists but not consolidated	• Eco-Emballages (Green Dot System) – EPR
Germany	Yes	Bio-Bin system and other mandatory systems	• Duales System Deutschland – EPR • Deposit-refund system
Greece	Yes, with fees to cover the service	No	• Green Dot System called HERRCo – EPR • KEPED – packaging waste system for waste oils • Supermarkets as individual systems
Ireland	Yes, with PAYT	Exists but not consolidated	• Repak – Green Dot system - EPR
Italy	Yes, with municipal waste tariff	Yes	• Consorzio Nazionale Imballaggi (CONAI) (Green Dot System) – EPR

Country	Residual/mixed waste system	BMW system	Packaging system
Luxembourg	Yes. With PAYT system	Yes (green bins)	• Valorlux (Green Dot System) – EPR
Netherlands	Yes, with levy	Yes, mandatory	• Compulsory deposit return systems for refillable glass bottles and one-way packaging • Nedvang – EPR • Stichting Retourverpakking Nederland – one-way deposit-bearing PET containers for soft drinks and water larger than 0.5litre • Wastepaper and waste cardboard – EPR
Norway	Yes, with waste tariffs	Yes, with organic waste tax	• Norsk Resy AS – packaging waste and corrugated and solid board packaging – EPR • Norsk GlassGjenvinning AS for glass • Norsk MetallGjenvinning AS for metal • Gront Punkt Norge for plastic packaging, beverage cartons and carton packaging • Norsk Resirk AS – deposit-refund system for beverage packaging, steel and aluminium cans, plastic bottles non-refillable
Portugal	Yes, by water consumption fee	No	• Sociedade Ponto Verde (Green Dot System) – EPR • Valormed (Medicine packaging waste) – EPR • Marão mineral water system (private) – deposit-refund system for one way PET bottles of Marão trend mark
Spain	Yes for Catalonia, with a landfill tax, incineration tax	Only in Catalonia for municipalities with > 5,000 inhabitants	• Ecoembes SL (Green Dot System) – EPR • Ecovidrio, for glass packaging – EPR
Sweden	Yes	Yes	• EPR for several waste streams like packaging and waste paper • Deposit refund systems for cans, plastics and glass bottles • Returpack – deposit refund-system for all plastic and metal beverage containers for ready-to-drink beverages, including refillable glass bottles
Switzerland	Yes, with Canton tax	Yes, whenever possible	• Beverage bottles – EPR • Reusable packaging – deposit refund system • PET-Recycling Schweiz – packaging PET one way – EPR • IGORA for aluminium cans – EPR • Ferro-Recycling for tinplate – EPR • VetroSwiss for glass – EPR, mandatory system • Disposable packaging in PVC – obligatory deposit • Wastepaper and cardboard system related to municipalities/local authorities
United Kingdom	Yes, landfill tax only for companies, local authorities or other	Yes, LATS and for garden waste (from	• PRN system – EPR • PERN system – EPR

Country	Residual/mixed waste system	BMW system	Packaging system
	organization	civic amenity)	

In addition to existing SWM systems handling related material and cash flows at scales, there is also a need for building up proper information exchange platforms capable of offering decision makers the basis for the assessment of relevant projects, programs, and plans. Member states must provide information to supply European waste information systems like Eurostat, EIONET, and ReportNet. Eurostat's main role is to process and publish comparable statistical information at European level (Eurostat, 2009). The Eurostat data centre on waste is responsible for providing robust data, indicators and other relevant information for assessing the effectiveness of the community waste policy (Eurostat, 2009). The functions of the Eurostat information system are divided into four sectors which correspond to the various stages in the processing of data from their collection to their dissemination including production (collection, validation and storage of the data and meta-data), storage of the reference data (acceptance of the information), use of the reference data (visibility/security and find/deliver), and dissemination of information (Dubois, 1997).

EIONET is a collaborative network of the European Environment Agency (EEA) and its member states, connecting National Focal Points in EU and accession countries, European Topic Centres, National Reference Centres, and Main Component Elements. EIONET provides a mechanism whereby National Focal Points in European countries can make documents available to the EEA and also retrieve documents of interest from the EEA. The integrated electronic workplace environment allows online collaboration between environmental personnel across Europe. EIONET supports the collaborative process and reduces the reporting burden for environmental protection agencies across Europe (EEA, 2002).

ReportNet aims to develop common tools and a shared information infrastructure as the European Environmental Information System; and it is based on a set of inter-related tools and processes which all build on the active use of the World Wide Web (EIONET, 2009). ReportNet is EIONET's infrastructure for supporting and improving data and information flows (EIONET, 2009). With this platform, ReportN*et al*so aims at providing an effective network infrastructure whereby collaboration can be achieved so that the environmental reporting burden of the EU member states can also be reduced (EEA, 2002).

3.5.3 Comparative analysis

To understand how to deal with multiple alternatives and a plethora of outcomes regarding systems analysis for SWM in European countries, a comparative analysis were conducted based on 218 applications in this paper. Such a comparative analysis is presented in Fig. 3.3. Sometimes the assessments conducted for waste management overlap with several SWM systems simultaneously; such cases happened when a specific decision maker responsible for waste management at a geographical area of interest is the same as the decision maker inside such boundaries handling several subsystems.

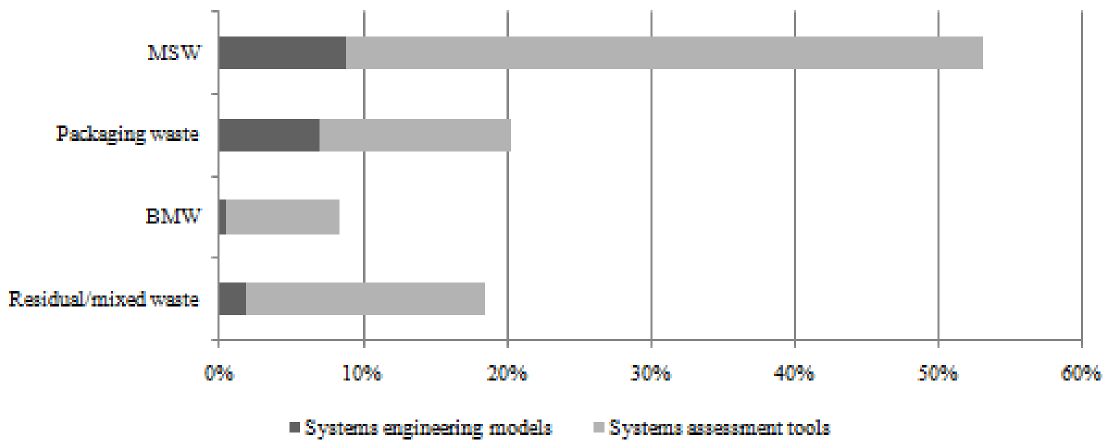


Fig. 3.3 Systems analysis applied for solid waste management systems in Europe

Fig. 3.3 clearly indicates that 1) more cases were associated with MSW, 2) the studies on residual/mixed waste and packaging waste streams were received equal emphasis, and 3) BMW received the least attention. Such phenomena can be explained by the fact that the BMW system is less established in European countries, as opposed to other parts of the world. Comparing the relative distribution between groups of models and tools for systems analysis, the most common practices for waste management in European countries are those using various systems assessment tools rather than system engineering models.

Table 3.4 further confirms the same observations after the realization of application metrics of systems assessment tools.

The comparison across the boundaries implies that systems assessment tools have been applied to evaluate and help in decision making based on environmental issues have great potential to integrate other aspects, like economics or social impacts.

Table 3.4 Application metrics of systems assessment tools

SWM systems	Systems assessment tools	References
MSW	To collect and share information flows in SWM	Nationale Reststoffenbeurs, 1986; Waste Exchange UK, 2000; CIWM, 2003; Dall *et al.*, 2003; Becker *et al.*, 2007; Denmark Waste Exchange, 2008, LUA NRW, 2006; International Synergies Limited, 2007; Fahy, 2007; Economie, 2008; Jean-Gerard, 2008; APA, 2008; IHK Recyclingborse, 2008; IWEN, 2008; EIONET, 2009; Mochty, 2009
	To understand environmental impacts in SWM system with respect to pollutant fate and transport, or the waste itself	Dahlbo and Assmuth, 1997; Obernosterer and Brunner, 1997; Powell *et al.*, 1996,1999; Döberl *et al.*, 2002; Melloni *et al.*, 2003; Bolze, 2004; Beigl and Salhofer, 2004; Sokka *et al.*, 2004; Ecobilan, 2004; Xará *et al.*, 2005; Kirkeby *et al.*, 2005; Badino *et al.*, 2007; Mastellone *et al.*, 2009; Rigamonti *et al.*, 2009a; 2009b; Frakgou *et al.*, 2009
	To assess SWM plans, regulations, policies, and strategies	EU, 1997; Björklund *et al.*, 1999; Saarikoski, 2000; Arbter, 2001; Moberg *et al.*, 2002; Aumônier, 2002; Ministry of the Environment Government of Japan, 2003; Salhofer *et al.*, 2005; Pladerer *et al.*, 2007; Salhofer *et al.*, 2007; SEA Wiki, 2007; Escalante *et al.*, 2007; Buttol *et al.*, 2007; Cheshire County Council, 2007; SEPA, 2007b; Pisoni *et al.*, 2009; NLWA, 2009; Desmond, 2009
	To assess options for decision making in SWM systems	Sundberg, 1993; Karagiannidis and Moussiopoulos, 1997; Sivertun and Le Duc, 1998; Ljunggren, 1998, 2000; Wilson, 2002; Reich, 2002; Fiorucci *et al.*, 2003; Karagiannidis *et al.*, 2003; Skordilis, 2004; Muñoz *et al.*, 2004; Costi *et al.*, 2004; Viotti *et al.*, 2005; Eriksson *et al.*, 2005; Reich, 2005; Gentil *et al.*, 2005; Dornburg and Faaij, 2006; Jansen and Gerlo, 2006; Minciardi *et al.*, 2007; SEPA, 2007a; Ulli-Beer *et al.*, 2007; Bovea and Powell, 2006; Rodríguez-Iglesias *et al.*, 2007; Cherubini *et al.*, 2008; Gallo *et al.*, 2009; de Feo and Malvano, 2009; Federico *et al.*, 2009; Ekvall *et al.*, 2009; Tunesi and Rydin, 2009; Abeliotis *et al.* 2009
	To site infrastructures	Lahdelma *et al.*, 2002; EEA, 2003
	To assess part of the system including waste production steps with respect to perspectives	Finnveden *et al.*, 2002; Fell and Fletcher, 2007; Bovea *et al.*, 2007; Grosso *et al.*, 2008; Karadimas and Loumos, 2008
Residual/ mixed waste	To collect and share information flows in SWM	LUA NRW, 2006
	To assess environmental impacts related to SWM infrastructures	Harrop and Pollard, 1998; Coutinho *et al.*, 1998; Snary, 2002; Allgaier and Stegmann, 2003; Verro *et al.*, 2003; Capuzzo and Farina, 2003; Cossu *et al.*, 2003; Boerboom *et al.*, 2003; Marques and Hogland, 2003; Belgiorno *et al.*, 2003; Belfiore *et al.*, 2005; Zorzi *et al.*, 2005; Morra *et al.*, 2005, 2006; Masi *et al.*, 2007; Bour and Zdanevitch, 2007; Cangialosi *et al.*, 2008; Moutavtchi *et al.*, 2008; Perkoulidis *et al.*, 2010
	To assess SWM options and the system itself	Loeschau and Rotter, 2005; van der Linden and Torfs, 2005; Chanchampee and Rotter, 2007
	To evaluate operations occurring in the SWM system (collection, treatment, and disposal)	Ecobilan, 2004; Bergsdal *et al.*, 2005; Emery *et al.*, 2007; Wittmaier *et al.*, 2009

SWM systems	Systems assessment tools	References
	To site infrastructures	Vaccari *et al.*, 2005
	To assess policies and economic instruments	Nilsson *et al.*, 2005; Björklund and Finnveden, 2007
Packaging waste	To collect and share information	GS1, 2008
	To assess management options for a specific packaging material, considering environmental perspectives	Finnveden *et al.*, 1994; Kaila, 1998; Ryberg *et al.*, 1998; Person *et al.*, 1998a,b; Widheden *et al.*, 1998a,b; Frees *et al.*, 1998; Detzel *et al.*, 2003; Frees *et al.*, 2004; Pancaldi *et al.*, 2005; Schmidt *et al.*, 2007; Dahlbo *et al.*, 2007
	To assess management options for a specific packaging material considering targets to be established	Dalager *et al.*, 1995; Fehringer and Brunner, 1997
	To analyze the specific parts of the system, including collection, treatment, and disposal	Baumann *et al.*, 1993; Finnveden and Ekvall, 1998; Holmquist, 1999; Rutegård, 1999; Ibenholt and Lindhjem, 2003; Ecobilan, 2004
	To assess and compare different SWM systems applied to a specific packaging waste	Frees and Weidema, 1998; Ekvall *et al.*, 1998; Jahre, 1998; Ekvall and Bäckman, 2002; Hischier *et al.*, 2005; Heilmann and Winkler, 2005; Dahlbo *et al.*, 2005; Vercalsteren *et al.*, 2007
	To assess policies	Bruvoll, 1998; Wäger *et al.*, 2001
BMW	To assess and improve the system, including environmental perspectives	Björklund *et al.*, 2000; Wassermann *et al.*, 2003; Shmelev and Powell, 2006; Güereca *et al.*, 2006; Schmidt and Pahl-Wostl, 2007; EUNOMIA, 2007
	To understand environmental impacts in SWM system with respect to pollutant fate and transport, or the waste itself	Boldrin and Christensen, 2007
	To understand the source of the waste streams	Purcell and Magette, 2007; 2009
	To compare system outputs with substitute products	Eriksson *et al.*, 2002
	To compare technologies applied to the collection, treatment and disposal in SWM systems	Edelmann and Schleiss, 1999; Danish EPA, 2003; Lang *et al.*, 2006a,b

Given that EU regulations have given emphasis to EIA in SWM, the inclusion of EIA has become favored by decision makers at national, regional, and local levels. Thus, the primary stage of decision analysis normally leads to assess a suite of management options, evaluate managerial and strategic plans, and collect and share information. Likewise, climate change and resources depletion are emerging issues of most concern, which are even more influential when managing SWM, and decisions and policies were oftentimes made with the aid of LCA or LCI in public institutions.

With fewer applications, systems engineering models were capable of studying waste production processes and assessing the interactions in numerous types of SWM systems addressing impacts from technical to social, and to economic perspectives. However, such applications are not easy to implement since the necessary assumptions being made may or may not be realistic. As a consequence, systems engineering models have not been applied to the same extent as systems assessment tools in EU member states. Oftentimes, these models are not geared toward helping decision makers' needs. Their contribution is often limited to use a mathematical functional form structured to derive strategic guidelines and/or orientations in a SWM system. Sometimes, the mathematical outputs are contradictory with existing ideas that have already embedded in decision makers' minds. In such cases, CBA may be defined or refined well to fit in the LCA framework in the decision making arena, the application potential may be improved. Table 3.5 further confirms the observations across MSW, residual/ mixed waste, packaging waste, and BMW after the realization of application metrics of systems assessment tools.

Table 3.5 Application metrics of systems engineering models

SWM systems	Systems engineering models	References
MSW	To predict solid waste production	Brahms and Schwitters, 1985; 2009Dennison *et al.*, 1996a; 1996b; Andersen *et al.*, 1998, Patel *et al.*, 1998; EEA, 1999; Navarro-Esbrí *et al.*, 2002; Lebersorger *et al.*, 2003; Beigl and Lebersorger,
	To optimize the system for choosing the best option	Kaila, 1987; Hokkanen and Salminen, 1997; Gottinger, 1988; Cosmi *et al.*, 1998; Komilis, 2007
	To assess recycling rate	Huhtala, 1997
	To site infrastructures	Mitropoulos *et al.*, 2009
	To analyze specific parts of the system	Tanskanen and Melanen 1999; Villeneuve *et al.*, 2005, 2008
	To assess the system	MCCK and Consultancy, 1998
Residual/ mixed waste	To site infrastructures	Arnold and Terra, 2006
Packaging waste	To analyze how to reach recycling targets	Radetzki, 1999; Angst *et al.*, 2001
	To study/predict waste production	Bach *et al.*, 2004; Maunder *et al.*, 2006
	To analyze the specific parts of the system, like collection, treatment disposal	Hanley and Slark, 1994; Powell *et al.*, 1995; Tucker *et al.*, 1998; Wäger *et al.*, 1998; Ekvall and Bäckman, 2001; Petersen and Andersen, 2002
	To understand and know social cost and benefits of different	Vigsø, 2004

	packaging waste systems	
	To assess policies	McHenry *et al.*, 2003
BMW	To assess the system	Le Bozec *et al.*, 2009

Table 3.6 presents individual efforts in each country level; apparently all European countries involved can be classified into three groups according to their application of systems analysis methodology. Countries that have mostly applied systems analysis techniques include Italy, Sweden, the United Kingdom, and Denmark; countries with a moderate number of applications include France, Germany, Austria and Finland; and countries with low interest in such applications include Spain, Greece, Ireland, Luxembourg, Portugal, Belgium, the Netherlands, Norway and Switzerland. Why do such discrepancies occur, given that all EU member states are under the same European guidelines? It is mainly due to national differences in waste management policies within each country. In countries where more systems analysis practices have been applied, the driving forces for the applications are related to the need for solving multi-faceted environmental issues linked with various facilities while complying with international, EU, regional, and local regulations.

Table 3.6 Number of articles applied to study SWM systems in European countries

Countries		AT	BE	DE	DK	ES	FI	FR	GR	IE	IT	LU	NL	PT	SE	UK	NO	CH	Total
Systems engineering models	CBA	1	0	0	2	0	0	1	0	1	0	0	0	0	2	3	2	0	12
	FM	4	0	0	1	1	0	0	1	2	0	0	0	0	0	1	0	0	10
	SM	0	0	0	0	0	1	3	0	0	0	0	0	0	0	1	1	1	7
	OM	0	0	1	0	0	2	0	3	0	1	0	0	0	0	0	0	0	7
	IMS	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	2
Systems assessment tools	MIS/DSS/ES	2	1	**7**	2	0	0	1	1	1	**4**	1	1	1	1	**3**	0	0	**26**
	SD	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	2
	MFA	2	1	2	0	1	2	0	0	0	1	0	0	0	0	0	0	1	10
	LCA/LCI	1	2	7	**13**	6	2	**13**	0	0	**10**	0	2	1	7	5	1	2	**72**
	RA	1	0	0	0	0	0	1	0	0	7	0	1	0	1	2	0	0	13
	EIA	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	2
	SEA	4	0	0	0	0	1	0	0	1	1	0	2	0	0	3	0	0	12
	SoEA	0	0	2	0	0	0	0	0	0	3	0	0	0	4	1	0	2	12
	SA	1	0	0	1	0	5	0	3	2	5	0	1	0	**10**	2	0	1	**31**
Total		**16**	4	**19**	**19**	8	**15**	**19**	8	7	**32**	1	8	3	**25**	**23**	4	7	**218**

Note: Country abbreviations - AT (Austria), BE (Belgium), DE (Germany), DK(Denmark), FI(Finland), FR(France), GR(Greece), IE(Ireland), IT(Italy), Luxembourg(LU), NL(Netherlands), NO(Norway), PT(Portugal), CH(Switzerland), SE(Sweden), ES(Spain), UK(United Kingdom)

Sweden, a country where sustainability principles were heavily promoted by national/international entities and its waste management policies were born based on EPR, can be selected as a particular case. Such sustainability principles have infiltrated into the municipality level, such as Stockholm, which is the city where the development of models to aid in decision making of SWM was favored. In Denmark, the packaging refundable system was assessed due to the presence of the Packaging Waste Directive. In Italy, due to a legal action against it within EC in a waste crisis that had plagued Naples and the Campania region for more than ten years, Mastellone *et al.* (2009) conducted a MFA to provide scientific support for decision makers who were managing the waste crisis. These three cases, including Sweden, Italy, and Denmark, concur with our observations.

Concerning countries which have applied systems analysis techniques only moderately, motivations for doing so originated from the political concern with regard to environmental impacts due to SWM. For exemple, in France, Ecobilan-Pricewaterhouse Coopers developed several assessments for Agence de l'Environnement et de la Maîtrise de l'Energie (ADEME) concerning SWM. Additional motivations were tied to the need to comply with European regulations and to manage the waste streams in a sustainable way. As for those European countries with no prevalent applications of systems analysis techniques, the true reason is that they were lacking sustainable development concepts and communication channels among stakeholders in the waste management regime.

Even though some European countries did not promote systems analysis practices for SWM solely, it did not imply that the issues of SWM were ignored. According to the OECD (2009), LCI/LCA was deemed the most popular system assessment tool in the EU so far. In the nexus of environmental management and industrial ecology, LCA is actually a normalized method in connection with the norm family ISO 14040 (ISO 14040, 2006). In reality, LCA may be integrated with other system assessment tools while standing up to close scrutiny and achieving a higher level of evaluation for SWM system wide. In addition, further advantages via applying LCA/LCI can be assured by the elaboration of environmental indicators in the scope of ISO 14025 (ISO 14025, 2006) (Gallo *et al.*, 2009). Both ISO 14040 and ISO 14025 allow the gap to be bridged between waste management and industrial ecology. For example, at this juncture, the Netherlands applied LCA, MFA, CBA and EIA collectively to materials management, which includes product use phase and waste production phase.

SA models, mainly developed in Sweden, were the second mostly applied system assessment tool in the EU for SWM because of the possible linkage between SWM and energy recovery (e.g., waste-to-energy) given the presence of a large number of incineration plants. Besides, the ORganic WAste REsearch (ORWARE) model, mainly developed in Sweden, combines the concept of LCA and MFA to simulate and assess MSW and BMW systems. This type of system analysis that is considered highly novel in Sweden has been widely applied. On the other hand, the application of MIS/DSS/ES, the third mostly applied systems assessment tool in the EU, is related to the need to provide information flows among the EU member states and to evaluate how member states carry out legislative measures. Such a mandatory process was defined simultaneously by several European Directives and Regulations, like EU 91/692/EEC (EU, 1991), EU 2003/35/EC (EU, 2003), and Waste Statistics Regulation nº 2150/2002 (EU, 2002). Almost all European countries had already developed MISs at regional and national levels, like in the North Rhine-Westphalia region in Germany, Italy, Denmark (ISAG information system), United Kingdom, and Austria. Such MISs may also improve connections of waste producers and consumers, which have been channeled for possible waste exchange activities almost all over Europe. Many European projects in this direction have been developed. A salient case co-developed by Austria, Belgium, Germany, and the Netherlands is the EUDIN (European Data Interchange of Waste Notification System which is an electronic data interchange platform for waste transportation control within, into and out of Europe boundaries. It consists of an electronic data base that enables an electronic exchange of the data of the notification form and the movement/tracking form (EUDIN, 2002). Such an information sharing platform can be further integrated and improved to develop more powerful ESs and DSSs for waste management. Further justification for the use of MIS/DSS/ES has been for the siting of infrastructures like landfills and incineration plants since some systems engineering models for siting infrastructures can be handled by GIS to make spatial interactions comprehensive and understandable for decision makers instead of domain experts in SWM systems (Chang and Wang, 1996d; Barlishen and Baetz, 1996; MacDonald, 1996b; Haastrup *et al.*, 1998).

3.6 FUTURE PERSPECTIVES OF SYSTEMS ANALYSIS FOR SOLID WASTE MANAGEMENT IN EUROPE

3.6.1 Current status and limitations

The assessment of SWM by using systems analysis techniques allows decision makers to learn about total system complexity. A quantifying *complexity factor* requires evaluating interfaces. Whereas system assessment tools provide a wealth of composite measures of complexity inside procedures/components and between them, joint formulation of human factors and physical/biochemical features in system engineering models in concert with those well defined procedures/components brings about a considerable contribution to the improvement of SWM. Such a successful joint endeavor across the boundaries between models and tools can be evidenced by the development of EUDIN, LATS, Green Dot system, and ORWARE.

Without good practices or guidelines, however, such joint endeavors inevitably tend toward the cost-effectiveness principle. For example, a potential social factor that can drive the implementation of systems analysis toward a cost-ineffective condition is the NIMBY (Not in my backyard) syndrome. More human factors that may tilt the balance/scales of the SWM system should be certainly included to address the impacts from socioeconomic conditions, policy instruments, and regulatory requirements such as EU Directives, national regulations, and regional or local plans and strategies. With the aid of the technology hub proposed in Figure 1, which pinpoints the synergistic effect between system engineering models and system assessment tools, the SWM communities become able to get over the hurdle of system complexity to some extent. A salient case of regulatory requirement is the mandatory EIA and SEA for some specific cases due to the emergency of European Directives 85/337/EEC (EU, 1985) and 2001/42/EC (EU, 2001), respectively. While European Directives with mandatory targets and features are significant, national policy instruments may drive more incentives to achieve the prescribed goals by a more cost-effective, efficient, and forward-looking way. In this pathway, CBA, LCA, MFA, and others may be glued together to support high end analysis.

3.6.2 Gaps on knowledge of waste management

All of these system complexities may encourage the creation of a system of systems (SoS), which may include large-scale concurrent and distributed subsystems in relation to the WHP. In other words, each SoS might be a collection of task-oriented or dedicated subsystems that pool their resources and capabilities together to obtain a more specific goal from an integrated solid waste management perspective. The selected system assessment tools in support of developed scenarios are the workhorses that may enrich the SoS and empower the systems analysis when dealing with contemporary, emerging challenges. These challenges include but are not limited to climate change, resource depletion, and energy crisis as they are the long-term challenges facing SWM communities. Systems engineering models in concert with system assessment tools may be capable of contributing to a fundamental understanding of environmental, technical, economic and social aspects of SWM systems in response to these challenges. To quantify the pros and cons of each alternative at the EU or national level, however, green accounting might be an additional tool for tackling these challenges at different scales.

With the increase of stakeholders' involvement, information flows in and out of the SWM systems increase the complexity. Additionally missing data and information in terms of both quantity and quality can make such high end systems analysis difficult to advance. Thus, the need to effectively and efficiently collect data through identifiable data sources, recognized pathways, and involved agencies, may be justified with respect to the related complexity via the development of MISs, ESs and DSSs. The need for a Quality Assurance System (QAS) to enhance confidence in data and information products can be assured. Such electronic platforms (e.g., MIS, ES, and DSS) would be helpful to support decisions and facilitate research to gain deeper knowledge in SWM. Only this way will it become possible to reach the end-of-waste criteria, supporting waste recycling, treatment and disposal and create a more sustainable management.

3.6.3 Research needs for the future

SDS is a precautionary principle and European directives have reflected it to some extent. However, waste prevention programs at the EU level should have an important role in system analysis because failures can occur in association with environmental, economic and social aspects. These waste prevention plans in SWM systems generally have multi-objective, interactive, dynamic, and uncertain features that complicate the applications of modeling and assessment techniques. The application of EPR, such as the Green Dot system, should be expanded as an integral part of modern SWM systems, since it may provide a possible route to maximize resources utilization and confirm the sustainability. To achieve this goal, carrying out site-specific and process specific CBA, MFA, LCA, EIA, would be required. With these site-specific and process specific inputs, next-generation systems engineering models would be able to reflect environmental impacts through an integrated approach. It should lead to consider more options across waste treatment technologies at all planning, construction, and operational stages, and evaluate more policy instruments to promote waste prevention, reuse and recycling.

To achieve these high-end decision analyses, systems engineering models may be simultaneously and flexibly integrated with system assessment tools in the context of IMS or may be sequentially applied in multiple stages so that the results from one model or tool are the inputs needed for the next one. Given that the New Directive of Waste defines public participation in the assessment of waste management plans and waste prevention programs through SEA, conducting such quantitative decision analyses should include more stakeholders in the decision making process. With this trend, future CBA, LCA, MFA, EIA and SEA might become a multitude of essential models and/or tools that may be mandatory in specific situations. On many occasions, we envision that LCA should be designed based on the framework of MFA since the object to be assessed is a process, not a product. Besides, an MIS would be essential to manage information flows from different sources, support large-scale systems analyses in search of some adaptive solid management strategies, and assess not only technology-based options but also market-based instruments.

3.7 Conclusions

MSW management is normally seen as a major decision making issue with respect to sustainable development in all local communities of the EU. Due to the lack of appropriate system analysis methodologies to define, evaluate, optimize or adapt their waste treatment strategies and to meet the progressive targets set up at the EU level, this paper reviews all the possible trends, and evaluates the present situation of SWM systems in the EU countries in terms of waste processing systems, policy and decision making issues. Facing all regulatory agencies, industrial, and municipalities in the EU, whereas the Southern EU countries (e.g. Portugal, Greece, Spain) require developing measures to implement more integrative SWM systems and reach the objectives of the EU directives, the Central EU Countries (e.g. Germany, Austria, The Netherlands, United Kingdom, France) and certain Northern countries (e.g. Norway) need models and tools in order to rationalize their technological choices and management strategies.

With a thorough literature review and elaborate investigation on how the system analysis techniques were developed and applied in these EU countries, a few future foci in research presented were also organized in this paper for public and private sectors to determine their future strength, thereby achieving the sustainability goals easily. These few milestones required for future development can be carved up front for EU members as follows:

- Deepen the structure of systems engineering models in the context of IMS which may incorporate more multi-faceted features covering economic, environmental, social, ecological, political; cultural, and managerial aspects for the sustainability assessment of current and future SWM systems.

- Provide synergistic tools to account for uncertainty associated with economic, environmental, social, ecological, political; cultural, and managerial aspects for SWM systems.

- Develop large-scale system analysis techniques in order to combine system assessment tools such as EIA, LCA, MFA, and even green accounting with systems engineering models such as optimization models to assess global warming potential, energy saving, and resources conservation practices so as to achieve sustainable waste management goals.

- Investigate both carbon and water footprints for all waste management alternatives as an integral part of system analysis leading to support complicated decision making and policy analyses under the global change impacts.

- Improve current waste management informatics such as MIS, DSS, and ES for the fulfillment of targets of environmental management and data reporting requirements to the EU in the context of cyber infrastructure applications.

- Conduct more cohesive CBA to support resources conservation plans such as Green Dot or the deposit refund systems – packaging waste in SWM systems proposed by European Directives and incorporate more advanced assessments in terms of economic, environmental or social behavior of such systems with the aid of some system assessment tools such as LCA and MFA.

- Expand all ideas described above at different spatial and temporal scales.

With these efforts above, it is believed that the trends of current SWM systems and the perspectives o future SWM in association with the potential applications via integrating a plethora of different systems engineering models with a variety of system assessment tools should lead to improved insights and generate a suite of better management policies and strategies needed for the future.

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CHAPTER IV. LIFE CYCLE ASSESSMENT AT SOLID WASTE MANAGEMENT SYSTEM IN SETÚBAL PENINSULA, PORTUGAL

4 RELIABILITY-BASED LIFE CYCLE ASSESSMENT FOR FUTURE SOLID WASTE MANAGEMENT ALTERNATIVES IN PORTUGAL

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(submitted to International Journal of Life Cycle Assessment)

4.1 ABSTRACT

Background, aim, and scope This paper presents a study related to the application of the reliability-based life cycle assessment (LCA) to assess different alternatives for solid waste management in the Setúbal peninsula, Portugal. The current system includes waste collection, transport, sorting, recycling, and mechanical and biological treatment (MBT) by means of aerobic treatment and landfill. In addition, some future expansion plans are discussed.

Materials and methods The proposed eighteen alternatives were examined with respect to six impact categories based on a customized life cycle inventory (LCI). All the alternatives are designed to comply with the targets prescribed in the Packaging and Packaging Waste Directive and the Landfill Directive. These eighteen alternatives were eventually assessed by using the reliability-based LCA methodology with respect to some uncertain parameters and scenarios.

Results and discussion The results show that solutions based on anaerobic digestion at the MBT followed by energy recovery are the most advantageous options. Overall, recycling may help to avoid most environmental impacts. Alternatives which treat massively biodegradable municipal waste (BMW) are also competitive. In addition to the recycling options, electricity production is also an influential determinant which affects the results. The uncertainty analysis focused on testing different energy-from-waste options (like landfill and MBT biogas electricity production) and different recycling substitution ratios. Such a quantitative analysis is proved effective to confirm the reliability of the LCI in the study.

Conclusions In order to improve the sustainability of the solid waste management (SWM) system, final suggestions may concentrate on the closure of aerobic MBT, the enhancement of anaerobic digestion MBT treatment, and the maximization of energy recovery from high calorific fractions of the waste streams. However, the option of stabilized residue applications can not be encouraged at this stage, especially due to the absence of Portuguese regulations to control the quality of organic products issuing from biological treatment units.

Keywords: *life cycle assessment, municipal solid waste management system, MBT, RDF, uncertainty analysis*

4.2 INTRODUCTION

Life Cycle Assessment (LCA) is an important tool complementing other systems analyses for sustainable development in an urban region. To achieve the sustainability goals, however, the long-established cost effectiveness approach is becoming obsolete whereas cost-benefit analysis may also have to be revised in the future. It is believed that LCA-based planning techniques in concert with cost-benefit information may lead to an insightful assessment of sustainable solutions for solid waste management (SWM).

Applications of LCA techniques to SWM systems in Europe started in the 90's of the last century. The first case of its kind took place for assessing the management of waste beverage packaging systems in Denmark. This pioneering work aimed to determine the best solutions for different packaging waste materials such as paper, cardboard, and glass (Dalager et al. 1995; Ekvall et al. 1998; Frees et al. 1998; Frees and Weidema 1998; Person et al. 1998a, b; Ryberg et al. 1998). The evaluation of the existing technologies to treat biodegradable municipal waste (BMW) was also carried out (Dalemo et al. 1997). Since the beginning of the 21st century, the life cycle inventory (LCI) and the LCA applications have been widely applied to many real world cases due to the fast growth of software packages exclusively for SWM. These include but are not limited to IWM-1 and 2 (White et al. 1995; McDougall et al. 2001), WISARD/WRATE (Ecobilan 2004; Buttol et al. 2007), and EASEWASTE (Christensen et al. 2007). In parallel with the rapid development of such tools, the general-purpose LCA software packages such as SimaPro (Pré Consultants), GaBi (PE International), CMLCA (Leiden University), and TEAM (Ecobilan) have also been customized to conduct a wealth of LCA practices for waste management (Ekvall and Finnveden 2000; Finnveden et al. 2002; Moberg et al. 2002; Muñoz et al. 2004; Finnveden et al. 2005).

Kirkeby et al. (2005) assessed technology-based SWM scenarios for the Aarhus municipality in Denmark and the LCA confirmed that there is no significant difference between anaerobic digestion and incineration based on their LCI. Using a Spanish municipality as a test case, Bovea and Powell (2006) found that the SWM planning scenarios with energy recovery may achieve significant improvements in terms of mitigation of environmental impacts. Rodríguez-Iglesias et al. (2007) showed that incineration without pre-treatment of waste streams is the worst scenario, and proper integration between anaerobic digestion and incineration would certainly lead to a better option. Escalante et al. (2007) and Buttol et al. (2007) pointed out that recycling with energy recovery is one of the most advantagous options from the environmental point of view. Recently, de Feo and Malvano

(2009) discussed the assessment of twelve scenarios showing that a SWM system based on recycling and material recovery without incineration would be preferable. Some studies like that of Tunesi and Rydin (2009) have not reached any final conclusion about which would be the best way to manage municipal solid waste (MSW).

A previous review showed that LCA results for SWM systems do not reach the same conclusion oftentimes, even following the ISO Standards family - ISO 14040 (ISO 2006a), which ensures minimization of assessment discrepancies based on the same set of criteria. This uncertainty may be due to: 1) the methodological assumptions being made are uncertain and may potentially influence the results (Finnveden et al. 2009); 2) the system boundaries considered by these studies, in regard to whether or not to include specific equipment or life cycle emissions of energy consumed by SWM systems may lead to imprecise conclusions (Cleary 2009); 3) the data used for LCA, which translate geographic differences between the data sources and the location of the study, could also make the conclusion biased (Cleary 2009); 4) the lack of data for validation of the LCI applied may end up as increased uncertainty in LCA (Winkler and Bilitewski 2007) and can restrict the conclusions that may be taken from a specific study (Finnveden et al. 2009); and 5) the impact categories assessed, due to the fact that different coverages can affect the conclusions because not all types of category impacts are equally well understood in a typical LCA (Finnveden et al. 2009). Hence, a typical LCA with a variety of conditions may compound state-of-the-art data analytical techniques which would be of importance in the LCA-based decision analysis.

In Portugal, the LCA applications for SWM decision making have been rarely applied. The only application found in the literature is the LCA for the Oporto municipality conducted by Xará et al. (2005). Nevertheless, it is vital to ensure the compliance of SWM according to the European Waste Management Directives, like the Packaging and Packaging Waste Directive 2004/12/EC (EC 2004) and the Landfill Directive 1999/31/EC (EC 1999), both of which brought environmental perspectives into economic decision making. Besides, the New Waste Framework Directive 2008/98/EC (EC 2008) brought new perspectives into waste management based on waste hierarchy. Such hierarchical preference for waste management implies the desirable priority of reduction, reuse, recycling, energy recovery and disposal in sequence. However, when applying the referred waste hierarchy, Member States in European Union (EU) will be expected to take measures to encourage the options which deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy, and justifying life-cycle implications with “system thinking” in association with the overall impacts of the production and management of such types of waste (EC 2008). During the field implementation, site-specific LCA at the local level is essential for the characterization of an all-inclusive or most relevant impact assessment since the LCA outcome being conducted and acquired for other places might not always be transferable.

This paper uniquely develops an LCA to analyze the environmental impacts produced by a SWM system in Portugal. It takes eighteen specific management alternatives into account in a comparative way individually or collectively. These alternatives cover several combinations of biological treatments with or without source separation of BMW. All of the alternatives consider a common issue of packaging waste collection to reach packaging waste recycling targets in the prescribed Directive. The inclusion of LCA in waste management leads to improvement of the understanding of how a SWM system can comply with the legislative requirements and search for the optimal alternatives from environmental and social perspectives simultaneously. In this paper, based on the ISO 14040 Standards, such an LCA was carried out and followed by an uncertainty analysis along with a reliability assessment.

4.3 DESCRIPTION OF THE STUDY AREA

Setúbal peninsula is located in the district of Setúbal with an area of 1,522 km^2 and has 714,589 inhabitants (AMARSUL 2009). The area is divided in nine municipalities, as shown in Fig. 4.1. For being an independent MSW system managed by a regionalization basis, AMARSUL is the company owned by the local municipalities that has been responsible for managing the MSW since 1997. The SWM system is composed of nine recycling centers, two material recovery facilities (MRFs), two landfills, one transfer station, and one aerobic mechanical biological treatment (MBT).

Nowadays, AMARSUL promotes the separation of paper/cardboard, glass and light packaging (plastics, metals and composite packaging) waste by means of curbside recycling systems. Each type of waste is collected separately in three specific containers, and then sent directly to the MRF for recycling, material recovery, and reuse. The remaining waste fractions in households, which is normally destined for final disposal at landfills, are then collected through a door-to-door and/or bin collection scheme, In the case of Sesimbra municipality, the waste stream is first sent to the transfer station, and then finally disposed of at sanitary landfills. The residual waste after waste separation and recycling collected from Setúbal municipality is transported to an aerobic MBT plant where the “stabilized residue” can be converted as fertilizer to be applied as agriculture soil-amendment materials.

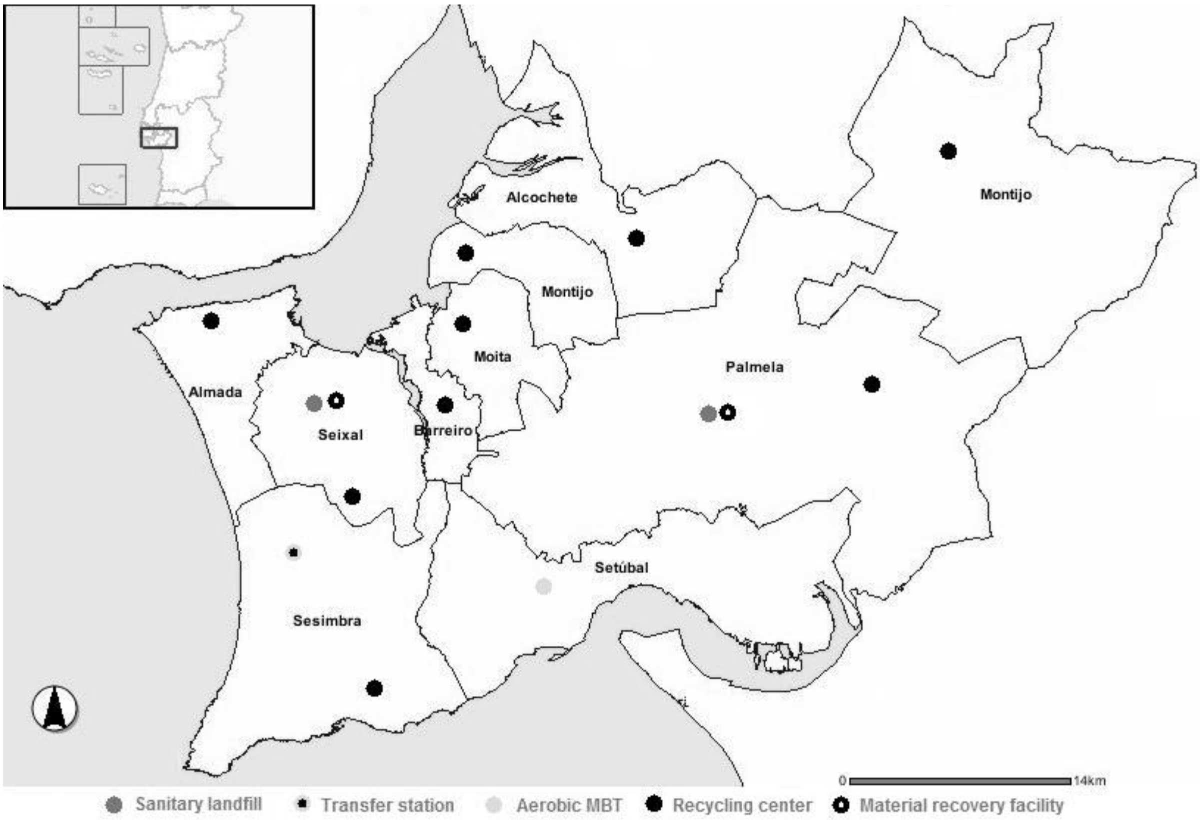

Fig. 4.1 The geographical location of Setúbal peninsula SWM system

Within this MSW system, it has recently become necessary to make some changes in order to comply with the Packaging and Packaging Waste Directive (EC 2004) and Landfill Directive (EC 1999). The National Plan for MSW (designated as PERSU II) decided to pursue the construction of several more MBT units. An anaerobic digestion (AD) MBT unit, with a mechanical treatment to separate recyclables and high calorific material to produce refuse derived fuel (RDF), is under planning. It is expected that this unit will work with two separate lines, one of which is related to the biodegradable municipal solid wastes (BMW) and the other is for the residual waste streams. The RDF may be combusted in an incinerator to generate electricity. The existing aerobic MBT plant will be maintained as usual. It is expected that both MRF plants, which are currently fitted with manual sorting, will be later fitted with two automatic sorting units.

4.4 MATERIALS AND METHODS

In this paper, a customized LCA methodology was developed and applied to conduct a comparison of waste management alternatives for the Setúbal peninsula SWM system. According to the ISO 14040 (ISO 2006a), a LCA consists of four major stages: goal and scope definition, life cycle inventory, life cycle analysis, and interpretation of the results. The following sections present a detailed description of each stage in our application.

4.4.1 Goal and scope definition

The aim of the study was to apply the LCA procedure to the SWM system of the Setúbal peninsula in order to compare waste management alternatives subject to the targets associated with both the Packaging Waste Directive and the Landfill Directive in such a way that could promote sustainable development. A schematic of the SWM to be analyzed is shown in Fig. 4.2, which generally covers all stages of SWM involved from raw waste pick-up to the delivery to bins, to some intermediate processing units, and to the final disposal at landfills. Both anaerobic digestion MBT lines are represented as two separate processes with and without RDF production. The LCA provided in this paper is of attributional type. We applied the "zero burden assumption", suggesting that waste carries none of the upstream environmental burdens into the SWM system (Ekvall et al. 2007).

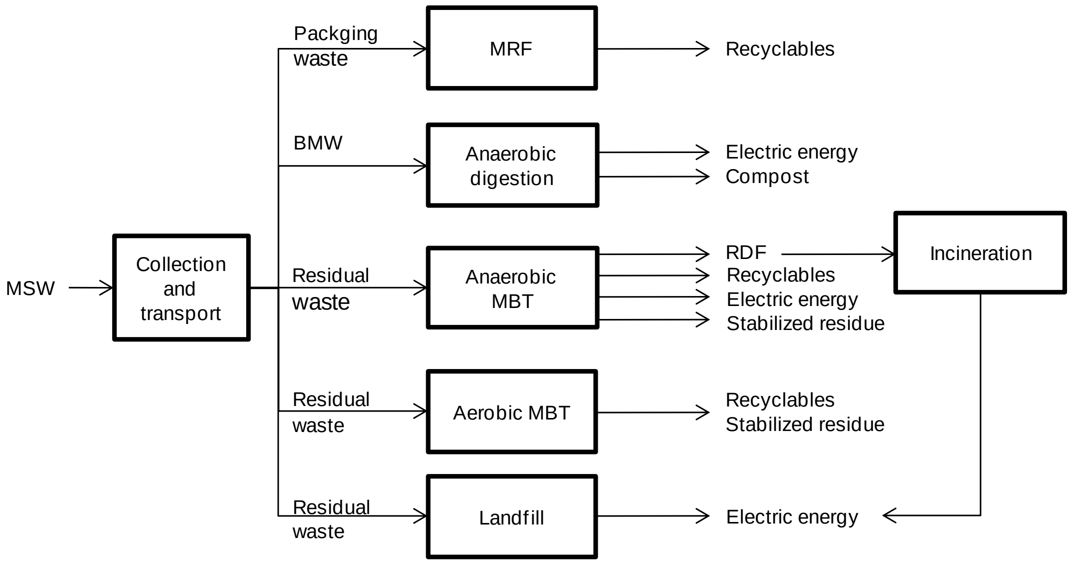

Fig. 4.2 The schematic of SWM system at Setúbal Peninsula

These SWM processes include collection and transportation of residual waste and recyclables, waste treatment, waste transport from waste treatment facilities to the final destination, energy-from-waste or waste-to-energy, and landfilling. Several final destinations for recyclables are located in Spain rather than Portugal, specifically for the cases when handling composite packaging and ferrous and non-ferrous metals packaging materials.

Based on this system, Table 4.1 presents the eighteen management alternatives for assessment plus the current situation (base scenario). These alternatives include waste collection and separate recycling of the three packaging materials through bin systems, which handle 12.4% of the current MSW in the study area. This MRF system is responsible for compliance with the prescribed target in the Packaging Waste Directive.

Table 4.1 The distribution of waste streams associated with each alternative in the SWM system

Fraction (%) Option	Alternatives							
	0/0*/0’	1/1*	2/2*/2’	3/3*	4/4*/4’	5/5*/5’	6/6*	Base
MRF	12.4	12.4	12.4	12.4	12.4	12.4	12.4	4.8
Anaerobic digestion BMW	5.4	0	0	13.3	0	7.5	28.7	0
Anaerobic digestion MBT	28.2	0	33.9	0	49.6	38.9	0	0
Aerobic MBT	13.2	49.7	15.8	32.6	0	0	0	13.8
Landfill with ER	40.8	37.9	37.9	41.7	38.0	41.2	58.9	81.4

* Alternatives considering RDF production plus incineration of high calorific fraction
‘ Alternatives not considering RDF production but incineration of high calorific fraction

Alternative 0 refers to the predicted change that will take place in the Setúbal peninsula waste management system. The remaining alternatives were designed to examine some special options for complying with the Landfill Directive. For example, alternative 1 emphasizes the inclusion of aerobic MBT; alternative 4 signifies the use of AD MBT; alternative 6 examines the specific case of using a BMW anaerobic digestion line. In general, alternatives 0, 3, and 5 are options for differing intermediate processing. Separation of high calorific fractions of waste for energy recovery was considered through the production of RDF and the direct burning of high calorific fractions in municipal incinerators.

The creation of Table 4.1 is based on the total amount of waste produced in 2008, which is 421,726 tonne. According to Finnveden (1999), having identical amounts of waste treated in different scenarios makes it possible to simplify a comparative analysis by neglecting the production and use of the materials. Based on the investigation of average waste composition data of MSW region wide, the waste stream has 31.69% putrescibles, 14.13% paper and cardboard, 11.35% of plastics, 5.83% of glass, 4.14% of composites, 1.82% of metals, 2.07% of wood, 11.72% of textiles, 15.33% of fine particles, and 1.92% of others (EGF 2009). Hardware equipment, such as bins, buildings, and trucks, were excluded from the LCA. However, the use of fuels, electricity, and auxiliary materials for shipping and handling were included in the LCA. The Portuguese electricity generation mix considered is composed of 28.1% of coal, 8.37% of fuel oil, 30.5% of natural gas, 0.55% biomass, hydro 25%, waste 7%, geothermic 0.33%, and wind 0.15%.

In an LCA with multiple products, as in our study, it is necessary to set up the methodological framework. According to the ISO 14044 (2006), the system boundary should be geared toward expanding the product system to include the additional functions related to the co-products to avoid allocation. In this LCA, the material recycling, energy recovery, and fertilizers application (i.e., stabilized residue waste) of MSW were included in the LCA as co-products, which collectively resulted in an expansion of the system boundary. In this LCA, the emissions resulting from the referred operations were included as the baseline information as the emissions of those competing products and energy recovery potential resulting from those alternative operations were also considered for the purpose of comparisons. In this context, the system can be expanded to include additional burdens of co-product processing and the avoided burdens of any processes being dropped (Tillman et al. 1994; Guinée et al. 2002; Thomassen et al. 2008; Finnvedden et al. 2009).

To ensure a correct implementation in regard to the avoided burden through successful MSW recycling and reuse, the co-products in the expanded system boundary should have the same function as the raw products. The substitution ratios are then applied considering closed-loop and open-loop procedures. Table 4.2 presents the substitution ratios for recovered materials and energy consumed. In the cases where the substitution ratio is 1:1, they have been considered as a closed-loop procedure, allowing the hypothesis that no changes occur in the inherent properties of the recycled materials (Rigamonti et al. 2009b).

Table 4.2 Products obtained from the SWM system and the assumptions for LCA

Product obtained	Substitutes assumed	Substitution ratio
Cardboard from recovered paper and cardboard	Cardboard from virgin pulp	1:0.833
Glass produced from recovered glass processed	Glass from virgin materials	1:1
Tubes from PE recycled	Tubes from virgin PE	1:1
Multi-layer packaging materials from recycled PET	Multi-layer packaging from virgin PET	1:0.625
Recycled EPS lightweight soil	Virgin EPS lightweight soil	1:1
Paper from composite packaging materials recycled	Paper from virgin pulp	1:0.625
Outside furniture blocks from recycled mixed plastics	Outside furniture blocks from wood	1:1
Ferrous metals from recycled ferrous metals	Pig iron	1:1
Aluminum ingot from recycled aluminum metals	Aluminum ingot from virgin aluminum	1:1
Compost	N, P, K, Ca and Mg fertilizers	1:1 (based on nutrient content)
Electricity	Electricity mix consumed in Portugal	1:1

For example, 1 kg of recycled glass can replace 1 kg of virgin glass without considering degradation of the material during the recycling so that the quality of the secondary material may not be worse than that of the primary material (Rigamonti et al. 2009b). The materials included in this situation are glass, metals, polyethylene (PE) plastics, plastic wood, fertilizers, and electricity. Specifically, 15 % of the electricity consumed in Portugal was purchased from Spain, and the ratio can also be taken into account, with a proportion of 85/15, for carrying out the LCA. PE, expandable polystyrene (EPS), and plastic wood are specific cases having 1:1 substitution ratio, since they only occur once in the sense that degradation of the material is not considered.

In the cases where the substitution ratio is < 1, an open-loop allocation procedure is applied since degradation of the material should be considered, like the cases of polyethylene terephthalate (PET), paper/cardboard and paper from composite packaging. The calculation of substitution ratios was based on the limited number of times through which a specific material can be recycled and reused repeatedly (Rigamonti et al. 2009a). For PET, the limit number of recycling with respect to losing physical properties considered was five times (Comieco 2008). Concerning paper from composite packaging, the same limit may be applied given that the proportion of paper in those packaging (0.75%) may be assumed and the calculation procedure adopted by Rigamonti et al. (2009b) may be applied. The substitution ratio adopted for PET is from the Institute for Prospective Technological Studies (Delgado et al. 2007).

4.4.2 Life cycle inventory

The Life Cycle Inventory (LCI) is the second phase of the LCA. It is an inventory of input/output data related to the SWM system that is being studied. It involves the collection of the data which is necessary to meet the goals of the defined study (ISO 2006b). In accordance with the scope of the study, an LCI was prepared for the waste management activities specified in Fig. 4.2. Umberto 5.5 software package was used to support the LCA.

Concerning each operational unit analyzed in the AMARSUL system, a short description of the data and assumptions considered for prescribed scenarios is provided. First of all, some of the information applied for our systems analysis was provided by the Empresa Geral do Fomento (EGF), co-owner of the SWM system at the AMARSUL, which is responsible for the management of this MSW system, and the Portuguese Environment Agency (APA). The rest of information was supplied by the Umberto software library and the selected data sources such as machinery specifications provided by the vendors.

4.4.2.1 Waste collection and transport

Waste collection is routinely performed by municipalities. The service can be carried out by municipalities or by hiring private collection companies. MSW is temporary discarded into road side containers (bins), and collection vehicles can remove waste inside the bins periodically. Table 4.3 lists the requirements needed to perform the collection and transport processes. Transportation between those operational units inside the AMARSUL system was not considered since the AMARSUL does not have a BMW collection system. The approach used herein was to assign the same shipping distance and diesel fuel consumption to the municipalities that may treat BMW in a future AD MBT unit in parallel.

Table 4.3 Data requirement for collection and transport waste life cycle stage

Waste collection and transport	MSW	BMW	Packaging waste	Paper/cardboard waste	Glass waste
Distance (km)	1,699,646	121,355	641,334	446,296	179,672
Diesel fuel consumption (l/100 km)	49.6	49.6	65.0	94.6	78.3
References	Gomes and Rodrigues (2010); Pinto (2010); Canta (2010); Aleixo (2010); Didelet (2010); Valério (2010)		EGF (2009); Gomes (2009)		

In the AMARSUL system, the MSW composition is as follows: 70% of food waste, 15% of green waste, 5% of plastics, 1.9% glass, 0.25% ferrous metals, 0.15% of non-ferrous, 0.65% of others and 7.05% of fines, adapted from a BMW characterization program of Lisbon metropolitan area (Vaz, 2009). Packaging waste is composed of 2.45% putrescibles, 10.58% paper and cardboard, 60.8% of plastics, 3.98% of glass, 12.71% of composites, 4.98% of ferrous metals, 0.21% of non-ferrous metals, 0.02% of wood, 1.01% of textiles, 1% of fine particles, and 0.53% of others, provided by EGF (2009). Other default characteristics were collected from literature values like Rotter (2004), Dehoust et al. (2002) and Fricke et al. (2002). Emissions resulting from waste collection and shipping were modeled based on Borken et al. (1999), Knörr et al. (1997), Schmidt et al. (1998), and EMEP/EEA (2009).

4.4.2.2 Sorting plants

Since the AMARSUL is highly likely to have an automatic sorting plant in the future, the technology assessment was carried out for this reason. The packaging waste materials to be sorted are high-density polyethylene (HDPE), low-density polyethylene (LDPE), EPS, PET, mixed plastics, glass, composites packaging, ferrous and non-ferrous materials. Data derived was based on processing one tonne of packaging waste in this recycling operation as shown in

Table 4.4. However, manual sorting will still be employed when handling the paper/cardboard waste streams. Table 4.4 also shows the auxiliary material consumptions during this operation on a per tonne basis and these data are useful for the life cycle impact assessment.

Table 4.4 Operational units consumptions and requirements

Operational units	Operational requirements	Auxiliary materials (per ton waste input in the operation)	References
Packaging MRF	Material recovery rate: 90%	Electricity (kWh): 20.92 Diesel (l): 2.01 Lube oil 8l): 0.20 Steel (kg): 1.20	Rodrigo and Castells (2000); EGF (2009); Rodrigues (2009)
Paper/ cardboard MRF	Material recovery rate: 90%	Electricity (kWh): 5.35 Diesel (l): 0.64 Lube oil (l): 0.01 Steel (kg): 1.20	Rodrigo and Castells (2000); EGF (2009)
AD	Mechanical step: Refuse – 2.8% Ferrous metals recovery rate for recycling – 99%	Electricity (kWh): 34.8 Diesel (l): 1.16(l) Lube oil (l): 0.12	EGF (2009)
	Biological process: Biogas production – 380 m^3/t organic waste Post-composting Decomposition rate – 30% Maturation step: Rejects (%) – 5	Biological process: water (l) – 279 Post-composting: Electricity (kWh) - 10 Structural material (%) – 5 Maturation step: Electricity (kWh): 10 Water (%) – 20	Vogt *et al.* (2002); EGF (2009); APA (2009)
AD MBT	Material recovery for recycling: mainly metals, 95% Material recovery for RDF (when applied): 98 of high calorific material	Electricity (kWh): 34.8 Diesel (l): 1.16 (l) Lube oil: 0.12 (l)	EGF (2009)
	Biological process: Biogas production – 380 m^3/t organic waste Post-composting Decomposition rate – 50% Maturation step: rejects (%) - 10	Biological process: water (l) – 279 Post-composting: electricity (kWh) - 10 structural material (%) – 5 Maturation step: electricity (kWh): 10 water (%) – 20	Vogt *et al.* (2002); EGF (2009)
Aerobic MBT	Material recovery for recycling: glass: 1%, plastic: 7%, ferrous metals: 97%; non ferrous metals: 14%	Electricity (kWh): 34.8 Diesel (l): 0.5 Lube oil (l): 0.12	EGF (2009); Wallmann and Fricke (2000)
	Biological step:	Electricity (kWh): 10	Fricke and Müller (1999);

	decomposition rate - 65% Maturation step: decomposition rate - 20%	Diesel (l): 0.12 Water: 2% for biological step; 20% for maturation Structural material: 8.2%	EGF (2009); Vogt *et al.* (2002)
Landfill	Annual precipitation (JNS): 1550 mm Leachate production during phase A (N24T1) 40% Leachate production during phase B (N25T1) 8% Duration phase A (PHAA) 10 years Duration phase B (PHAB) 20 years	Electricity (kWh): 0.002 Mechanical energy (kJ): 10.99 Heat energy (kJ): 1.6	Rettenberger (1996); Rettenberger and Stegmann (1997); Weber (1990); Eggels and van der Ven (1995); BUWAL (1998)

4.4.2.3 Anaerobic digestion

Within the AMARSUL system, it is expected to adopt a combined MBT unit, in which two separate lines will be laid out to process MSW and BMW, respectively. In regard to the BMW processing line, a small mechanical treatment process will be installed to remove unnecessary matter like metals and plastic waste destined for biological treatment. Organic waste portions delivered to the BMW unit may be decomposed in a thermophylic, dry anaerobic digestion process, resulting in a digestate. This digestate material may be sent to a post-composting unit to have the residual organic waste decomposed, producing fresh compost. To produce useful compost, a maturation phase must be arranged to produce mature compost, which can be applied for agricultural use. The requirements applied to this phase are shown in Table 4.4.

Emissions occurring during the anaerobic digestion mainly result from biogas burning, wastewater treatment, and gas treatment. Biogas can be used to produce electricity and heat in the process, and the amount of emissions can be modeled based on some previous work (Soyez et al., 2000; Vogt et al., 2002). Wastewater characteristics were drawn from Loll (1994; 1998) and Vogt et al. (2002) for a typical treatment process with aeration tank, reverse osmosis, sewage sludge drying through flotation, and dehydration included (EGF 2009). To model such a wastewater treatment plant, data from Martinho et al. (2008) and Yamada and Jung (2007) were used. The biogas treatment process may be simulated and predicted based on a biofilter, in which the average air pollutant concentration and treatment efficiency were applied with the aid of literature data (den Boer et al. 2005).

4.4.2.4 Anaerobic digestion MBT

The AD MBT will be located in the Seixal municipality. This unit is composed of mechanical sorting to remove recyclables and combustible fraction for RDF production, allowing the remaining fractions to be sent to the anaerobic digestion unit. Normally, the mechanical sorting process includes flail mills, trammels, magnetic separator, eddy currents separator, and ballistic separator. Sometimes manual sorting is included too to separate materials for recycling and RDF. Table 4.4 lists the material consumption and requirements that are needed to simulate the process.

After being sorted, the remaining fractions of waste with mechanical and biological recovery potential may be treated by the thermophylic, dry anaerobic digestion resulting in a digestate with several decomposed substances. The residual parts may be decomposed further through the use of an aerobic treatment process. It may lead to the production of fresh compost. After this process, fresh compost is still not mature, and it must be deposited in piles for eleven more weeks, to produce mature compost. The main parameters used to model the AD phase are listed in Table 4.4. The biogas produced as an integral product of the AD MBT process may be used to generate electricity. The final residuals may be used as daily cover materials at landfills. The engineering design basis applied to model the emissions in AD MBT was considered the same as those applied in anaerobic digestion of BMW.

4.4.2.5 Aerobic MBT

An aerobic MBT is composed of a mechanical sorting processing unit and a biological treatment processing unit, respectively. The mechanical processing unit is designed to remove the waste stream that is not relevant for the biological treatment unit. Concerning the mechanical separation, which also includes manual sorting, the materials removed for recycling are mainly ferrous and non-ferrous metals as well as some glass and plastics.

In an aerobic treatment process, the requirements applied to decompose organic fraction of waste are presented in Table 4.4. From such a MBT process, the main output is the "stabilized residue", which must be landfilled or used as the daily cover materials in landfills. In this MBT, there is no wastewater generation and contaminated air may be treated by a biofilter. The engineering design basis applied to model this biofilter was considered the same as those applied for the other similar biological treatment processes previously described.

4.4.2.6 Landfill

The waste stream which goes to a sanitary landfill has different sources varying from mixed MSW to residuals associated with several operational units in the MSW management system. The emissions from landfills diffuse into air, soil, and water. Typical sanitary landfills have two types of collection systems. One is for leachate collection and the other is for biogas collection. The existing one in the AMARSUL system is for the collection of biogas (i.e., methane gas) collection to produce electricity. Landfill methane gas emissions are due to biological decomposition and meteorological conditions at the local scale. The landfill module in UMBERTO was then built based on several sources in this study (Rettenberger 1996; Rettenberger and Stegmann 1997; Weber 1990; Eggels and van der Ven 1995; BUWAL, 1998). The formula used to quantify the methane gas production was derived based on some literature values as adapted below (Tabasaran and Rettenberger 1987):

$$Ge = 1.868 * Co * (0.014\ T + 0.28),$$

in which Ge is the potential methane gas production at long term [m^3/t waste], 1.868 is gas production rate resulting from decomposition per kg of organic waste [biogas/kg C] (note that [(22.4 L biogas/mol)/(12 g C/mol) = 1.868 L biogas/g C]), and Co is the content of the organically degradable carbon in waste in household waste [kg Co/t waste] (i.e., typical figures are 170 to 220 kg/t) Co [m^3 biogas/kg Co. Within the current model Co is calculated based on the C content of biologically degradable organic waste. (0.014T + 0.28) is temperature dependent decomposition rate [in °C] (note that for household waste landfill denominated by T lies between 30 to 35 °C).

Air emissions due to biogas management can be attributed to direct emissions, from burning biogas and diffuse emissions from landfill. Diffuse emissions are linked with the arrangement of biogas collection system during landfill operation and post-closure (phases A and B, respectively). Based on Umberto module, it has been considered 25% of the biogas collected is released through direct emissions. During phase A, it has been considered that 30% of biogas is released. During phase B, this number is potentially up to 70%. It is assumed that for entire landfill life around 50% of biogas from phases A and B is actually produced. Hence, in phases A and B, we have (75/100)(30/100)(50/100) = 11.25% and (75/100)(70/100)(50/100) = 26.25% of diffuse biogas. For the amount of collected biogas, it may be estimated as (75/100)(50/100)(1-70/100) for phase A and (75/100)(50/100)(1-30/100) for phase B.

Landfill gas (LFG) recovery happens by using a gas turbine. The emissions from LFG burning and electricity production were calculated based on the average values collected by den Boer *et al.* (2005). In regard to landfill leachate, its production is also divided into phases A and B. The temporal horizon is 100 years. Leachate production level depends on annual average precipitation as well as water content inside landfill. In operation phase (phase A), leachate production can be estimated in between 10-50% of total annual precipitation (Schwing 1999). After closure (phase B), leachate production can be as low as 5-10% of total annual precipitation. In UMBERTO module, default values for phase A is 40% and for phase B 8% (Rettenberger and Schneider, 1996). Such values are applied for every type of waste. It is assumed to have a residual water content of 15% by weight in which only 24% may end up diffuse emissions, implying that only 76% may be collected, based in German landfills (Schwing 1999). It is also assumed that leachate collection system can collect 90% of leachate produced. If landfilled waste has a density that is equal to 1 t/m^3 land use requirement can then be determined by the ratio between the volume of waste landfilled and the soil given that 20-meter of height was applied.

4.4.2.7 Products shipping

Recyclables, compost, high calorific fraction and RDF resulting from MSW management system have to be transported to their final destination. The shipping distance parameters are listed in Table 4.5. These distances were obtained by using the Google map tool (Google maps 2010) and diesel consumption record collected from transportation companies based on 25 liters/100 km (JMFF 2008).

Table 4.5 Distances between MSW management system and final ends for products

Products transport	Distances (km)	
	Pre-processors	Recyclers/Incineration[1]/Agriculture[2]
Ferrous metals	241.3	521.5
Non-ferrous metals	259.3	592.2
PE	0	238.6
PET	0	210.7
EPS	0	293.0
Mixed plastics	0	524.0
Paper/cardboard	339.9	811.2
Composites	210.2	1116.5
Glass	233.0	60.5
RDF[1]	0	45.4
Compost[2]	0	73.7

4.4.2.8 Auxiliary materials and recyclables

Auxiliary materials like electricity, diesel production and burning, and lubricating oil consumption in MSW management systems were discussed by Frischknecht *et al.* (1996), GEMIS database (GEMIS 2001), ifeu (2009), EMEP/Corinair (2007) and EMEP/EEA (2009). In the case of lubricating oil, the data used in this study were adapted from Martinho and Pires (2009). The rest of auxiliary materials used in this study were drawn from literature. Expansion of the system boundary due to the processing of recyclables is summarized in

Table 4.6. The specific auxiliary materials used during recycling processes were modeled as well based on relevant items in Table 4.6.

Table 4.6 Summary of LCI data sources for expanded systems and avoided products

Type of Data	Sources of Data
PET recycling, mixed plastics recycling, glass pre-processing and glass recycling	Alves (2010); ProBas (2004); APA (2009); Mata (1998)
RDF production	Fricke *et al.* (2003)
RDF incineration	UBA (1999); Achernbosch and Richers (1997; 1999); Schäfl (1995); Valorsul (2008)
Paper and cardboard pre-processing, composites packaging pre-processing	Rodrigo and Castells (2000)
Paper and cardboard recycling	ProBas (2004); APA (2009)
PE recycling	Arena *et al.* (2003)
EPS recycling	Silva (2010)
Composites recycling	Stora Enso (2008)
Ferrous metals pre-processing	Rodrigo and Castells (2000)
Ferrous recycling	ETH Zurich (2008)
Aluminum metals pre-processing	Rodrigo and Castells (2000)
Aluminum recycling	Boustead (2000)
Auxiliary materials production	APA (2009); APME (1995); Patyk and Reinhart (1997); BUWAL (1998); Ecoinvent (1996); GEMIS (2001)
Avoided products, including fertilizers	APA (2009); BUWAL (1998); ProBas (2004); Mata (1998); ifeu (1994)

4.4.3 Life cycle impact assessment

Our LCA was then carried out using the Umberto 5.5 (2009) software package with the aid of the entire LCI as described in the previous section. Following the methodology suggested by the ISO 14040-44 standard (ISO 2006a, b), environmental indicators were obtained for different impact categories. The characterization factors applied to each impact category are those proposed by the CML 2000 method (Guinée, 2002). The impact categories studied were: abiotic depletion, acidification, eutrophication, global warming, human toxicity, and photochemical oxidation. Life cycle impact assessment can then be carried out by linking these designated impact categories with those prescribed operational efforts in Table 4.1.

The differentiated contribution of each operation unit associated with each alternative can be summarized in Fig. 4.3. All of them consider the compliance with the Packaging and Landfill Directives as priority. All of them are compliant with the Packaging and Landfill Directives. Finally, the ultimate environmental impact in terms of each selected life cycle impact category associated with these eighteen alternatives can be independently calculated and presented by a comparative approach in Fig. 4.4. The following subsections discuss the pros and cons of each alternative with respect to the ultimate environmental impact associated with these designated impact categories.

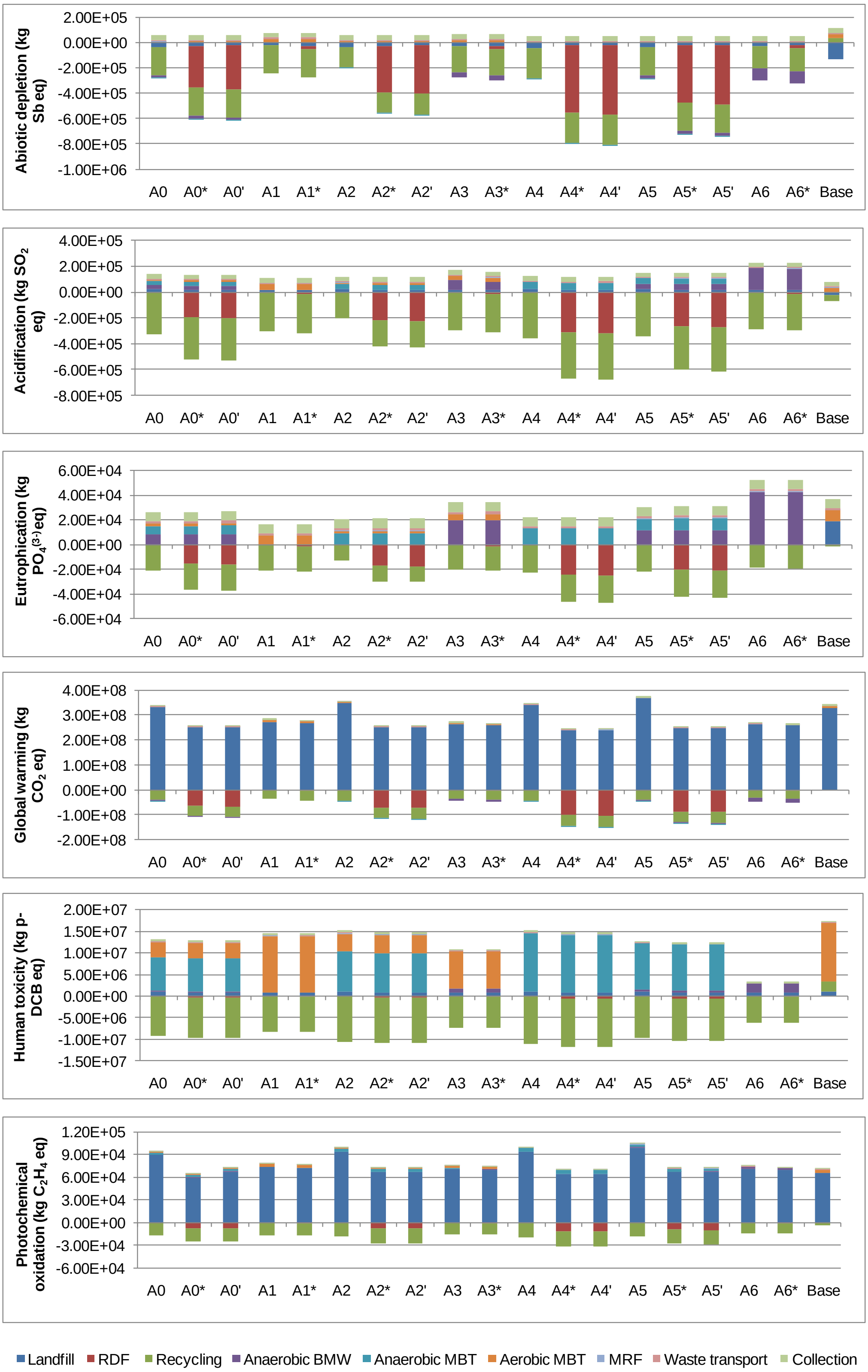

Fig. 4.3 Contribution made by each stage of the waste management life cycle to each impact category

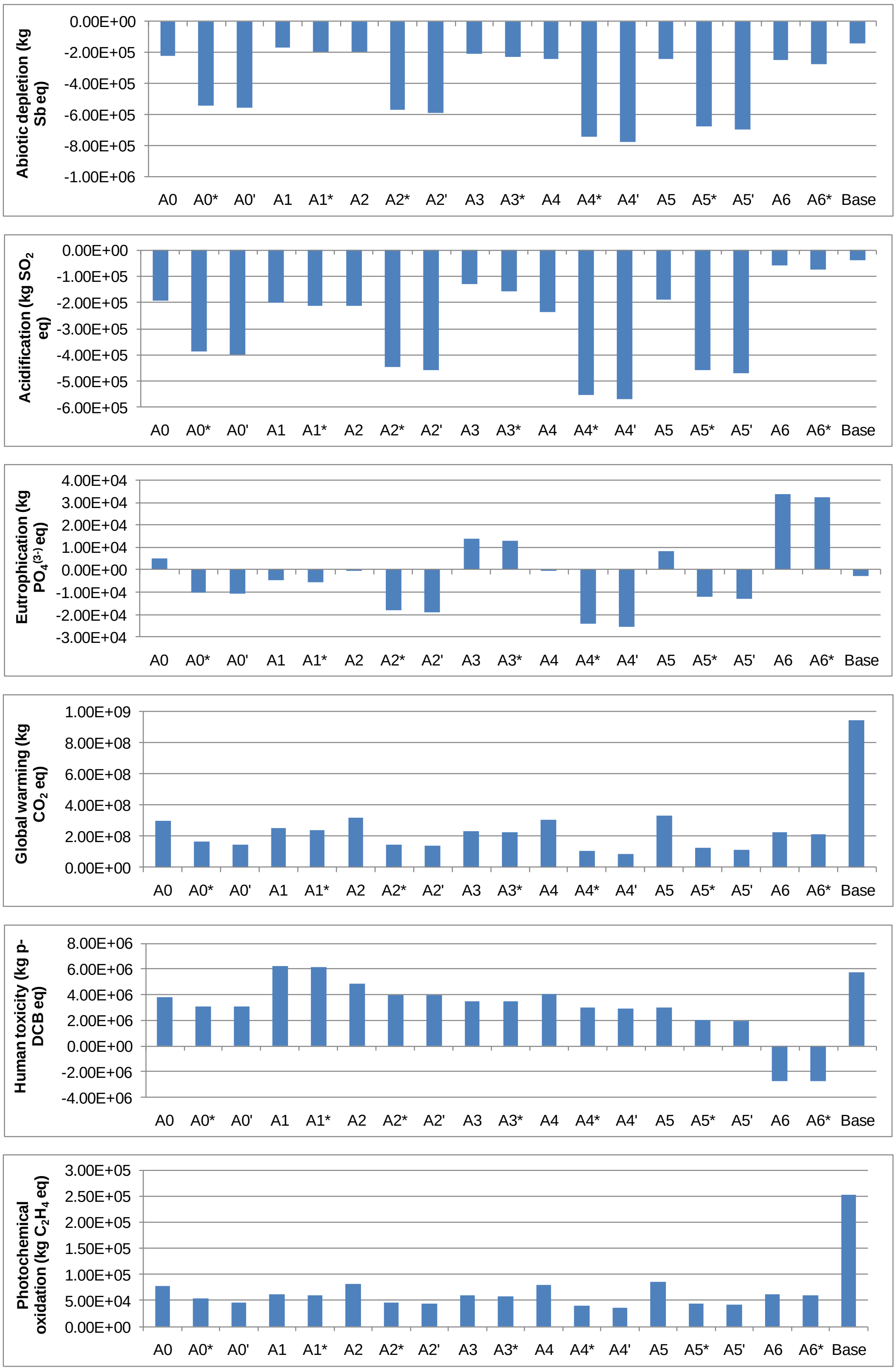

Fig. 4.4 Net contribution of each scenario to each impact category

4.5 DISCUSSION OF IMPACT ASSESSMENT

4.5.1 Depletion of natural resources

This impact category indicator is related to the extraction of natural resources (including energy resources) such as iron ore, crude oil and wind energy, which are regarded as non-living materials (Guinée et al., 2001). The alternatives being assessed in our case study exhibit clear traits. It is indicative that those alternatives that have more options for resources substitution end up having better environmental performances. Those preferred alternatives include A4', A4*, A5', A5*, A0', A0* and A2' and A2* that can produce more electricity from high calorific fraction direct burning or RDF, and biogas combustion. Naturally, alternatives with higher recycling rates due to separation at the MBT plants are favored in comparison with those that can only recycle packaging waste at the source location. Landfills with methane gas recovery leading to the generation of electricity and the avoidance of the consumption of fuel resources were favored too. The worst scenario is the base scenario which is literally the current situation.

4.5.2 Acidification

Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, and ecosystems and materials (buildings) (Guinée et al. 2001). Since all alternatives present the impact of acidification due to the lower emissions of NOx, SO2, and ammonia, alternatives A4', A4*, A5', A5*, A2', A2*, A0' and A0* particularly signify such cons due to the combustion of RDF and high calorific fractions of waste, which substitutes energy production in the power industry. On the contrary, alternatives A6 and A6* present less advantages due to more compost production given that compost application induces the release of ammonia, and consumes more energy via burning more fuel. The base scenario is the worst one again in this impact category.

4.5.3 Eutrophication

Eutrophication covers all potential environmental, ecological, and public health impacts due to the presence of nutrients including nitrogen and phosphorus species. Nutrient enrichment may cause an undesirable shift in species composition and surplus biomass production in both aquatic and terrestrial ecosystems (Guinée et al., 2001). The alternatives with lower release of nutrient substances include A4', A4*, A2', A2*, A5', A5*, A0' and A0*. Those alternatives were picked up because of the reduced emissions of these nitrogen substances via burning

RDF and high calorific fractions of waste. However, A1 and A1* were screened out since there is no wastewater discharge during the operation of an aerobic MBT unit since the AMARSUL can reuse all wastewater effluents. This explanation can also justify the good results in the base scenario. Alternatives that consider compost production and soil amendment (A6, A6*, A3 and A3*) are penalized for the same reason as in the acidification category impact assessment.

4.5.4 Climate change

Climate change is defined as the impact of human emissions on the radiative forcing (i.e. heat radiation absorption) of the atmosphere. Most of the climate relevant emissions enhance radiative forcing, causing the temperature of the Earth's surface to rise, which is referred to as the "greenhouse effect" (Guinée *et al.* 2001).

Recycling contribute to reduce global warming potential (GWP) substantially across all alternatives. The other important factor that is intimately linked with GWP is the amount of BMW that goes into landfill and the amount of RDF production and burning. Alternatives A4', A4*, A5', A5*, A2', A2*, A0' and A0* were selected as the best alternatives due to the production of electricity by using biogas and high calorific fractions or RDF. Besides, all these selected alternatives can redirect considerable amount of waste streams that would otherwise be destined for landfilling leading to the generation of more GWP. This is why alternatives A4, A5, A2, and A0 were not favored as well as the base scenario.

4.5.5 Human toxicity

This impact category is related to the negative impacts of the toxic substances released to the environment on human health. MBT plants have higher potential of emissions of heavy metals, thereby creating negative environmental impact. The emissions from MBT plants are related to indirect emissions not only from electricity and auxiliary materials production (lube oil, diesel), but also from wastewater produced during anaerobic digestion in the MBT process. Once organic waste from source separation has a lower heavy metals content than organic waste from commingled MSW, leaching effects during decomposition are considerably higher at an MBT than the process during anaerobic digestion after source separation, which renders the alternatives A6 and A6* as the cases with lowest negative environmental impact on human health.

Because of the lack of generation of wastewater in an aerobic MBT, alternatives A1 and A1* are not favored. This can be explained by the fact that the avoided sub-systems (i.e., wastewater treatment) are not enough to compensate the heavy metals being emitted from an aerobic MBT and the consequence of no electricity production using biogas. In general, other biological treatment plants in dealing with selective organic waste fractions may have considerably lower emissions of heavy metals and hydrocarbons when compared with MBT units.

4.5.6 Photochemical oxidation

Photochemical formation is the formation of reactive chemical compounds such as ozone in the troposphere, resulting from the reaction when sunlight interacts with some primary air pollutants. These reactive compounds may be harmful to human health and ecosystems, and may also damage crops. The relevant areas of protection are human health, man-made environment, natural environment and natural resources (Guinée et al., 2001). The major fraction of tropospheric ozone formation occurs when nitrogen oxides (NOx) and volatile organic compounds (VOCs) react triggered by sunlight. NOx emissions are common in combustion processes. Therefore, the alternatives A4', A4*, A5', A5*, A2', A2*, A0' and A0*, which divert more waste streams for RDF production or direct burning of high calorific fraction via AD MBT, may present better environmental performance in terms of the generation of NOx because less fossil fuels will be needed for combustion in power plants. Consequently, alternatives A4, A5, A2, A0 and base case were not favored due to the need of more electricity produced by using fossil fuels for operation.

Overall, recycling may contribute to avoid most environmental impacts. Alternatives using AD MBT burning of high calorific fractions of waste directly in incineration are classified as better solutions with a minimal difference from massive RDF production alternatives. This is due to the reduction in electricity consumption. Alternatives with high energy-from-waste potential exhibit comparative advantages over most of the counterparts such as aerobic MBT alternatives. Alternatives A6 and A6* with the inclusion of the recycling organic waste fraction are also competitive. Waste collection, transport, and sorting process present similar environmental impacts in terms of energy consumption.

4.5.7 Uncertainty analysis and reliability-based LCA

Most LCA practices aforementioned were based on the known or assumed data without the consideration of uncertainty. According to ISO 14040 (2006a), uncertainty analysis consists of a systematic procedure to quantify various sources of uncertainty introduced from many aspects via cumulative effects. They include the data variability, scarcity and imprecision, as well as model or methodological uncertainty. In this study, the proposed uncertainty analysis focused on addressing both uncertainties associated with modeling assumptions and data variability.

One way to assess modeling assumption is to change the database being used to support modeling the operational units and auxiliary processes. Those databases apply different reference basis to reach the input-output balance. In this study, a database created by PE International (2010a, b) was applied to model Portuguese landfills in 2006 in which the electricity module was developed in 2002. After analyzing the possible changes of landfills and associated electricity production, Fig. 4.5 clearly indicates that such changes could significantly affect the LCA outcome. With these changes, alternatives A4' and A4* are still the best options in terms of abiotic depletion, acidification, eutrophication, global warming and photochemical oxidation. As for human toxicity, A4' and A4* still appear to be the best, which is significantly different with the previous results in LCA. This difference is related to the electricity database developed by PE International, which presents higher amounts of heavy metals when compared with GEMIS/ifu data applied earlier. Similar effects are evident in other impact categories, like acidification and eutrophication.

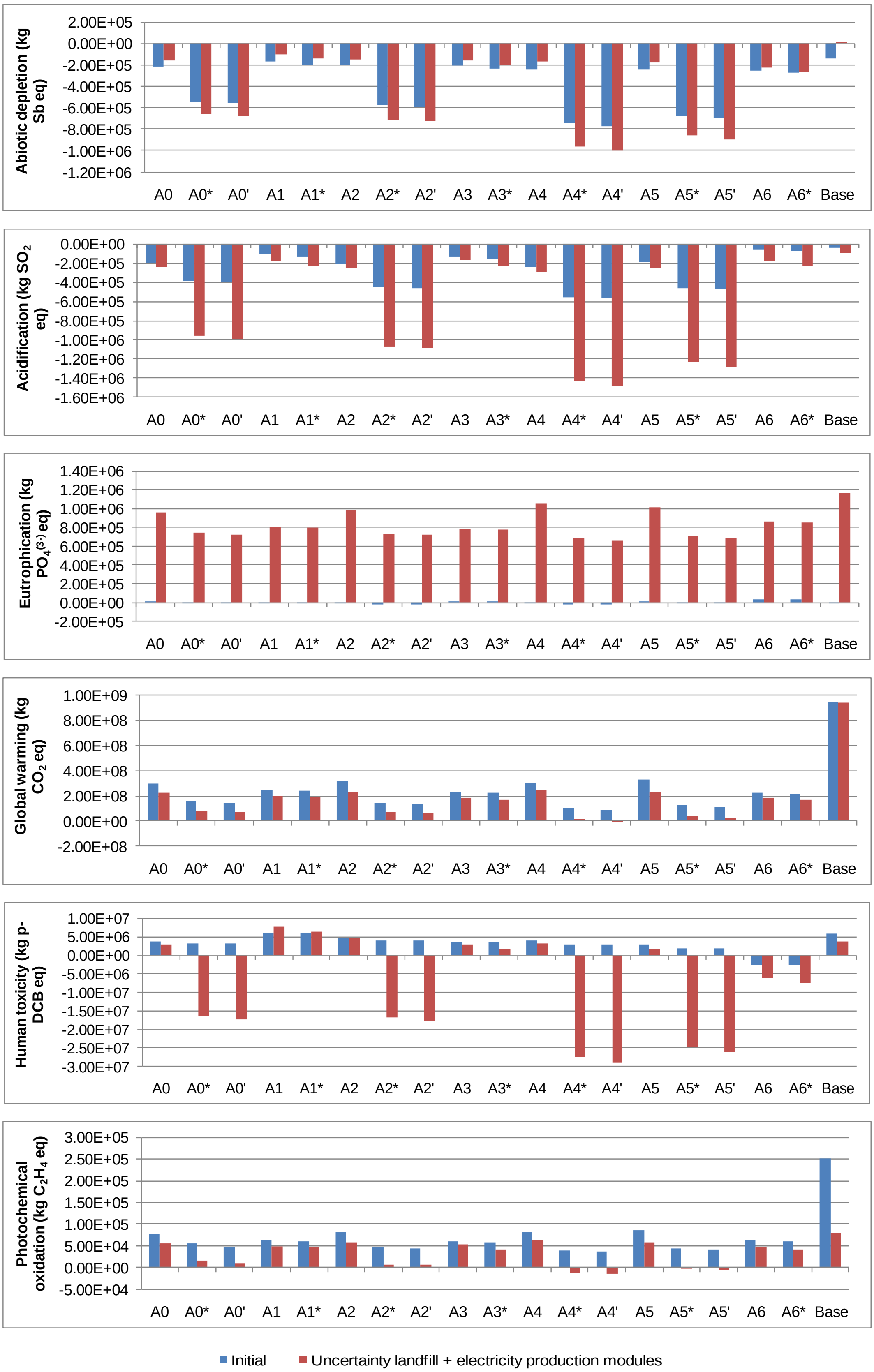

Fig. 4.5 Comparison of results obtained by modifying electricity mix and landfill modules

Data variability was simulated by adjusting the levels of biogas production at these MBT plants and electricity consumption during paper/cardboard recycling by means of a Monte

Carlo simulation practice based on uniform distributions via 10,000 iterations. The range of data simulated for biogas production at the anaerobic digesters is between 320 and 450 m3/ODM (ODM: organic dry matter). Such a range may be seen in multiple sources, including literature values (Weiland, 2000) and statements given by some EGF's experts. Besides, the range of electricity consumption for paper recycling varies from 2,592 to 6,500 kJ/kg based on literature values (EIPPCB, 2001) and sources from the Portuguese paper/cardboard recycling industries (APA, 2009). When applying LCA to this region-based study, Fig. 4.6 and Fig. 4.7 collectively reveal such data variations would not significantly alter the LCA results.

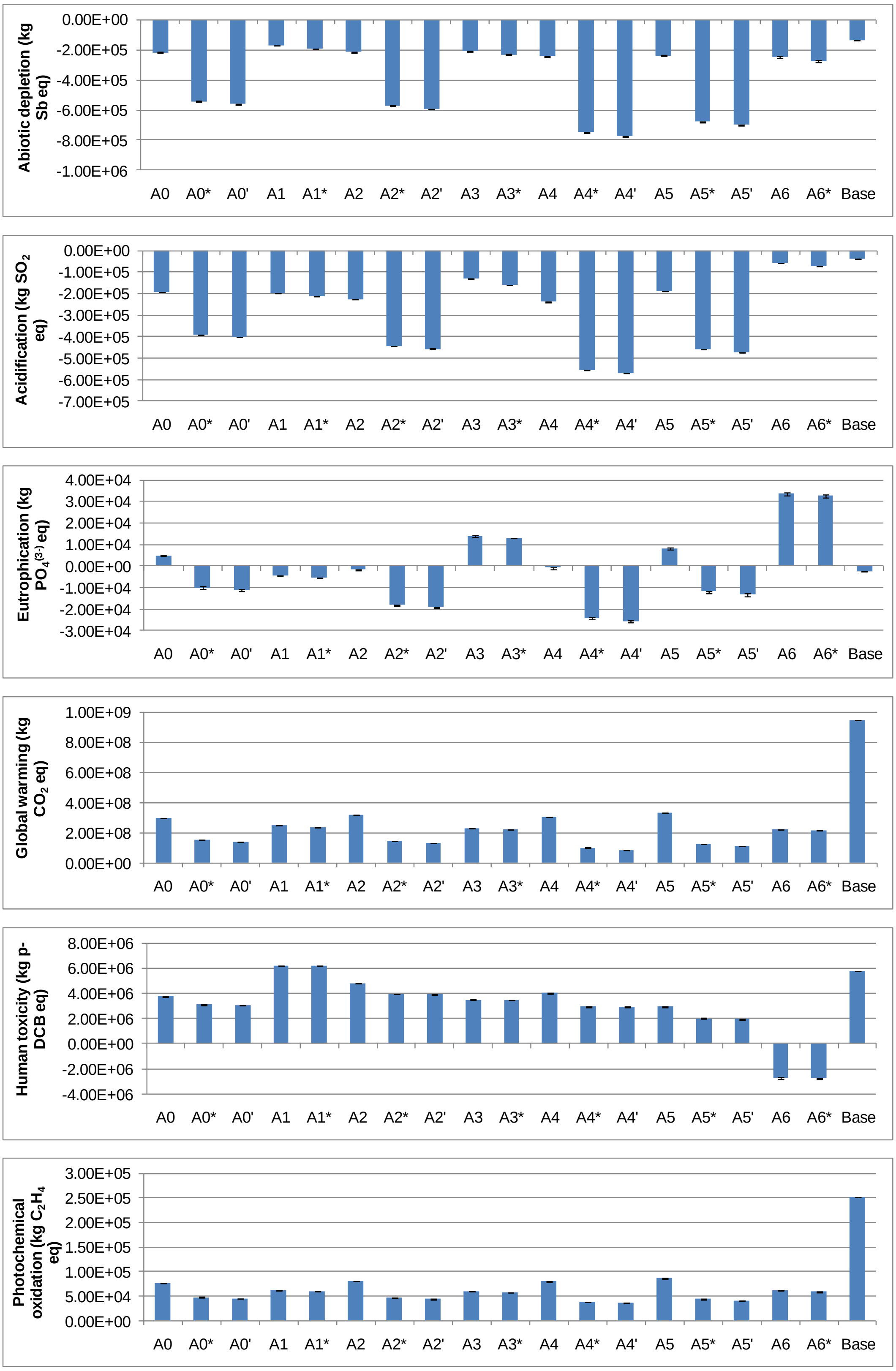

Fig. 4.6 Comparison of results obtained by testing biogas production in anaerobic digestion MBT processes

In addition to the concerns about data variability, there are many other dimensions of uncertainty analysis in LCA. Our reliability-based LCA is tied to the three scenarios being studied and all of them need to be tested for the robustness of the LCA outcome. The first test can be set up to answer the question: “how will the LCA results be affected by using different Portuguese electric energy LCI?” To explore this further, possible changes can be made based on the GEMIS database of Portuguese electric energy for 2030, available at ProBas (2004). The expected Portuguese electricity in 2030 has coal 30.9%, crude oil 1.18%, natural gas 35.6%, waste 4.36%, biomass 2.5%, geothermic 0.24%, hydro 24.3%, and wind 1%. Such a reliability-based LCA study can then be characterized by an increased use of non-fossil fuels for power generation that would affect the credit of RDF production. As the result of such changes, Fig. 4.8 reveals that alternative A4’ is not only the best option in terms of abiotic depletion, acidification, and global warming, but also the leading one, far ahead of the rest of the preferred alternatives such as A5’, A2’ and A0’.

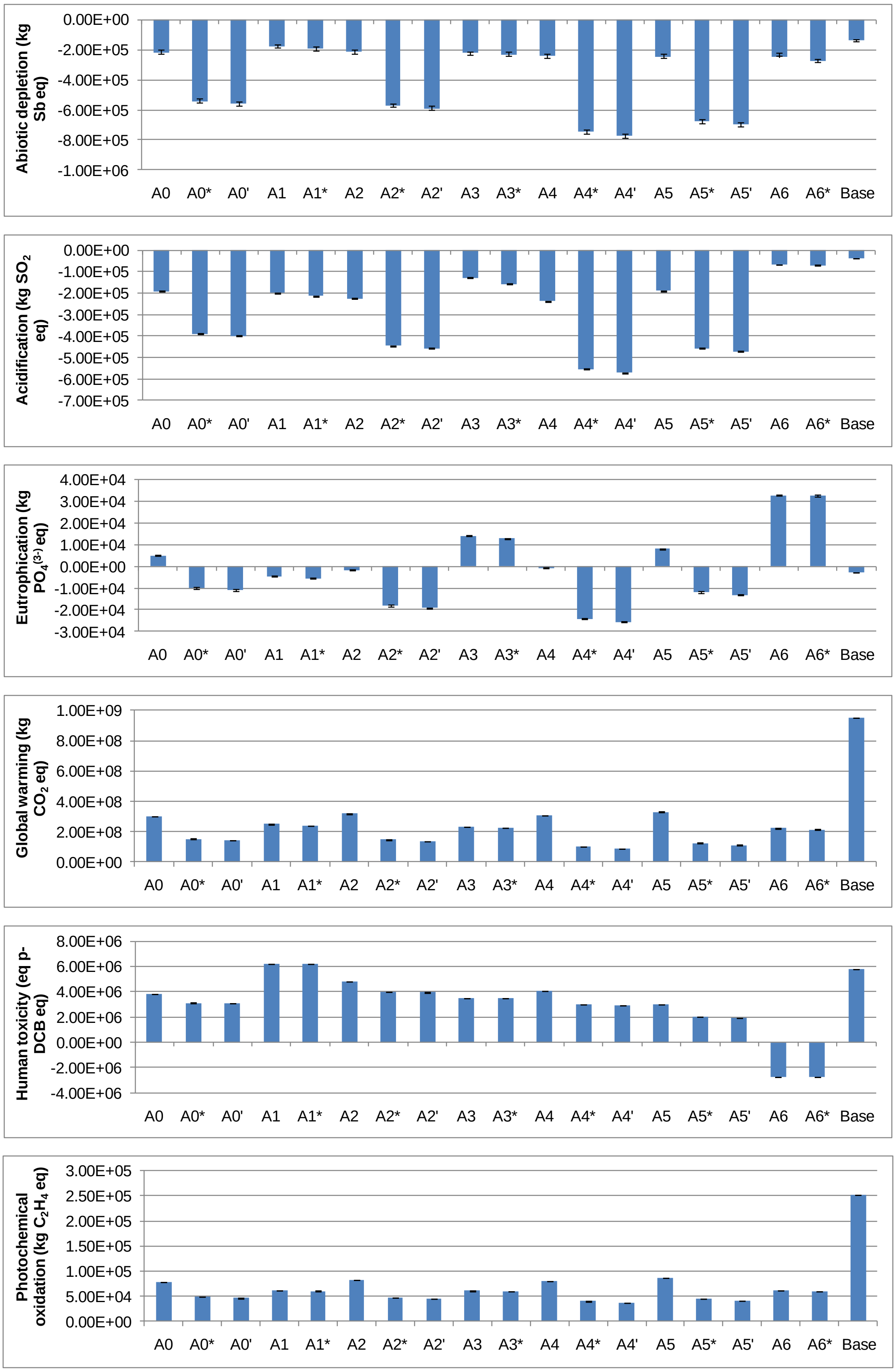

Fig. 4.7 Comparison of results obtained by testing electricity consumption in paper/cardboard recycling process

The testing of the second scenario is related to the selected substitution ratio of some recyclables as indicated in Table 4.2. In such a reliability-based LCA study, changes due to the inherent differences of properties between these recycled materials and virgin materials were ignored, thereby ending up a ratio of 1:1 for testing (Bovea *et al.* 2010; Rigamonti *et al.* 2009). If this is not the case, Fig. 4.8 also confirms that alternative A4' is still the best option in terms of abiotic depletion, acidification, global warming, eutrophication, and photochemical oxidation. The third testing scenario deals with the situational awareness that the biological treatment units may collect a fair amount of recyclables after the installation of the automatic sorting equipment for RDF production. If we remove this premise, all the residual materials will be destined for landfilling based on the average values as shown in Table 4.7. Simulation results assure that both A4' and A5' have equivalent advantages in terms of abiotic depletion, global warming, and photochemical oxidation, both of which may be selected as the best options.

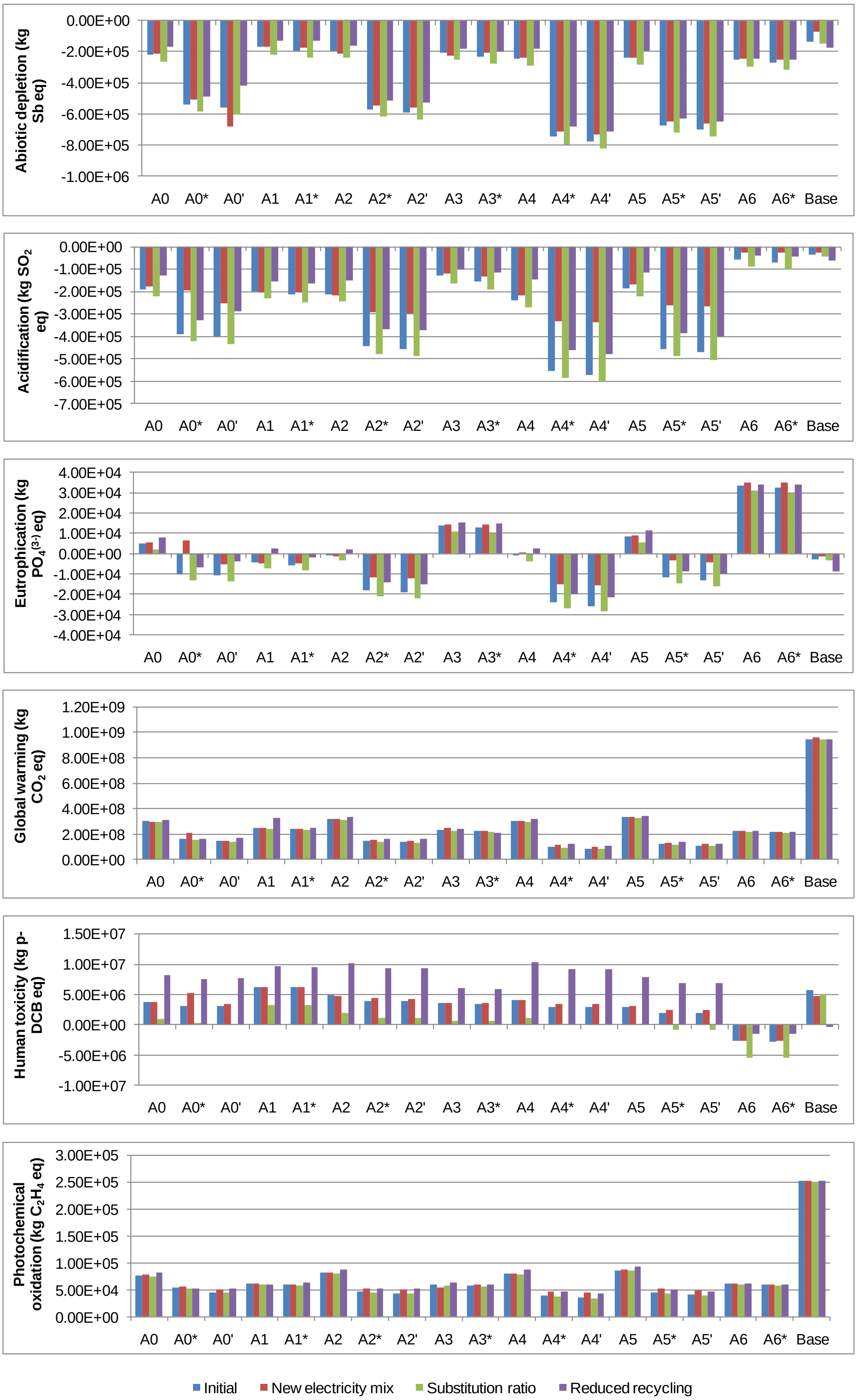

Fig. 4.8 Comparisons of results obtained by modifying electricity mix, substitution environmental performances

Table 4.7 Reliability analysis considering different quantities of recycled materials

Recycled materials	Initial quantity (t)							Uncertainty analysis quantity (t)
	A0 group	A1 group	A2 group	A3 group	A4 group	A5 group	A6 group	
Glass	17,700	17,600	17,815	17,590	17,920	17,920	17,475	17,475
PET	1,890	1,950	1,910	1,895	1,880	1,880	1,780	1,780
EPS	200	200	200	200	200	200	200	200
PE	2,340	2,580	2,410	2,350	2,310	2,310	1,900	1,900
Mixed plastics	2,310	2,340	2,320	2,310	2,310	2,310	2,255	2,255
Composites	1,250	1,140	1,260	1,135	1,320	1,320	1,140	1,137
Ferrous metals	3,310	3,700	3,590	2,730	3,645	3,770	1,322	1,060
Non-ferrous metals	715	460	810	480	980	980	535	365
Paper/cardboard	20,340	19,650	20,410	19,650	20,755	20,755	19,655	19,650

4.6 CONCLUSIONS

The eighteen alternatives that address the current practices and possible future expansion options in 2013 were analyzed and compared with each other in the present study based on the existing LCA technologies. Initial findings clearly indicate that combination of anaerobic digestion and MBT followed by the energy recovery of the high calorific fraction of waste is an advantageous option to manage MSW, since it may not have detrimental effects in terms of abiotic resources depletion, acidification, global warming, and photochemical oxidation. Options from which the anaerobic digestion of BMW was considered can simply contribute to the reduction of the human toxicity impact. The environmental advantage of the production of RDF that is compared with direct burning of the high calorific fraction of waste in incinerators was not salient. In this case, RDF production can be justified more from a point of view of shipping advantage rather than from an energy-from-waste process itself. In fact, the LCA results show that the promotion of biological treatment is a better solution, especially when energy recovery is considered for electricity production. However, none of the alternatives studied are favored across all the impact assessment categories considered. The existence of two lines, in anaerobic digestion MBT, as biological treatment options, in which one for BMW and the other for MSW, is a positive option at least from the environmental point of view. However, the environmental impacts related to compost application have not yet been quantified fully in terms of carbon sequestration in soil and soil erosion prevention that could bring up some more positive effects and a significant environmental advantage. Hence, with reservation, we would not encourage stabilized residue applications at this juncture.

Reliability-based assessment contemplates the influence with respect to three scenarios related to electricity production, varying substitution ratio for recycling, and ignorance of recyclables that can be possibly obtained at the MBT plants. In particular, the analysis confirmed the efficacy of reliability-based LCA. The uncertainty analysis was concerned with the biogas production at a MBT plant and the electricity consumption in paper/cardboard recycling both of which are related to data assumptions. To explore the modeling assumptions, uncertainty analysis focused on the use of different input-output modules for landfill operation and electricity production. With changing data assumptions, the LCA still maintains the options previously chosen without regard to the uncertainties of concern. Yet the reliability-based assessment in dealing with modeling assumptions brings up different amounts of air pollutants in the assessment. This resulted in changes in the category impact of human toxicity, which is significantly different to the previous results in LCA. It can thus be concluded that the three scenarios in the context of reliability-based LCA significantly contribute to the contemplation in decision making.

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CHAPTER V. MULTICRITERIA ANALYSIS OF SOLID WASTE MANAGEMENT SYSTEM IN SETÚBAL PENINSULA, PORTUGAL

5 AN AHP-BASED FUZZY INTERVAL TOPSIS ASSESSMENT FOR SUSTAINABLE EXPANSION OF THE SOLID WASTE MANAGEMENT SYSTEM IN SETÚBAL PENINSULA, PORTUGAL

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5.1 ABSTRACT

Recent challenges in solid waste management in Europe are intimately tied to the fulfillment of the prescribed targets of recycling and organic waste recovery in response to the requirements of European Directives. Challenges with characterizing and propagating uncertainty, and validating predictions permeate decision making. In order to retrieve the societal ramifications in decision making, this study integrates the analytic hierarchy process (AHP) and the technique for order performance by similarity to ideal solution (TOPSIS) for alternative screening and ranking to help decision makers in a Portuguese waste management system. To underscore the role of uncertainty in decision making for alternative ranking, a fuzzy interval multi-attribute decision analysis was carried out to aid in environmental policy decisions. While AHP was used to determine the essential weighting factors, screening and ranking was carried out by TOPSIS under uncertainty expressed by using an interval-valued fuzzy (IVF) method. Such an AHP-based IVF-TOPSIS approach driven by a set of weighting factors associated with the selected criteria has been proven useful for final ranking via an iterative procedure. The practical implementation was assessed by a case study in Setúbal Peninsula, Portugal for the selection of the best waste management practices under an uncertain environment, which is geared toward the target fulfillment in the future.

Keywords: waste management, uncertainty, multi-criteria decision making, life cycle assessment, sustainable decisions

5.2 INTRODUCTION

In Portugal, it is vital to ensure the full compliance with the targets required by the European Directives for solid waste management, such as the Packaging and Packaging Waste Directive 2004/12/EC (EC, 2004) and Landfill Directive 1999/31/EC (EC, 1999). Facing such challenges, Portugal needs to comply with packaging recycling targets before 2011. For organic waste, the targets established for 2009 and 2013 aiming to divert 50% and 65% of organic waste produced based on the 1985 generation basis, respectively have been delayed until 2013 and 2020. In addition to complying with Landfill and Packaging Directives, a new challenge arose from the New Waste Framework Directive 2008/98/EC (EC, 2008) in which it is imperative that waste management systems provided by Member States (MS) should take into account the general environmental protection principles with regard to precaution and sustainability, technical feasibility and economic viability, protection of resources as well as the overall environmental, human health, social, and economic impacts. In other words, waste management practices would be related to a series of trade-offs among different stakeholders having different objectives, making the operation more difficult to decision makers to reach a cordial decision. These trade-offs therefore involve considering relevant technical, economic, environmental, and social criteria that may be delineated by either quantitative or qualitative ways or both. Such challenges facing in the decision making arena have to be well addressed by a more scientifically credible approach to reach a sustainable solution.

Within this context, several sources of uncertainties can be addressed during waste management, which can affect the compliance of Directives' targets and the choice of the best waste management solution. The Directive targets are information (or innovation) to be spread through as national law or regulations. However, as the science and technology evolve over time we will never have perfect knowledge after all to ensure the right choice that makes implementation of the waste management practices an educational process. No matter which choice to be made, government agencies have to translate estimated changes into direct impacts on the affected entities and transform direct impact into changes in final demand for the waste management of those entities. These waste management entities mainly include Green Dot System, (i.e., it is named Sociedade Ponto Verde in Portugal) and relevant private sectors which will use those products such as recyclables, compost, electricity, etc. Some more changes can be induced by Pay-as-You-Throw (PAYT), which is a successful instrument but has not yet applied in Portugal.

The information diffusion process (see Fig. 5.1) for social education to promote the PAYT is hard to be characterized and such unattended consequences or complications may affect future election at both regional and local levels. In fact, the existence of an innovation is seen to cause uncertainty in the minds of potential adopter (Berlyne, 1962; Rogers, 1962; Nimmo, 1985) causing a lack of predictability. Challenges arise from that the general public receiving the information of PAYT has to respond as quickly as possible in a short period of time to be able to comply with the waste management targets. However, the predicted measures of how to achieve this goal with a soft computing model might be implemented in the field for evaluation. Yet metrics for validation and mathematical constructs that are useful for describing uncertainties in decision making as a whole are lacking.

Not only the uncertainty of the projections of PAYT implementation but also uncertainties from model parameters, type of models, inherent process uncertainties, uncertainties due to lack of knowledge about a specific process or processes, or uncertainties embedded in decision making could affect the final outcome. This necessitates creating a new spectrum of uncertainty quantification (UQ) that has been recognized as a critical element necessary for continued advancement in handling of waste management and societal sustainability.

Since information diffusion function is a fuzzy classifying function (Chongfu, 1997) fuzzy sets theory can be applied to cope with the complexity to some extent. This thrust covers a diversity of approaches to deal with uncertainty from different disciplines, reflecting differences in the underlying literature. The general framework of fuzzy reasoning allows handling much of this uncertainty, where fuzzy systems employ type-1 fuzzy sets, which represent uncertainty by numbers in the range [0, 1]. When something is uncertain, like a measurement, it is difficult to determine its exact value, and of course type-1 fuzzy sets make more sense than using sets (Zadeh, 1975a, b).

Because the nature of information, indicators and analyses used in waste management, a unique topology of uncertainties including unpredictability, structural uncertainty, and value/preference uncertainty in decision making, such as aleatoric and epistemic uncertainties, was investigated holistically in this study using the type-2 fuzzy sets (Karnik et al., 1999). According to Liang and Mendel (2000), applying type-2 fuzzy has been regarded as one way to increase the fuzziness of a relation and, according to Hisdal (1981), “increased fuzziness in a description means increased ability to handle inexact information in a locally correct manner”. Our disposition in handling such a decision analysis is to construct suitable interval-valued fuzzy (IVF) sets or type-2 fuzzy sets in this case study so as to characterize and quantify the uncertainty.

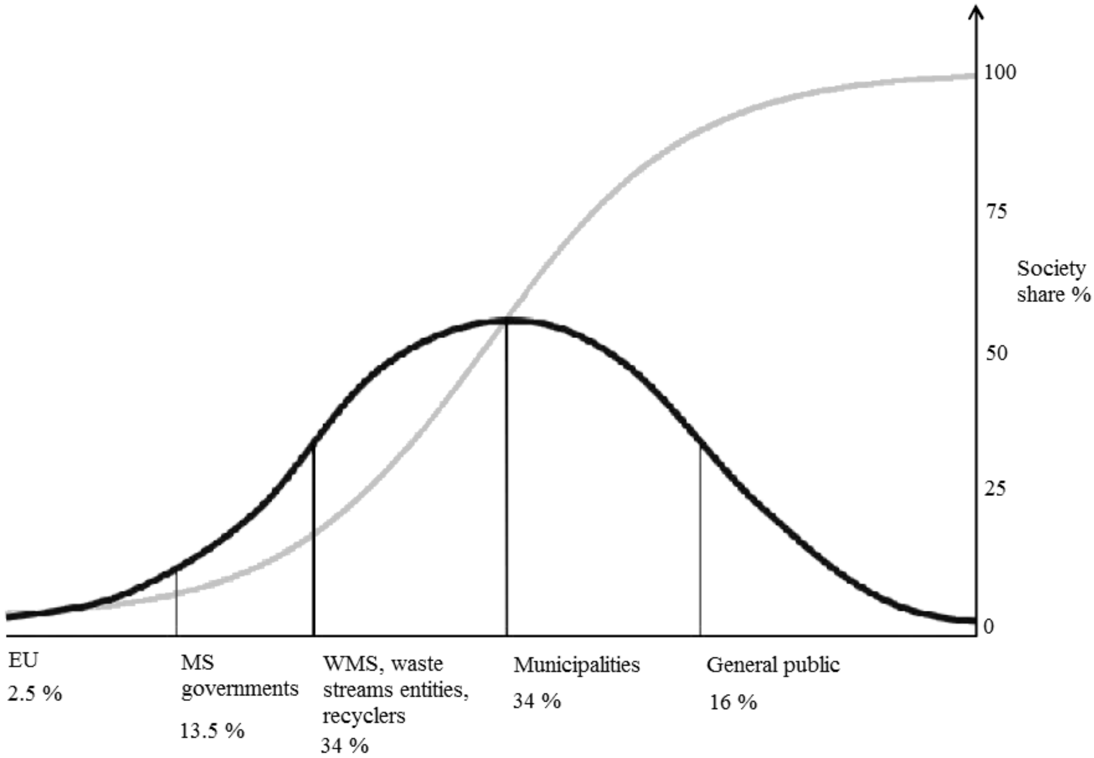

Fig. 5.1 Information diffusion through waste management stakeholders

According to Pohekar and Ramachandran (2004), in a multicriteria decision making (MCDM) process, a decision-maker is required to choose among quantifiable or non-quantifiable and multiple criteria. The objectives are usually conflicting and therefore, the solution is highly dependent on the preferences of the decision-maker leading to the generation of a compromised solution. The multi-attribute decision making (MADM) process that has been capable of helping decision making process by considering limited number of criteria, analyzing several alternatives (finite or infinite) is deemed a good framework. In the group decision making cases, different groups of decision-makers may be involved in such a MADM process. Each group brings along different criteria and points of view, which must be proposed within a mutual understanding framework.

The aim of this study is to integrate the analytic hierarchy process (AHP) and the technique for order performance by similarity to ideal solution (TOPSIS) to help decision makers in a Portuguese waste management system. To underscore the role of uncertainty in decision making for alternative ranking, a fuzzy interval multi-attribute decision analysis was carried out to aid in environmental policy decisions. While AHP was used to determine the essential weighting factors, screening and ranking was carried out by TOPSIS under uncertainty expressed by using an interval-valued fuzzy (IVF) method. It leads to the screening and ranking of 18 management alternatives to improve the sustainability of solid waste management in Setúbal region, Portugal. Through the use of a multi-attribute decision analysis under uncertainty, the chosen UQ methods help illustrate the sensitivity of various sources of uncertainty in decision making.

5.3 LITERATURE REVIEW

Several MADM methods have been applied in waste management, like ELECTRE (Roy, 1973, 1991), PROMETHEE (Brans et al., 1984) and GAIA (Brans and Mareshcal, 1994), AHP (Saaty, 1980), TOPSIS (Yoon and Hwang, 1985) and SAW. Table 5.1 summarizes the pros and cons of those MADM methodologies.

Table 5.1 Comparison of MADM methodologies applied to SWM

MADM methods	Description	Advantages	Disadvantages
SAW	• Value based method • Use of measurement of the utility of an alternative (Cheng et al., 2003)	• Easy to use and well understandable • Applicable when exact and total information is collected • Well-proven technique • Good performance when compared with more sophisticated methods (Chang and Yeh, 2001, Zanakis et al., 1998)	• Normalization is required to solve multidimensional problems
AHP	• Use of value based, compensatory, and pairwise comparison approach • Use of Hierarchical structure to present complex decision problem	• Applicable when exact and total information is collected • Decision problem can be fragmented into its smallest elements, making evidence of each criterion applied (Macharis et al., 2004) • Applicable for either single or multiple problems, since it incorporates qualitative and quantitative criteria. • Generation of inconsistency index to assure decision makers (Pohekar and Ramachandran, 2004)	• Due to aggregation, compensation between good scores on some criteria and bad scores on other criteria can occur (Macharis et al., 2004) • Implementation is quite inconvenient due to complexity (Tahriri et al., 2008) • Complex computation is required (Chou et al., 2008) • Time-consuming
TOPSIS	• Use of value based compensatory method • Measures the distances of the alternatives from the ideal solution • Selection of the one	• Easy to implement understandable principle • Applicable when exact and total information is collected • Consideration of both the positive and negative ideal solutions	• Normalization is required to solve multidimensional problems

	closest to the ideal solution (Cheng et al., 2002)	• Provision of a well-structured analytical framework for alternatives ranking (Geng et al., 2010) • Use of fuzzy number to deal with uncertainty problems	
ELECTRE	• Use of outranking method • Use of pairwise comparison, compensatory • Use of indirect method that ranks alternatives by means of pairwise comparison (Cheng et al., 2002)	• Applicable even when there is missing information • Applicable even when there are incomparable alternatives • Applicable even when incorporation of uncertainties is required • Applicable for quantitative and qualitative attributes	• Time consuming without using specific software due to complex computational procedure (Cheng et al., 2002) • May or may not reach the preferred alternative
PROMETHEE-GAIA	• Use of outranking method, pairwise comparison, and compensatory method • Use of positive and negative preference flows for each alternative in the valued outranking • generation of ranking in relation to decision weights	• Applicable even when there is missing information • Applicable even when simple and efficient information is needed (Queiruga et al., 2008)	•Time consuming without using specific software • When using many criteria, it becomes difficult for decision maker to obtain a clear view of the problem (Macharis et al., 2004)

The following review mainly includes part of the whole family of compensatory methods such as simple additive weight (SAW) (or simple weighted addition or weighted sum method), weighted product (WP), permutation method (PM), AHP, ELimination Et Choix Traduisant la REalité (ELECTRE), TOPSIS, Preference Ranking Organization Method for Enrichment Evaluation (PROMETHEE) – Geometrical Analysis for Interactive Aid (GAIA), interactive simple additive weighting (ISAW), LINear programming techniques for Multidimensional Analysis of Preference (LINMAP), linear assignment method (LAM), non-metric multi-dimensional scaling with ideal point and multi-attribute utility theory (MAUT), and more recent Simple Multi-Attribute Rating Technique (SMART). The chosen criteria are comprised of one technical criterion, seven environmental impact categories resulting from an independent life cycle assessment (LCA), three economic criteria, and three social criteria.

Developed by Roy (1973), outranking techniques do not assume that a single best alternative can be identified, and compares the performance of two or more alternatives at a time to identify the extent to which a preference for one over the other can be asserted (Linkov et al., 2006). The overall target of outranking models is the detailed description and structuring of the decision-making process rather than the determination of one optimal solution (Linkov et al., 2006). ELECTRE, PROMETHEE and GAIA are outranking methods, which may even assist the decision maker in cases of incomplete information (Queiruga et al., 2008). Besides strict preference and indifference, weak preference and incomparability of alternatives are also allowed (Brans and Mareschal, 1994). In addition, SAW, AHP and TOPSIS can be considered value measurement methods. The intention of such methods is to construct a means of a associating a real number with each alternative, in order to produce a preference order on the alternatives consistent with decision maker value judgments (Belton and Stewart, 2002). In other words, to the several criteria applied are given weights, which translate the importance of the criteria to decision makers.

With the existing features and potential applications with different purposes and domains, MADM methods may be applied for a variety of waste management issues. According to Cheng et al. (2002), waste management problems can be adequately addressed using SAW, TOPSIS and ELECTRE. SAW that has widely used in many fields is an easy tool for use by the decision makers (Cheng et al., 2002). TOPSIS have shown to be logic, and easily programmable in computational procedure (Önüt and Soner, 2008). However, both SAW and TOPSIS need the prior normalization to allow a correct integration of criteria and adequate comparison among alternatives.

Norese (2006) justified the application of ELECTRE III method that has a decision group support for waste management system based on: 1) a method which can prevent decision-maker from being asked questions that are too intricate, 2) it can be used in group decision-making (Hokkanen and Salminen, 1999), and 3) its multi-criteria model integrates different types of information in a transparent way and is easily elaborated and understood. In general, ELECTRE method is capable of handling discrete criteria of both a quantitative and a qualitative nature and provides complete ordering of alternatives (Rousis et al., 2008). PROMETHEE and GAIA also present success in waste management. PROMETHEE is a non-parametric outranking method for a finite set of alternatives (Brans et al., 1984). GAIA is a visualization method, which complements the PROMETHEE ranking method (Vego et al., 2008). PROMETHEE is also reclaimed to have simplicity, clarity, efficiency, and low information requirements (Queiruga et al., 2008). The same as the other outranking methods, PROMETHEE does not require merging criteria when they are too heterogeneous. AHP is an ideal method for ranking alternatives when multiple criteria and sub-criteria are present in the decision making process (Tahriri et al., 2008). The AHP decomposes decision problems into a hierarchical structure, and uses both qualitative and quantitative information to derive ratio scales between decision elements at each hierarchical level using pairwise comparisons

(Bello-Dambatta et al., 2009). AHP has been applied for waste management when linking with other MADM methods, such TOPSIS, since it can includes inconsistency index.

The choice for TOPSIS at this case study is justified by several reasons:

- It is a very applied method not only in solid waste management, but also at other areas, like economy and environment, manufacturing, tourist analysis, water resource management, transportation, project manager, inventory planning, and airline service evaluation, and cases mentioned by Dai et al. (2010);
- A simple method which can be developed in a spreadsheet (Kim et al., 1997);
- TOPSIS intends to find a compromise solution (Garcia-Cascales and Lamata, 2010), since it may identify the best solution that has the one more close to the positive ideal solution and farthest from negative ideal solution.
- TOPSIS compromise solution is quite similar to what happens during decision making process in waste management: most of the time, the best solution is not reached since the criteria are not in agreement, some must be maximized (like revenues from selling recyclables) and others minimized (like investment and operation costs); what can be a good option from cost perspective can bring considerable environmental and social issues (like a landfill near a habitation area).

For the weight criteria step, AHP is quite well-proven to be applied for this purpose, which ensures the application at this case. Also, due to the considerable number of criteria evaluated by decision makers, is important to measure the consistency of the evaluation, and AHP allows it.

5.4 DESCRIPTION OF THE STUDY AREA

Setúbal peninsula is located in the district of Setúbal with an area of 1,522 km^2 and has 714,589 inhabitants (AMARSUL, 2009). The area is divided in nine municipalities, as shown in Fig. 5.2. With a regionalization basis, the AMARSUL is the company owned by the local municipalities, which has been responsible for managing the MSW since 1997. The municipal solid waste generated in that area is manage by AMARSUL, a company owned by the local municipalities plus EGF company. The municipal solid waste (MSW) production per capita is 1.6 kg/day.

The SWM system is composed of nine recycling centers, two material recovery facilities (MRFs), two landfills, one transfer station, and one aerobic mechanical biological treatment (MBT). Nowadays, the SWM system in this area promotes separation of paper/cardboard, glass and light packaging (plastics, metals and composite packaging) waste by means of bring recycling systems. The remaining waste fractions (designated residual waste) can then be collected through door-to-door and/or bin collection schemes, which are directed for final disposal at landfills. In the case of Sesimbra municipality, the waste stream is first sent to the transfer station, and then followed by the final disposal at sanitary landfill. Yet the residual waste collected from Setúbal municipality is transported to an aerobic MBT plant where a stabilized residue can be produced as fertilizer to be applied as agriculture soil corrective materials.

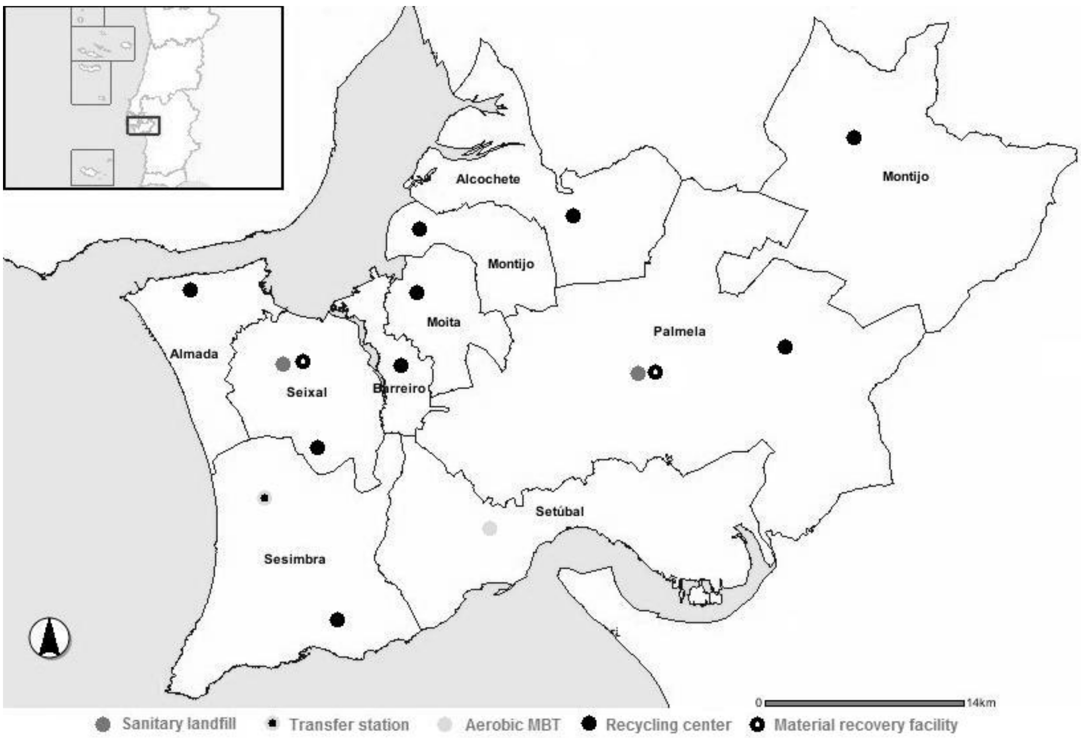

Fig. 5.2 The geographical location of Setúbal peninsula SWM system

Within this MSW system, there is a recent need to make some changes in order to comply with the Packaging and Packaging Waste Directive (EC, 2004) and Landfill Directive (EC, 1999). The National Plan for MSW (i.e., designated as PERSU II) decided to pursue the construction of several more MBT units. An anaerobic digestion (AD) MBT unit, with a mechanical treatment to separate recyclables and high calorific material to produce refuse derived fuel (RDF), is under planning. It is expected that this unit will work with two separate lines, in which one is related to the biodegradable municipal solid wastes (BMW) and the other is for the residual waste streams. The RDF may be combusted in an incinerator to generate electricity. The existing aerobic MBT plant will be maintained as usual. It is expected that both MRF plants with manual sorting will be replaced with two automatic sorting units. Fig. 5.3 illustrates the schematic of the predicted SWM system at Setúbal Peninsula.

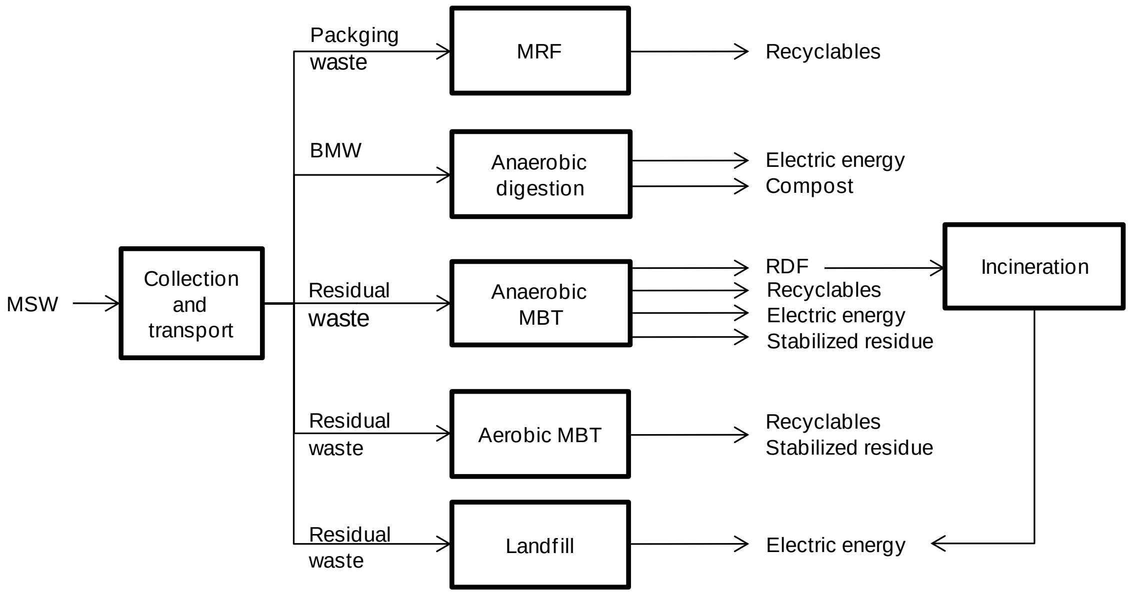

Fig. 5.3 The schematic of the predicted SWM system at Setúbal Peninsula

5.5 METHODOLOGY

5.5.1 Waste management alternatives

Based on such new regulations that AMARSUL must comply with, these 18 alternatives have been proposed and elaborated with respect to the preselected waste management technologies as shown in Table 5.2. The creation of Table 5.2 is based on the total amount of waste produced in 2008, which is 421,726 tonnes. Based on the average waste composition data region wide, the waste stream has 31.69% putrescibles, 14.13% paper and cardboard, 11.35% of plastics, 5.83% of glass, 4.14% of composites, 1.82% of metals, 2.07% of wood, 11.72% of textiles, 15.33% of fine particles, and 1.92% of others. All the alternatives can help comply with the actual need to reach the targets prescribed in the two new Directives. However, to reach both Directive targets simultaneously requires a behavioral change in Portuguese society. The two scenarios to analyze how targets were reached are:

- Baseline scenario: Targets may be reached without systematic involvement and evolution, meaning that it can be promoted by several external agents such as government, Green Dot Society (Sociedade Ponto Verde), and promotion campaigns that motivate a better environmental consciousness. The system may be financed by using water consumption tax for waste management to be included in the water billing system;

• Pay-As-You-Throw (PAYT) scenario: Targets can be reached by imposing an economic instrument – PAYT – to be implemented by various levels of MSW system managers.

Table 5.2 Alternatives proposed for the AMARSUL waste management system

Fraction (%) / Option	Alternatives							
	A0/A0*/A0'	A1/A1*	A2/A2*/A2'	A3/A3*	A4/A4*/A4'	A5/A5*/A5'	A6/A6*	Base
MRF	12.4	12.4	12.4	12.4	12.4	12.4	12.4	4.8
Anaerobic digestion BMW	5.4	0	0	13.3	0	7.5	28.7	0
Anaerobic digestion MBT	28.2	0	33.9	0	49.6	38.9	0	0
Aerobic MBT	13.2	49.7	15.8	32.6	0	0	0	13.8
Landfill with ER	40.8	37.9	37.9	41.7	38.0	41.2	58.9	81.4

* Alternatives considering RDF production plus incineration of high calorific fraction
' Alternatives not considering RDF production but incineration of high calorific fraction

Based on this system, Table 5.2 presents these 18 management alternatives for assessment plus the present situation designated as the base scenario. These alternatives include waste collection and separation of the three packaging materials through bin systems, which handle 12.4% of the content MSW in the study area. This MRF system is responsible for the compliance with the prescribed targets in Packaging Waste Directive. Alternative 0 refers to the predicted change that will take place in the Setúbal peninsula SWM system. The remaining alternatives were designed to examine some special options for complying with the Landfill Directive. For example, Alternative 1 emphasizes the inclusion of aerobic MBT; alternative 4 signifies the use of AD MBT; alternative 6 examines the specific case of using BMW anaerobic digestion line. In general, alternatives 0, 3, and 5 are options for a suite of intermediate processing. Separation of high calorific fraction of waste for RDF production was also considered in two options being defined for collecting the high calorific fraction from MRF refuse and from AD MBT separation.

5.5.2 Assessment criteria and criteria membership functions definition

The sustainability criteria may include different areas of waste management systems (Table 5.3).

Table 5.3 Evaluation criteria

Evaluation criteria	Description
Environmental criteria	
Abiotic depletion (AD)	Extraction of natural non-living resources. It is the difference between resources consumed during waste life cycle and resources consumption avoided from materials and energy substituted, in kg Sb eq.
Acidification (Ac)	Referent to acidifying pollutants emitted during waste life cycle. The calculation is the difference between impacts from waste life cycle less the avoided impact from substituted materials and energy, in kg SO_2 eq.
Eutrophication (Eut)	It is the consequence of high levels of macronutrients, such as nitrogen and phosphorous. It is the difference between eutrophication substances potential impact during waste life cycle and avoided impacts from substituted materials and energy, in kg PO_4^{3-} eq.
Global warming potential (GWP)	Represents the impact of greenhouse gases emissions on the radiative forcing of the atmosphere, inducing climate change. It is obtained from GHG potential impact from waste life cycle less the GHG impact from substituted materials, kg CO_2 eq.
Human toxicity (HT)	It is the difference from impacts on human health of toxic substances emitted less the avoided impacts from substituted materials and energy life cycle, in kg p-DCB eq.
Photochemical oxidation (PO)	Represents the formation of reactive chemical compounds, such as ozone, by action of sunlight on certain primary air pollutants. The calculation is provided from impact difference between waste life cycle and materials and energy substituted life cycles, in kg C_2H_2 eq.
Gross energy requirement (GER)	Amount of commercial energy that is required directly and indirectly by the process of making a good or service. It is the difference between energy consumed and energy produced, in kJ.
Economic criteria	
Investment (Inv)	Represents the amount to be expended to implement the alternative (in infrastructure, equipment, vehicles, land). In millions €.
Operational costs (OC)	Related to the amount to be expended during alternative operation, in material, electricity, maintenance, labor, and to financial costs like annuity. In €.
Operational revenues (OR)	The amount related to the profit obtained from selling products (energy, recyclables, compost) or with the avoidance of landfilling products (RDF, recyclables). In €.
Social criteria	
Economic efficiency (EE)	Represents the ratio between the waste fee applied to inhabitants and the net cost of MSW management system, in percentage.
Fee	It is the amount paid by population to finance MSW management system, in €/t.
Odor	It is referent to the impact of odors substances emitted during waste life cycle, in m^3.
Technical criteria	
Landfill space saving (LSS)	Ratio between waste not landfilled and total waste generated in a year, in percentage.

Seven stakeholders invited reflect versatile area of expertise and they are decision makers, technicians, environmentalists, inhabitants and experts who had been invited to respond to inquiries for the retrieval of weighting factors.

The criteria presented were selected considering the requirements of the new waste management philosophy brought by Thematic Strategy on the Prevention and Recycling of Waste (EC, 2008). That justifies the application of technical, environmental, economical and social aspects. For the technical aspect was considered the landfill space saving since this is the major aspect that waste managers may control, or else their non-renewable resource will be exhausted and more costs will be needed to construct a new landfill. Environmental criteria were obtained from LCA made for the alternatives elaborated in a companion study (Pires et al., 2010). The use of LCA is justified by New Waste Framework Directive 2008/98/EC (EC, 2008) in which suggested waste management plan should conform with the waste hierarchy from waste prevention to waste recycling and reuse, to incineration and energy recovery, and to landfill sequentially. However, when applying it, Member States shall take measures to encourage the options which deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy, and justifying life-cycle thinking on the overall impacts of the production and management of such waste (EC, 2008). LCA software used was UMBERTO 5.5 in this study to generate quantitative information. The environmental impact categories assessed were abiotic depletion (AD), acidification, eutrophication, global warming potential (GWP), human toxicity (HT) and photochemical oxidation (PO). Another important environmental criteria used was gross energy requirement (GER), also calculated for each alternative based on life cycle inventory data. Since Portugal is a country without producing fossil fuels, it is wise to look for waste management solutions in which net energy demand can be as low as possible. All the data used to perform the LCA may be seen in a companion study (Pires et al., 2010).

There are three criteria for addressing the economic aspects: investment, operational costs and operational revenues. Initial investment costs represent the amount needed to implement the waste management system. Concerning the use and operation of MSW facilities, to know cost and benefit during its life cycle is also relevant to choose which alternative is the best one. To calculate each costs/benefits category, several entities have been inquired to provide information and minimize gaps. They are summarized in Table 5.4.

Table 5.4 Summary of economic criteria calculation sources

Type of data	Sources of data
Infrastructures and equipments	
Collection and transport of MSW and recyclables	Local data from collection companies, Piedade and Aguiar (2010), EC (2001), EGF data
MRF unit	InCI (2010), Piedade and Aguiar (2010), AMARSUL (2009), EGF data
Aerobic MBT unit	AMARSUL (2009), Tsilemou and Panagiotakopoulos (2004, 2005), EGF data
Anaerobic MBT unit with/without BMW line unit	AMARSUL (2009), Tsilemou and Panagiotakopoulos (2004, 2005), EGF data
Landfill	AMARSUL (2009), Tsilemou and Panagiotakopoulos (2004, 2005), EGF data
Products	
Recyclables	SPV (2010)
Compost	AMARSUL (2009), EGF data
Electricity	MEI (2007)

Three social criteria were selected, including economic sufficiency, fees and odor. Odor was obtained from LCA. Since its impact can be considered a public health issue, odor issue was therefore classified as social criterion. Fees are the price paid by population to ensure the service of MSW disposal. Yet the Not-In-My-Bacyard (NIMBY) syndrome is a specific social impact that makes order issue linked with siting such new facilities. Fees are dependent with costs and revenues during a specific time framework. The importance of this criterion can be justified by the fact that fees are no-popular in Portugal and AMARSUL and municipalities would not favor this option. Without regard to the polluter pays principle (PPP), however, the MSW facilities have to be financed by municipalities with other sources. This justifies the use of economic sufficiency criteria. Economic sufficiency corresponds to the ratio between the amount paid by municipalities to AMARSUL to manage the waste stream and the total cost required. Overall, some criteria are self-explanatory, but the other may require further elaboration to avoid ambiguity and ensure sound understanding among the respondents.

All the criteria values for each alternative are presented in Table 5.5 and Table 5.6. It should be noticed that economic and social criteria for base scenario are overestimated due to the fact that recent biogas collection to produce electricity in the last few years has not received enough biogas. From environmental criteria point of view, the situation is the same, i.e., the base scenario is also overestimated with the best possible environmental performance.

Table 5.5 Evaluation matrix of alternative of waste management system in AMARSUL – environmental criteria

Alternatives	Environmental criteria						
	AD (kg Sb eq)	Acid. (kg SO_2 eq)	Eutrop. (kg PO_4^{3-} eq)	GWP (kg CO_2 eq)	HT (kg p-DCB eq)	PO (kg C_2H_2 eq)	GER (kJ)
A0	-2.2E+05	-1.9E+05	4.9E+03	3.0E+08	3.8E+06	7.7E+04	-1.3E+12
A0*	-5.4E+05	-3.9E+05	-1.0E+04	1.6E+08	3.1E+06	5.4E+04	-2.6E+12
A0’	-5.6E+05	-4.0E+05	-1.1E+04	1.4E+08	3.1E+06	4.6E+04	-2.8E+12
A1	-1.7E+05	-2.0E+05	-4.5E+03	2.5E+08	6.2E+06	6.2E+04	-1.4E+12
A1*	-1.9E+05	-2.1E+05	-5.5E+03	2.4E+08	6.2E+06	6.0E+04	-1.7E+12
A2	-1.9E+05	-2.1E+05	-6.1E+02	3.2E+08	4.8E+06	8.2E+04	-1.5E+12
A2*	-5.7E+05	-4.5E+05	-1.8E+04	1.5E+08	4.0E+06	4.6E+04	-2.7E+12
A2’	-5.9E+05	-4.6E+05	-1.9E+04	1.4E+08	4.0E+06	4.4E+04	-2.8E+12
A3	-2.1E+05	-1.3E+05	1.4E+04	2.3E+08	3.5E+06	6.0E+04	-1.7E+12
A3*	-2.3E+05	-1.6E+05	1.3E+04	2.2E+08	3.5E+06	5.8E+04	-1.8E+12
A4	-2.4E+05	-2.4E+05	-7.4E+02	3.1E+08	4.0E+06	8.0E+04	-1.8E+12
A4*	-7.5E+05	-5.5E+05	-2.4E+04	1.0E+08	3.0E+06	4.0E+04	-3.7E+12
A4’	-7.8E+05	-5.7E+05	-2.6E+04	8.7E+07	2.9E+06	3.6E+04	-3.8E+12
A5	-2.4E+05	-1.9E+05	8.3E+03	3.3E+08	3.0E+06	8.7E+04	-1.7E+12
A5*	-6.8E+05	-4.6E+05	-1.2E+04	1.2E+08	2.0E+06	4.4E+04	-3.4E+12
A5’	-7.0E+05	-4.7E+05	-1.3E+04	1.1E+08	2.0E+06	4.2E+04	-3.5E+12
A6	-2.5E+05	-5.7E+04	3.4E+04	2.3E+08	-2.7E+06	6.1E+04	-1.7E+12
A6*	-2.7E+05	-7.2E+04	3.3E+04	2.1E+08	-2.8E+06	5.9E+04	-1.8E+12
P.A0	-2.2E+05	-1.9E+05	4.9E+03	3.0E+08	3.8E+06	7.7E+04	-1.3E+12
P.A0*	-5.4E+05	-3.9E+05	-1.0E+04	1.6E+08	3.1E+06	5.4E+04	-2.6E+12
P.A0’	-5.6E+05	-4.0E+05	-1.1E+04	1.4E+08	3.1E+06	4.6E+04	-2.8E+12
P.A1	-1.7E+05	-2.0E+05	-4.5E+03	2.5E+08	6.2E+06	6.2E+04	-1.4E+12
P.A1*	-1.9E+05	-2.1E+05	-5.5E+03	2.4E+08	6.2E+06	6.0E+04	-1.7E+12
P.A2	-1.9E+05	-2.1E+05	-6.1E+02	3.2E+08	4.8E+06	8.2E+04	-1.5E+12
P.A2*	-5.7E+05	-4.5E+05	-1.8E+04	1.5E+08	4.0E+06	4.6E+04	-2.7E+12
P.A2’	-5.9E+05	-4.6E+05	-1.9E+04	1.4E+08	4.0E+06	4.4E+04	-2.8E+12
P.A3	-2.1E+05	-1.3E+05	1.4E+04	2.3E+08	3.5E+06	6.0E+04	-1.7E+12
P.A3*	-2.3E+05	-1.6E+05	1.3E+04	2.2E+08	3.5E+06	5.8E+04	-1.8E+12
P.A4	-2.4E+05	-2.4E+05	-7.4E+02	3.1E+08	4.0E+06	8.0E+04	-1.8E+12
P.A4*	-7.5E+05	-5.5E+05	-2.4E+04	1.0E+08	3.0E+06	4.0E+04	-3.7E+12
P.A4’	-7.8E+05	-5.7E+05	-2.6E+04	8.7E+07	2.9E+06	3.6E+04	-3.8E+12
P.A5	-2.4E+05	-1.9E+05	8.3E+03	3.3E+08	3.0E+06	8.7E+04	-1.7E+12
P.A5*	-6.8E+05	-4.6E+05	-1.2E+04	1.2E+08	2.0E+06	4.4E+04	-3.4E+12
P.A5’	-7.0E+05	-4.7E+05	-1.3E+04	1.1E+08	2.0E+06	4.2E+04	-3.5E+12
P.A6	-2.5E+05	-5.7E+04	3.4E+04	2.3E+08	-2.7E+06	6.1E+04	-1.7E+12
P.A6*	-2.7E+05	-7.2E+04	3.3E+04	2.1E+08	-2.8E+06	5.9E+04	-1.8E+12
Base	-1.4E+05	-3.7E+04	-2.7E+03	9.5E+08	5.8E+06	2.5E+05	-1.3E+12

Table 5.6 Evaluation matrix of alternative of waste management system in AMARSUL – economical, social and technical criteria

Criteria / Alternatives	Technical	Economic			Social		
	Landfill space saving (%)	Investment (10^6 €)	Operating. cost (€/y)	Operating. revenues (€/y)	Econ. efficiency (%)	Fee (€/t)	Odor (m^3)
A0	30	1.3E+02	4.0E+07	1.6E+07	55	69	1.5E+13
A0*	43	1.2E+02	3.9E+07	1.7E+07	58	66	1.2E+13
A0'	44	1.2E+02	3.9E+07	1.7E+07	58	65	1.2E+13
A1	42	1.0E+02	3.4E+07	1.5E+07	67	53	1.4E+13
A1*	43	1.0E+02	3.4E+07	1.5E+07	67	53	1.3E+13
A2	23	1.3E+02	4.0E+07	1.7E+07	56	64	1.6E+13
A2*	43	1.2E+02	3.9E+07	1.7E+07	57	63	1.2E+13
A2'	44	1.2E+02	3.9E+07	1.7E+07	58	62	1.2E+13
A3	43	1.1E+02	3.4E+07	1.7E+07	73	58	1.3E+13
A3*	44	1.1E+02	3.4E+07	1.7E+07	73	58	1.2E+13
A4	21	1.3E+02	3.9E+07	1.8E+07	61	58	1.6E+13
A4*	45	1.2E+02	3.8E+07	1.7E+07	63	57	1.2E+13
A4'	46	1.2E+02	3.8E+07	1.7E+07	64	56	1.1E+13
A5	25	1.3E+02	3.8E+07	1.8E+07	63	62	1.6E+13
A5*	44	1.2E+02	3.8E+07	1.8E+07	65	60	1.2E+13
A5'	45	1.2E+02	3.7E+07	1.8E+07	66	59	1.2E+13
A6	38	1.1E+02	3.3E+07	1.8E+07	87	61	1.2E+13
A6*	38	1.1E+02	3.3E+07	1.8E+07	87	61	1.1E+13
P.A0	30	1.3E+02	4.0E+07	1.6E+07	100	69	1.5E+13
P.A0*	43	1.2E+02	3.9E+07	1.7E+07	100	65	1.2E+13
P.A0'	44	1.3E+02	3.9E+07	1.7E+07	100	65	1.2E+13
P.A1	42	1.0E+02	3.4E+07	1.5E+07	100	53	1.4E+13
P.A1*	43	1.0E+02	3.4E+07	1.5E+07	100	53	1.3E+13
P.A2	23	1.3E+02	4.0E+07	1.7E+07	100	64	1.6E+13
P.A2*	43	1.2E+02	3.9E+07	1.7E+07	100	62	1.2E+13
P.A2'	44	1.2E+02	3.9E+07	1.7E+07	100	62	1.2E+13
P.A3	43	1.1E+02	3.4E+07	1.7E+07	100	58	1.3E+13
P.A3*	44	1.1E+02	3.4E+07	1.7E+07	100	57	1.2E+13
P.A4	21	1.3E+02	3.8E+07	1.8E+07	100	57	1.6E+13
P.A4*	45	1.2E+02	3.8E+07	1.7E+07	100	56	1.2E+13
P.A4'	46	1.2E+02	3.7E+07	1.7E+07	100	56	1.1E+13
P.A5	25	1.3E+02	3.8E+07	1.8E+07	100	62	1.6E+13
P.A5*	44	1.2E+02	3.8E+07	1.8E+07	100	59	1.2E+13
P.A5'	45	1.2E+02	3.7E+07	1.8E+07	100	59	1.2E+13

P.A6	38	1.1E+02	3.3E+07	1.8E+07	100	61	1.2E+13
P.A6*	38	1.1E+02	3.3E+07	1.8E+07	100	61	1.1E+13
Base	13	8.8E+01	2.3E+07	1.2E+07	107	31	3.8E+13

The crisp data applied have had to be translated into fuzzy membership functions. Concerning membership functions defined, in Fig. 5.4 are presented each membership for each criterion, with linguistic variables very good (VG), good (G), medium (M), poor (P) and very poor (VP). The memberships are triangular, since it used most often for representing fuzzy numbers (Ding and Laing, 2005). A triangular fuzzy number $\tilde{a}$ can be defined as a triplet (a_1, a_2, a_3), and such representation of membership functions can be realized by Fig. 5.4.

$$\mu_{\tilde{a}}(x) = \begin{cases} 0, x < a_1 \\ \dfrac{x - a_1}{a_2 - a_1}, a_1 \le x \le a_2 \\ \dfrac{x - a_3}{a_2 - a_3}, a_2 \le x a_3 \\ 0, x > a_3 \end{cases} \qquad (13)$$

It was intentional that membership does not reached membership=1, since the interval-valued triangular fuzzy number would be between [0, 1].

Relative degree of possible level of uncertainty was proposed to address the independent impact associated with different type of uncertainty in an iterative process. The iterative process can be stopped once we are sure that all types of uncertainty can be included. The degrees of uncertainty tested were based on the interval between linguistic classes, such as 5%, 50%, 100% and 125%.

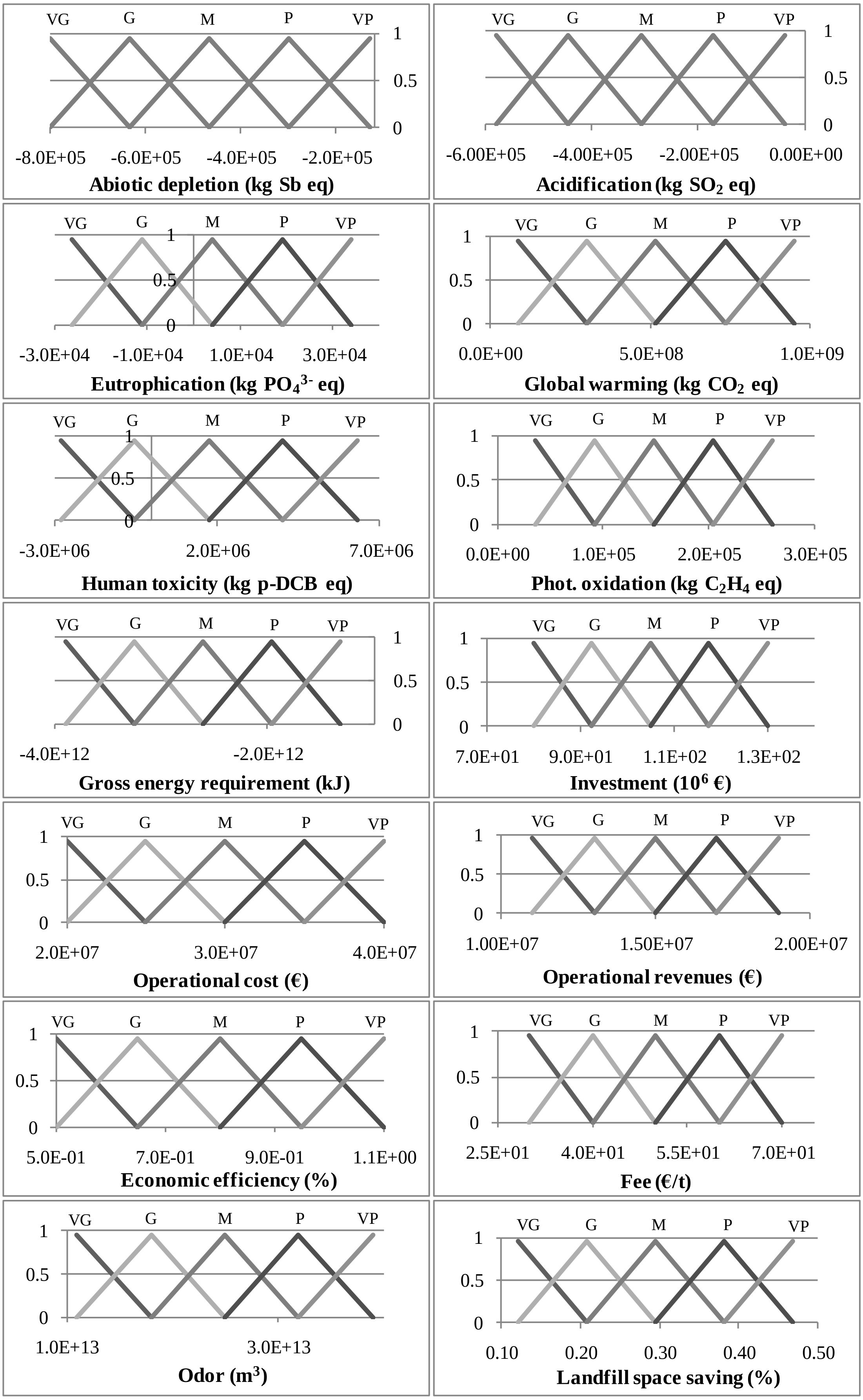

Fig. 5.4 Membership functions of the 14 criteria

5.5.3 AHP-based interval-valued fuzzy TOPSIS

The proposed method for the evaluation of waste treatment alternatives consists of two basic stages: (1) AHP computations to know criteria weights, and (2) evaluation of alternatives with IVF TOPSIS, where the best results may be expressed as an interval rather than an exact ideal solution. In the first stage, criteria defined for the assessment of the alternatives have been integrated in a decision hierarchy. AHP model is structured such that the objective, criteria, and waste management alternatives are on the first, second, and third level, respectively. A weighting factor associated with each of the criteria can be derived by AHP throughout a hierarchy process. Pairwise comparison matrices are formed to determine the criteria weights. Computing the geometric mean of the values obtained from individual evaluation can lead to the identification of the final pairwise comparison matrix. The weights of the criteria are calculated based on this final comparison matrix.

With the aid of the derived weighting factors, ranking of waste management alternatives can be determined by IVF TOPSIS method in the second stage. Based on the iterative process shown in Fig. 5.5, different intervals are defined with respect to the distance between linguistic variables that uniquely reflect the possible sources of uncertainty. In such an iterative procedure, it is expected that repeated calculations for testing several intervals that are intimately linked with the major sources of uncertainty. Beginning with an initial guess in regard to which range might be possible to reflect the fluctuations expressed by the interval, and might disturb the determination of a specific solution more close to the ideal solution. A schematic diagram of the proposed method can be seen in Fig. 5.5. Iteration might be terminated when all types of uncertainty can be fully taken into account.

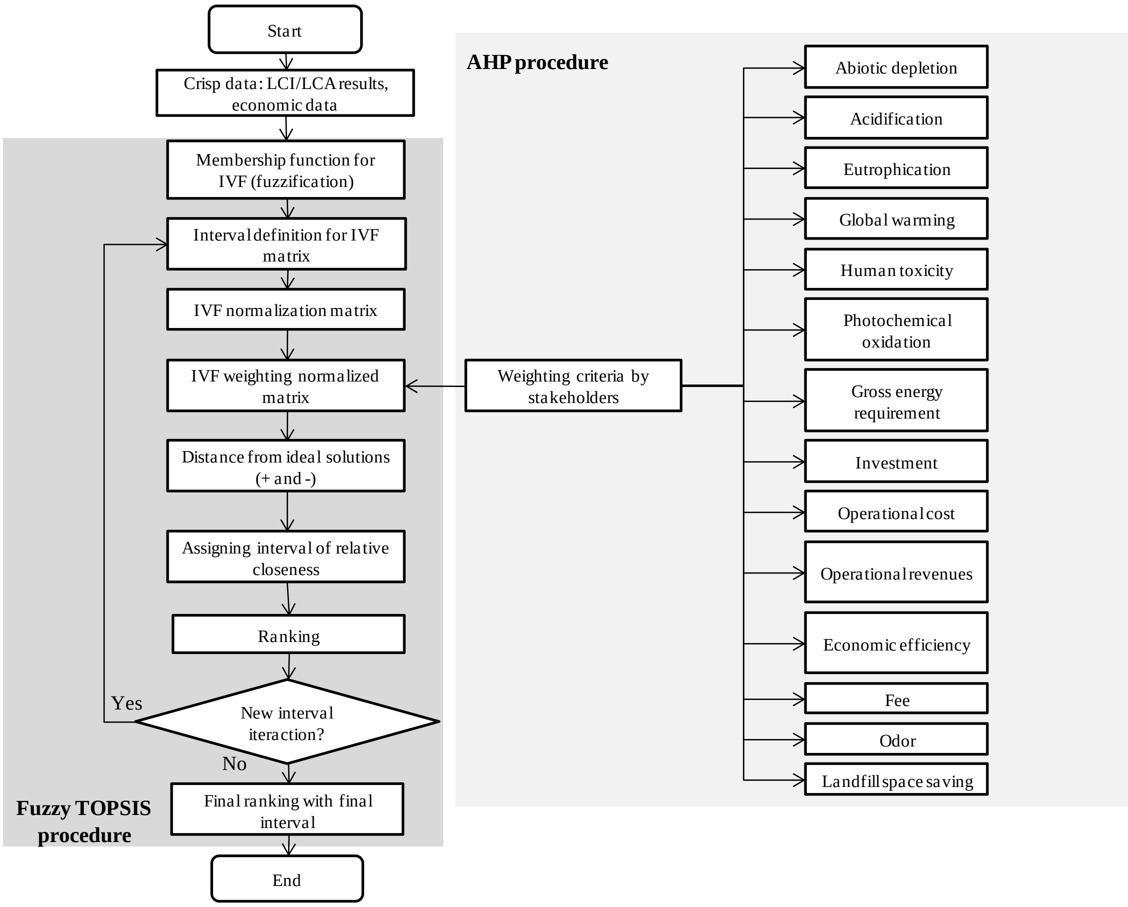

Fig. 5.5 Flowchart of the proposed method for waste management alternatives

5.5.3.1 Analytical hierarchy process

According to Saaty and Vargas (2001), the AHP is a basic approach in decision making. It is designed to cope with both the rational and the intuitive sources of uncertainty to select the best out of a number of alternatives evaluated with respect to several criteria. In this process, the decision maker carries out simple pairwise comparisons which are then used to develop overall priorities for ranking the alternatives. The AHP allows for inconsistency in the judgments and provides a means to improve consistency (Saaty and Vargas, 2001).

The AHP is developed based on the following five steps (Saaty, 1980):

- Define the problem, and determine the objective;
- Development of the hierarchy from the top (the objective from a general view point) through the intermediate levels (attributes and sub-attributes on which subsequent levels depends) to the lowest level (the list of alternatives);
- Employ a simple pair-wised comparison matrices for each of the lower levels;
- Undertake a consistency test; and

- Estimate relative weights of the components of each level.

For designing the pair-wised comparison matrices, the decision hierarchies can be organized based on a suite of criteria listed in the right portion of Fig. 5.5. The top level in such an AHP analysis is the selected goals, followed by some sustainable criteria. The goals of concern include environment, economic, social and technical aspects. The third level is comprised the break-down criteria expanded from these sustainable criteria. The relative importance of the criteria is rated by the nine-point scale proposed by Saaty (1980), as shown in Table 5.7.

Table 5.7 The AHP pairwise comparison scale (Saaty, 1980)

Intensity of weight	Definition	Explanation
1	Equal importance	Contribute equally to the objectives
3	Weak/moderate importance of one over another	Slightly favor one objective over another
5	Essential or strong importance	Strongly favor one objective over another
7	Very strong or demonstrated importance	An objective is favored very strongly over another; dominance demonstrated in practice
9	Absolute importance	Evidence favoring one objective over another is of the highest possible order of affirmation
2,4,6,8	Intermediate values between the two adjacent scale values	Used to represent compromise between the priorities listed above

The AHP decomposes decision problems into a hierarchical structure, through the pairwise comparison. Such comparisons are recorded in a comparative matrix A, which must be both transitive such that if, $i > j$ and $j > k$ then $i > k$, where *i*, *j* and *k* are alternatives; for all $j < k < i$ and reciprocal, $a_{ij} = \frac{1}{a_{\mu}}$. Priorities are then computed from the comparison matrix by normalizing each column of the matrix, to derive the normalized primary right eigen vector, the priority vector, by $A \times w = \lambda_{\max} \times w$, where *A* is the comparison matrix; *w* is the principal eigen vector; $\lambda_{\max}$ is the maximal eigen value of matrix A (Saaty, 2004).

The AHP provides a method of calculating a decision-makers inconsistency, the consistency index (CI) which is used to determine whether decisions violate the transitivity rule, and by how much (Bello-Dambatta et al., 2009). CI is defined by $CI = \frac{\lambda_{\max} - n}{n-1}$, where λ_{max} as above, n is dimension. Based on CI is possible to calculate consistency ratio, $CR = \frac{CI}{RI}$, where RI is the random index, being, at this case, for matrix order 14, RI is 1.57 (Lin and Yang, 1996).

The number 0.1 is the accepted upper limit for CR. If the final consistency ratio exceeds this value, the evaluation procedure has to be repeated to improve consistency. The measurement of consistency can be used to evaluate the consistency of decision makers as well as the consistency of overall hierarchy (Wang & Yang, 2007).

5.5.3.2 Interval-valued fuzzy TOPSIS

TOPSIS developed by Yoon and Hwang (1985) based upon the concept that the chosen alternative should have the shortest distance from the ideal solution and the farthest from the negative-ideal solution. A utility value *D(i)* for each alternative *i* is obtained by calculating the relative distance for *i* to the ideal solution, which can be described as follows (Jahanshahloo et al., 2006):

Step 1. Calculate the normalized decision matrix. The normalized value n_{ij} is calculated as

$$n_{ij} = x_{ij} \Bigg/ \sqrt{\sum_{j=1}^{m} x_{ij}^2}\,, \qquad j = 1,\ldots,m,\ \ i = 1,\ldots,n. \tag{1}$$

Step 2. Calculate the weighted normalized decision matrix. The weighted normalized value *vij* is calculated as

$$v_{ij} = w_i n_{ij}\,, \qquad j = 1,\ldots,m,\ i = 1,\ldots,n, \tag{2}$$

where *wi* is the weight of the *i*th attribute or criterion, and $\sum_{i=1}^{n} w_i = 1$.

Step 3. Determine the positive ideal and negative ideal solution.

$$\begin{aligned} A^+ &= \left\{v_1^+,\ldots,v_n^+\right\} = \left\{\left(\max_j v_{ij} \middle| i \in I\right), \left(\min_j v_{ij} \middle| i \in J\right)\right\}, \\ A^- &= \left\{v_1^-,\ldots,v_n^-\right\} = \left\{\left(\min_j v_{ij} \middle| i \in I\right), \left(\max_j v_{ij} \middle| i \in J\right)\right\}, \end{aligned} \tag{3}$$

where *I* is associated with benefit criteria, and *J* is associated with cost criteria.

Step 4. Calculate the separation measures, using the *n*-dimensional Euclidean distance. The separation of each alternative from the ideal solution and for the negative ideal solution are given as, respectively,

$$d_j^+ = \left\{ \sum_{i=1}^{n} \left(v_{ij} - v_i^+ \right)^2 \right\}^{\frac{1}{2}}, \qquad j = 1,\ldots,m. \tag{4}$$
$$d_j^- = \left\{ \sum_{i=1}^{n} \left(v_{ij} - v_i^- \right)^2 \right\}^{\frac{1}{2}}$$

Step 5. Calculate the relative closeness to the ideal solution. The relative closeness of the alternative A_j with respect to A^+ is defined as

$$R_j = d_j^- \Big/ \left(d_j^+ + d_j^- \right), \qquad j = 1,\ldots,m. \tag{5}$$

Since $d_j^- \geq 0$ and $d_j^+ \geq 0$, then $R_j \in [0,1]$.

Step 6. Rank the preference decreasing order.

To apply interval-valued fuzzy numbers in TOPSIS, is necessary to explain a little more in what consists IVF. According to Türksen (2006), in IVF, upper and lower bounds of membership are identified and the spread of membership, distribution is ignored with the assumption that membership values between upper and lower values are uniformly distributed or scattered with membership value of "1" on the $\mu(\mu(.))$ axis. Thus, the upper and lower bounds of interval-valued type 2 (or IVF) fuzziness specify the range of uncertainty about the membership values. A representation of a triangular IVF graphically is in Figure 4, being an IVF defined as $\tilde{x} = \left[\left(x_1, x_1' \right); x_2; \left(x_3', x_3 \right) \right]$

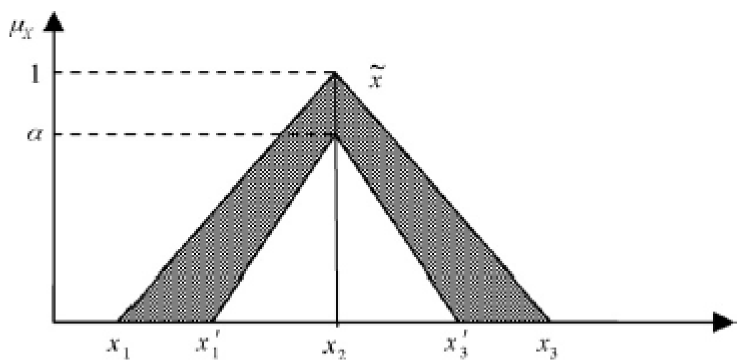

Figure 4. Interval-valued triangular fuzzy number (Ashtiani et al., 2009)

The developed AHP based IVF TOPSIS method has been based on the proposed method developed by Ashtiani et al. (2009).

Step 1. Given $\tilde{x} = [(a_{ij}, a'_{ij}); b_{ij}; (c'_{ij}, c_{ij})]$, the normalized performance rating as an extension of Chen (2000) can be calculated as:

$$\tilde{r}_{ij} = \left[\left(\frac{a_{ij}}{c_j^+}, \frac{a'_{ij}}{c_j^+}\right); \frac{b_{ij}}{c_j^+}; \left(\frac{c'_{ij}}{c_j^+}, \frac{c_{ij}}{c_j^+}\right)\right],\ i = 1,\ldots, n,\ j \in \Omega_b \tag{6}$$

$$\tilde{r}_{ij} = \left[\left(\frac{a_j^-}{c'_{ij}}, \frac{a_j^-}{c_{ij}}\right); \frac{a_j^-}{b_{ij}}; \left(\frac{a_j^-}{a_{ij}}, \frac{a_j^-}{a'_{ij}}\right)\right],\ i = 1,\ldots, n,\ j \in \Omega_c$$

$C_j^+ = \text{Max}\ c_{ij},\ j \in \Omega_b$

$a_j^- = \text{Min}\ a'_{ij},\ j \in \Omega_c$

Hence, the normalized matrix $\tilde{R} = [\tilde{r}_{ij}]_{n\times m}$ can be obtained.

Step 2. By considering the different importance of each criterion obtained from AHP method, the weighted normalized fuzzy decision matrix is constructed as: $\tilde{V} = [\tilde{v}_{ij}]_{n\times m}$, where $\tilde{v}_{ij} = [(g_{ij}, g'_{ij}); h_{ij}; (l'_{ij}, l_{ij})]$

Step 3. Ideal and negative ideal solution can be defined as:

$$A^+ = [(1,1); 1; (1,1)],\ j \in \Omega_b \tag{7}$$

$$A^- = [(0,0); 0; (0,0)],\ j \in \Omega_c$$

Step 4. Normalized Euclidean distance can be calculated:

$$D^-(\tilde{N},\tilde{M}) = \sqrt{\frac{1}{3}\sum_{i=1}^{3}\left[\left(N_{x_i}^- - M_{y_i}^-\right)^2\right]}$$

$$D^+(\tilde{N},\tilde{M}) = \sqrt{\frac{1}{3}\sum_{i=1}^{3}\left[\left(N_{x_i}^+ - M_{y_i}^+\right)^2\right]} \tag{8}$$

where $D^-(\tilde{N},\tilde{M})$ and $D^+(\tilde{N},\tilde{M})$ the primary and secondary distant measure, respectively. Thereby, distance of each alternative from the ideal alternative $\left[D_{i1}^+, D_{i2}^+\right]$ can be currently calculated, where:

$$D_{i1}^{+} = \sum_{j=1}^{m} \sqrt{\frac{1}{3}\left[(g_{ij}-1)^2 + (h_{ij}-1)^2 + (l_{ij}-1)^2\right]} \tag{9}$$

$$D_{i2}^{+} = \sum_{j=1}^{m} \sqrt{\frac{1}{3}\left[(g'_{ij}-1)^2 + (h_{ij}-1)^2 + (l'_{ij}-1)^2\right]}$$

Similarly, the separation from the negative ideal solution is given by $\left[D_{i1}^{+}, D_{i2}^{+}\right]$, where:

$$D_{i1}^{-} = \sum_{j=1}^{m} \sqrt{\frac{1}{3}\left[(g_{ij}-0)^2 + (h_{ij}-0)^2 + (l_{ij}-0)^2\right]} \tag{10}$$

$$D_{i2}^{-} = \sum_{j=1}^{m} \sqrt{\frac{1}{3}\left[(g'_{ij}-0)^2 + (h_{ij}-0)^2 + (l'_{ij}-0)^2\right]}$$

Those equations are employed to determine the distance from the ideal and negative ideal alternatives in interval values.

Step 5. The relative closeness can be calculated as follows:

$$RC_1 = \frac{D_{i2}^{-}}{D_{i2}^{+} + D_{i2}^{-}}, \quad RC_2 = \frac{D_{i1}^{-}}{D_{i1}^{+} + D_{i1}^{-}} \tag{11}$$

The final values of RC_i^{*} are determined as:

$$RC_i^{*} = \frac{RC_1 + RC_2}{2} \tag{12}$$

5.6 RESULTS AND DISCUSSION

5.6.1 Calculate criteria weights

The weights of the criteria to be used in evaluation process are calculated by using AHP method. In this phase, the stakeholders selected are given the task of forming individual pairwise comparison matrix by using the scale in Table 5.7. Geometric means of these values are found to obtain the pairwise comparison matrix on which there is a consensus, like is shown in Table 5.8.

Table 5.8 The pairwise comparison matrix for criteria

	AD	Ac	Eut	GWP	HT	PO	GER	Inv	OC	OR	EE	Fee	Odor	LSS
AD	1.00	0.96	2.94	2.26	0.54	0.27	0.43	1.24	1.78	1.18	0.71	0.64	0.57	0.63
Ac	1.04	1.00	2.35	2.14	0.47	0.21	0.37	0.77	0.79	0.63	0.42	0.40	0.33	0.58
Eut	0.34	0.43	1.00	1.55	0.27	0.22	0.40	0.63	0.74	0.78	0.42	0.39	0.54	0.54
GWP	0.44	0.47	0.64	1.00	0.28	0.19	0.32	0.47	0.48	0.43	0.42	0.38	0.37	0.50
HT	1.85	2.14	3.65	3.55	1.00	0.64	1.76	1.38	2.52	2.14	2.00	2.24	1.83	1.62
PO	3.77	4.74	3.73	5.38	1.56	1.00	3.09	4.34	4.68	4.68	4.88	4.27	5.10	3.29
GER	2.33	2.70	3.36	3.14	0.57	0.32	1.00	1.70	1.83	1.85	1.76	1.92	1.73	2.14
Inv	0.81	1.30	1.70	2.12	0.72	0.23	0.59	1.00	1.41	1.47	1.41	0.89	0.71	0.79
OC	0.56	1.26	1.26	2.08	0.40	0.21	0.55	0.71	1.00	0.42	0.33	0.28	0.35	0.77
OR	0.85	1.59	1.28	2.33	0.47	0.21	0.54	0.68	2.38	1.00	0.64	0.33	0.34	0.71
EE	1.40	2.40	2.38	2.38	0.50	0.20	0.57	0.71	3.03	1.57	1.00	0.55	0.40	0.52
Fee	1.57	2.52	2.54	2.64	0.45	0.23	0.52	1.12	3.53	2.99	1.82	1.00	1.51	0.73
Odor	1.74	3.03	1.85	2.69	0.55	0.19	0.58	1.41	2.82	2.90	2.49	0.66	1.00	1.16
LSS	1.59	1.71	1.85	2.00	0.62	0.30	0.47	1.26	1.41	1.40	1.92	1.37	0.86	1.00

Note: AD – abiotic depletion; Ac – acidification; Eut – eutrophication; GWP – global warming potential; HT – human toxicity; PO – photochemical oxidation; GER – gross energy requirement; Inv – investment; OC – operational costs; OR – operational revenues; EE – economic efficiency; LSS – landfill space saving

The results obtained from the computations based on the pairwise comparison matrix provided in Table 5.8, are presented in Table 5.9.

Table 5.9 Results obtained with AHP

Criteria	Weights (w)	Criteria	Weights (w)	λ_{max}, CI, RI	CR
AD	0.040	Inv	0.036	λ_{max} = 14.628	CR = 0.03
Ac	0.031	OC	0.046	CI = 0.05	
Eut	0.025	OR	0.058	RI = 1.57	
GWP	0.111	EE	0.079		
HT	0.220	Fee	0.080		
PO	0.098	Odor	0.067		
GER	0.056	LSS	0.053		

Note: AD – abiotic depletion; Ac – acidification; Eut – eutrophication; GWP – global warming potential; HT – human toxicity; PO – photochemical oxidation; GER – gross energy requirement; Inv – investment; OC – operational costs; OR – operational revenues; EE – economic efficiency; LSS – landfill space saving

The HT and GWP are the two more important criteria in the selection of waste management solution for AMARSUL system. Consistency ratio of the pairwise comparison matrix is calculated as $0.03 < 0.1$. So the weights are shown to be consistent and they are used in the selection process.

5.6.2 Evaluation of alternatives and determine the final rank

The initial selection of the relevant fuzzy interval membership functions would be versatile. With the aid of these stakeholders, we tend to select similar intervals to address the equal impact of the information diffusion across all domains with no time lag. When the iteration can be made possible by simply reducing the interval gradually after the initial selection across all fuzzy variables, it is now possible to observe that only when the interval is 1.25 times bigger than the linguistic classes is exactly when there is a change in ranking, like is shown in Table 5.10. The best solution for the AMARSUL system would be the implementation of anaerobic digestion MBT and anaerobic digestion plant of biodegradable municipal waste followed by the RDF production, which should be managed by the PAYT program. As a consequence, A5 is the best option that is related to PAYT program.

Table 5.10 Iteration procedure and respective rankings

Uncertainty tested and rankings							
5%	Rank	50%	Rank	100%	Rank	125%	Rank
0.0578	P.A5*	0.0473	P.A5*	0.0437	P.A5*	0.0464	P.A2
0.0577	P.A5'	0.0472	P.A5'	0.0436	P.A5'	0.0446	A2
0.0570	A5*	0.0465	A5*	0.0430	A5*	0.0444	P.A4
0.0568	A5'	0.0463	A5'	0.0428	A5'	0.0434	P.A5*
0.0564	P.A2	0.0425	P.A4*	0.0407	P.A1	0.0433	P.A5'
0.0541	P.A4	0.0424	P.A4'	0.0400	P.A2	0.0432	P.A0
0.0541	A2	0.0420	P.A1	0.0400	P.A4*	0.0428	A4
0.0525	P.A1	0.0413	P.A2*	0.0399	A1	0.0427	P.A1
0.0523	A4	0.0413	P.A2'	0.0399	P.A4'	0.0427	A5*
0.0519	P.A4*	0.0412	A1	0.0395	P.A1*	0.0425	A5'
0.0517	P.A4'	0.0408	P.A0*	0.0389	P.A2*	0.0420	A0
0.0516	A1	0.0407	P.A1*	0.0389	P.A2'	0.0420	A1
0.0512	P.A1*	0.0404	A4*	0.0387	A1*	0.0419	P.A5
0.0505	P.A2*	0.0404	A4'	0.0385	P.A0*	0.0416	P.A1*
0.0505	P.A2'	0.0404	P.A2	0.0381	A4*	0.0409	A1*
0.0503	A1*	0.0401	P.A0'	0.0381	A4'	0.0404	A5
0.0498	P.A0*	0.0399	A1*	0.0381	A2	0.0401	P.A4*
0.0496	A4*	0.0395	A2*	0.0379	P.A4	0.0400	P.A4'
0.0496	A4'	0.0391	A0*	0.0379	P.A0'	0.0391	P.A2*
0.0491	P.A0'	0.0387	A2'	0.0372	A2*	0.0391	P.A2'
0.0486	A2*	0.0384	A0'	0.0369	A0*	0.0388	P.A0*
0.0485	P.A0	0.0382	P.A4	0.0366	A2'	0.0383	A4*
0.0480	A0*	0.0382	A2	0.0364	A4	0.0383	A4'

Uncertainty tested and rankings							
0.0477	P.A5	0.0365	A4	0.0363	A0'	0.0382	P.A0'
0.0477	A2'	0.0359	P.A3	0.0360	P.A0	0.0381	P.A3
0.0472	A0'	0.0353	P.A3*	0.0355	P.A3	0.0376	P.A6*
0.0470	A0	0.0352	P.A6*	0.0350	P.A3*	0.0375	P.A3*
0.0459	A5	0.0349	A6*	0.0349	P.A6*	0.0374	A2*
0.0451	P.A3	0.0345	A3	0.0348	P.A5	0.0373	A6*
0.0446	P.A3*	0.0344	Base	0.0347	A0	0.0372	A0*
0.0443	P.A6*	0.0342	P.A6	0.0346	A6*	0.0369	A3
0.0440	A6*	0.0340	A3*	0.0343	A3	0.0369	A2'
0.0437	A3	0.0339	A6	0.0341	P.A6	0.0368	P.A6
0.0433	P.A6	0.0336	P.A0	0.0338	A6	0.0366	A0'
0.0431	A3*	0.0335	P.A5	0.0337	A3*	0.0365	A6
0.0429	A6	0.0322	A0	0.0332	A5	0.0363	A3*
0.0420	Base	0.0318	A5	0.0319	Base	0.0321	Base

5.6.3 AHP effects in decision making

In the case in which criteria weights are equally important and 5% uncertainty is assumed with respect to the same degree of information diffusion among stakeholders in this practice, decision analysis would turn out to be different and the options obtained in this situation are presented in Table 5.11. The same best alternative can be reached. The change of weights would signify the higher importance of economic consideration though. In this context, it is verified that economic group that is one of the four groups, including environmental, economic, social and technical group cannot alter the final option dramatically.

Table 5.11 Iteration procedure and respective rankings with and without weighted criteria

Uncertainty tested and rankings			
5%, weighted criteria	Rank	5%, without weighted criteria	Rank
0.0578	P.A5*	0.0578	P.A5*
0.0577	P.A5'	0.0575	P.A5'
0.0570	A5*	0.0567	A5*
0.0568	A5'	0.0564	A5'
0.0564	P.A2	0.0531	P.A4*
0.0541	P.A4	0.0528	P.A4'
0.0541	A2	0.0517	P.A1
0.0525	P.A1	0.0508	P.A2*
0.0523	A4	0.0508	P.A2'
0.0519	P.A4*	0.0506	A1

Uncertainty tested and rankings			
0.0517	P.A4'	0.0506	P.A0*
0.0516	A1	0.0505	P.A1*
0.0512	P.A1*	0.0497	A4*
0.0505	P.A2*	0.0497	A4'
0.0505	P.A2'	0.0495	P.A2
0.0503	A1*	0.0495	P.A0'
0.0498	P.A0*	0.0493	A1*
0.0496	A4*	0.0485	A2*
0.0496	A4'	0.0484	A0*
0.0491	P.A0'	0.0479	A2'
0.0486	A2*	0.0477	A0'
0.0485	P.A0	0.0473	P.A4
0.0480	A0*	0.0471	A2
0.0477	P.A5	0.0463	A4
0.0477	A2'	0.0461	P.A3
0.0472	A0'	0.0459	P.A3*
0.0470	A0	0.0455	P.A6*
0.0459	A5	0.0453	A6*
0.0451	P.A3	0.0452	A3
0.0446	P.A3*	0.0449	Base
0.0443	P.A6*	0.0448	P.A6
0.0440	A6*	0.0444	A3*
0.0437	A3	0.0440	A6
0.0433	P.A6	0.0432	P.A0
0.0431	A3*	0.0430	P.A5
0.0429	A6	0.0429	A0
0.0420	Base	0.0422	A5

For this particular case study, the stakeholders have called on based on their considerable professional background in environmental, economic, social and technical criteria. Yet the current AHP procedure could only account for seven agents, namely one from each group at minimum. This is deemed insufficient to represent the possible opinions from all stakeholders involved in decision making. The response collected from our stakeholders involved highlight that they felt quite difficult to compare an environmental impact against an economic criterion, since both of which have so much difference in nature, making final scoring difficult. This should be taken into consideration when incorporating the LCA results into the decision making process since some of the stakeholders probably cannot comprehend the implications of LCA and consequences of the environmental impacts. A possible way to minimize such influence could be the enhancement of communication during the implementation of the AHP.

5.6.4 Interval effects in fuzzy interval scheme

Varying degree of uncertainty may be assumed with respect to the different degree of information diffusion among stakeholders in this section to signify the sensitivity of fuzzy classes. Changes can be reported based on the most sensitive retardation of information diffusion so that the final option may be altered. A5 is no longer the best option and the best one becomes alternative A2, which is based on the implementation of anaerobic and aerobic MBTs units, including PAYT program.

5.7 CONCLUSIONS

The selection of waste management strategies to improve sustainability in the AMARSUL system is a challenging issue when reaching the targets at national level set by the European Directives. There are many alternatives that can be geared toward reaching such goals, but how the policy information can be propagated from government to all stakeholders of the general public and how the stakeholders respond to this urgency would be uncertain. If the new measures like PAYT have to be in place associated with 18 alternatives into decision making process to promote the odds of success, a scientific methodology (i.e., UQ) to assess waste management alternatives should be available. Through the use of interval-valued triangular fuzzy numbers to express linguistic uncertainty embedded in the decision process, a MADM model in this study provides us with an objective screening and ranking procedure with respect to environmental, economic, technical and social criteria partially supported by a stand-alone LCA. Both AHP and TOPSIS are seamlessly integrated and applied to retrieve criteria weights for alternative selection. Whereas IVF TOPSIS is employed to determine the priorities of the alternatives, the weights derived from AHP reveal the impacts in a societal context in decision-making.

The model has been proven adequate in this case study, since the uncertainty embraced during the decision analysis ensures that a variety of sources of uncertainty can be collectively characterized by the IVF scheme. Final success of this thrust in the AMARSUL system is tied to the handling of recycling programs, and the selection PAYT, and the choice of the best solution. Overall, future work may be directed to improve the retrieval of the weights through different methods other than AHP to further address risk associated with uncertainty simultaneously.

5.8 ACKNOWLEDGMENTS

The authors are grateful for the support from Seixal, Palmela, Almada, Montijo and Moita municipalities, Empresa Geral do Fomento company, Extruplás company, and also for the national associations Plastval, Fileira Metal, Recipac. We acknowledge all the data and reports cited and used in this study.

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CHAPTER VI. GLOBAL CONCLUSIONS

6 GLOBAL CONCLUSIONS

The main objective of this thesis was to develop a system analysis model which could help decision makers to manage MSW management system in a sustainable way. The systems analysis techniques applied have been LCA and MCDM to the AMARSUL waste management system, located in Setúbal peninsula, Portugal.

As a global conclusion, the systems analysis model developed has presented potential to be applied in waste management systems, since it is capable of include all sustainable aspects, include all stakeholders potentially involved and belonging to SWM system, can model the waste life cycle, being a easy understandable methodology of helping in decision making process. Due to its holistic view, the model can be used to justify decision and the reasoned implementation of ISWM. The best alternatives reached have been solutions with anaerobic digestion MBT and BMW, with energetic recovery of RDF in incineration plant, including PAYT scheme to reach packaging waste Directive. The results reached during LCA have pointed out other solution, the massive anaerobic digestion MBT including energetic recovery pf high calorific fraction (without RDF production); however, uncertainty occurring during European Directive targets diffusion, as well the inclusion of other criteria was enough to bring solution P.A5* as the best one at AMARSUL system.

The limitations of the applied SA method has been the time-consuming feature, data missing, static model procedure, being the variation focused on the alternatives proposed. This can be a problem if decision maker does not consider correct waste management alternatives to be chosen. Although, the uncertainty and reliability analysis conducted during LCA and MCDM prove the robustness and confidence of the model.

The research of this study was organized in four main lines, which are summarized in the next paragraphs. In each case a brief discussion will be presented, including the fulfillment of the specific objectives, the validation of the research assumptions and the enumeration of the limitations found during this work. In a final part the future research developments will be discussed.

Role of systems analysis in waste management systems

The systems analysis observed during literature review have shown their appliance in SWM systems occurs in the few decades. They can be divided in two main groups: system engineering models, which include CBA, FM, SM, OM and IMS; and systems assessment tools, considering MIS/DSS/ES, SD, MFA, LCA, RA, EIA, SEA, SoEA and SA. Their application have been focusing on solving practical SWM management issues, like locating facilities, compare treatment and disposal technologies, assess systems options performance, number and capacity expansion, optimizing of waste flows inside of SWM system, compare alternatives of managing SWM system for planning purposes, waste production forecast, to predict recycling targets considering social behavior, to assess waste policy proposals (like taxes, fees).

The advantages of systems analysis application have been translated in a more effective waste management, cost optimization, inclusion of environmental impacts and risks into decision making; also bringing social considerations have been included into the system. Whereas system assessment tools provide a wealth of composite measures of complexity inside procedures/components and between them, joint formulation of human factors and physical/biochemical features in system engineering models in concert with those well defined procedures/components brings about a considerable contribution to the improvement of SWM (Chapters III).

Concerning the application into European countries, and into European SWM systems, the main SWM where systems analysis have been applied is MSW and not specific waste streams since this is the most ancient system to manage waste. Countries that have mostly applied systems analysis techniques include Italy, Sweden, the United Kingdom, and Denmark; countries with a moderate number of applications include France, Germany, Austria and Finland; and countries with low interest in such applications include Spain, Greece, Ireland, Luxembourg, Portugal, Belgium, the Netherlands, Norway and Switzerland (Chapter III). Also in this approach, the most applied models found are LCA, SA and MIS/DSS/ES.

Focusing on southern European countries, which include Portugal, the application of systems analysis models are regulation dependent which can impose the use of such methods in SWM; without regulation, the use of those methods concerning sustainability can be meaningless. It should not be forgotten that system analysis models can be time and data consuming, representing an extra cost to SWM management. An alternative to regulatory incentive, an economic motivation would be relevant to promote system analysis application, namely using models which focus on economic optimization or considers economic aspects during decision making process.

Nevertheless, the models or tools described have individual limitations and none of them has considered the complete vision of the whole waste management cycle, from prevention of waste through to final disposal, except the LCA (Chapter II). Simplification is unavoidable if a time-interesting model results are intended to be use in decision making, which reduces the accuracy of results. Ideal solution procedures normally yield a balance between simplifications of the analysis and the soundness of capturing the essential features resulting in additional complications in systems analysis for SWM (Chapter II). To ensure that results are reliable is imperative the definition of guidelines.

Systems analysis models more adequate to promote sustainable decision making for waste management system

SA and LCA have been the most applied systems assessment models; OM the most applied from system engineering models. Such induces that the joint application of this two groups (instead of methods from the same group) would answer to a more diversified range of SWM problems. For that reason, the combined model of LCA and MCDM, a SA model, concerning the used nomenclature, would be the more correct and justified proposal to be tested in this study.

By definition, SA models refers to the integration of different methodologies in such a way that is geared toward obtaining an analysis, an evaluation or a planning that approaches several management aspects in which the sustainability implications may be emphasized and illuminated (Chapter II). If the methodologies chose to perform SA could answer to the definition proposed, SWM system decision and management will considered the sustainability philosophy. LCA can incorporate all waste life cycle and can calculate potential environmental impact criteria; MCDM can have all different objective functions to be minimized or maximized, depending of its nature, being those objectives representative of the different aspects considered in sustainable management.

LCA ability to model waste management systems and help on decision making

LCA application in the case study was affordable and has produced reliable results. The use of UMBERTO software allows to decision makers to understand the waste life route, not only in AMARSUL boundaries, but all life cycle. Such means that it includes more than one SWM system, at least MSW system and packaging waste system. Graphic presentation of final results is the best option to show results to decision makers, since LCIA results can be misunderstood.

LCA construction requires considerable amount of data, which should be mostly from specific local features. However, if the LCA is focusing to compare alternatives not already implemented, is obvious the use of data not local specific. However, technological approximate data have always been applied. Also, the preclusion of verifying results can be a disadvantage for those decision makers which “need to see to believe it”. It can also be difficult to interpret the results if a decision maker is a non-waste specialist.

With the aid of various data sources, the present practices of LCA for 18 waste management alternatives have shown significant implications in the context of sustainable solid waste management in AMARSUL. With the aid of our customized LCI, the best and the worse options for waste management in our study area may be quantified for decision making under uncertainty although it is a time-consuming and laborious process (Chapter IV).

MCDM ability to help in decision making

The method MCDM is, by definition, a decision making tool. Its ability to be applied in SWM systems is notorious, since it can relate attributes of different purposes (maximization or minimization) and of different units and sizes. The AHP-based Fuzzy Interval TOPSIS method applied was applied successfully, considering weight criteria procedure and inclusion of main stakeholders opinion, and including uncertainty occurring during the implementation of European Directives, resulting into a final and distinguished result.

Future research

The systems analysis model developed to help in decision making presents few limitations and drawbacks which could be improved in a future research:

- To include more sustainable issues, mainly social ones, is determinant to conduct research relating to SWM systems. The missing data concerning PAYT population effects have not allowed the inclusion of this economic instrument from a complete integrated perspective, which could be modeled.

- The system analysis model thus not models reaching targets, what is a limitation which should be solved. Besides, PAYT is not quite implemented in Portugal, which does not allow the application of foreign data into a national situation, highly dependent of public opinion.

- Besides market-based instrument referred, is important to develop a method to include public behavior in source separation collection schemes, since in Portugal there is no economic incentive to promote it. However, the difficult of implementing this is the missing of studies and information which could be used to integrate the model developed.

- Deepen the structure of systems engineering models in the context of IMS which may incorporate more multi-faceted features covering economic, environmental, social, ecological, political; cultural, and managerial aspects for the sustainability assessment of current and future SWM systems (Chapter II).

- Develop large-scale system analysis techniques in order to combine system assessment tools such as EIA, LCA, MFA, and even green accounting with systems engineering models such as optimization models to assess global warming potential, energy saving, and resources conservation practices so as to achieve sustainable waste management goals (Chapter II).

- Waste production phase should be included into the model developed, to understand how prevention of waste can be account into the model. However, this phase is not of the AMARSUL responsibility, being necessary to promote studies to allow the inclusion of this phase into the system analysis model.

ANNEX I – DATA COLLECTION FOR LCA

7 ANNEX I – DATA COLLECTION FOR LCA

In Chapter IV, the methodology, assumptions and results from LCA have been presented. Although, a description of the procedure applied could help the readers to understand how the results have been obtained, in detail LCI. The present annex is referent to the LCI step in LCA methodology.

To develop the LCI step, several assumptions needed to be made in what concerns requirements, quality, assumptions and limitations of data used.

Concerning data requirements, there are two types of data which are relevant to be described. Direct data, related to specific systems processes, and secondary data, related to common processes like electricity, fuels, transport, materials produced outside Portugal and other processes applied to several MSW life cycle phases. Direct data has been obtained from companies belonging to the AMARSUL system (EGF company), environmental licenses and other specific bibliography. Concerning secondary data, such have been inventoried through library existing in UMBERTO, specific for the places where such processes or materials exist or were produced.

In cases where were no data available, different approaches were taken, like adaptations from existing data, to avoid that such life cycle phases would be incomplete. Concerning data quality, there has been defined three criteria to characterize the data: relevance, accessibility and confidence. The relevance is referent to the ability of data being representative of specific process; confidence is concerned to the consistence and data precision, while accessibility is dedicated to the consistence and reproducibility of data. Based on such criteria, has been developed a decision key to choose the data to be admitted in the LCA study. Concerning relevance, data can be characterized according to:

- Temporal reference: primary data collected were representative of existing situations, from date 1997 to 2008. Secondary data were collected and published last then 20 years;
- Geographical reference: there has been always the concern of obtaining primary and secondary data emphasized at geographical location. In cases where such was not possible, general data have been used;

- Technological reference: for primary data has been given preference to specific data belonging to MSW management system AMARSUL. When not possible, bibliography was having been applied. For secondary data, the representative technologies related to materials production or process at national level have been preferred, being collected information at national but also European level;

The data collected have been considered representative of the system, once the criteria have been fulfilled.

Concerning limitations and assumptions, such are registered and reported in the thesis, being additionally tested through sensitivity analysis. The main limitations found during LCA study has been the following:

- Detailed degree of LCA is dependent of detailed degree from data collected in EGF;

- When specific data was not available to simulate life cycle, approaches have been made with existing data from UMBERTO library;

- When primary data were not available, less recent data have been used;

- The LCA study is valid for national situation, although there have been used average and generic data, which in cases where the same technology is used, LCA could be adapted to other European realities.

7.1 LIFE CYCLE INVENTORY

Concerning LCI, this phase consists in collecting data and calculation procedures to quantify systems inputs and outputs. There were inventoried material and energy balances, from the system into background and internally.

The next sub-chapters will describe the assumptions and data used to provide the inventories used in life cycle assessment for each scenario. Primary data are from specific processes from different scenarios; secondary data results from databases.

7.1.1 Municipal solid waste description

According to 'Decreto-Lei n.º 178/2006, de 5 de Setembro' (MAOTDR, 2006), municipal solid waste is the waste resulting from dwellings as well other waste which, by its nature or composition, is similar to household waste. It is usual a corresponding between urban waste/municipal solid waste, being a embracing terminology which reports to waste from domestic source. It also include waste from services, commerce and industry, and also healthcare institutions, with a similar composition of domestic waste (APA, 2006). Physical waste composition provided by EGF (2009) is defined in Table 7.1.

Table 7.1 Physical composition of waste

Components	MSW (%)
fermentables	27.59
green and garden waste	4.10
paper/cardboard packaging	7.25
newspapers and magazines	5.14
other paper/cardboard	1.74
plastic packaging (PP, PVC, PE, PET)	2.23
other plastic packaging	1.15
plastic film	6.92
other plastic	1.04
glass packaging	5.60
other glass	0.23
composites packaging	1.66
other composites	2.48
textiles	2.82
sanitary textiles	8.90
ferrous packaging	1.23
other ferrous	0.27
non-ferrous packaging	0.23
other non-ferrous packaging	0.09
wood packaging	0.43
other wood	1.65
other packaging	0.42
others (inert)	1.50
fines (< 20 mm)	15.33

Other features are needed to model MSW treatment. Moisture, dry organic matter, dry organic biodegradable, heavy metals and nutrients contents are presented in Table 7.2 (Vogt *et al.*, 2002, Fricke *et al.*, 2002, Dehoust, *et al.*, 2002, Rotter, 2004). It is important to remember that for textiles have been considered 50% textile and 50% leather, since in Portugal there is no physical characterization by the last component.

Table 7.2 Chemical composition of waste components

Components	DM (%)	ODM (%DM)	Bio ODM (%ODM)
commingled fermentables	36	87	100
source separated fermentables	45	87	100
commingled green waste	36	84	100
source separated green waste	43	84	100
paper/cardboard	72	87	98
plastic	87	95	5
glass	99	0	0
composites packaging	87	91	78
other composites	87	80	58
textiles	76	85	55
sanitary textiles	73	50	25
ferrous metals (Iron)	89	0	0
non-ferrous metals (Aluminium)	89	0	0
wood	89	90	50
others (inert)	97	0	0
fines (< 20 mm) (equal ½ fermentables and ½ inert)	53	87	100

Concerning elemental analysis, MSW components composition is presented in Table 7.3, also from also from (Vogt *et al.*, 2002, Fricke *et al.*, 2002).

Table 7.3 Elemental composition of MSW components

Components	C (%ODM)	H (%ODM)	O (%ODM)	N (%ODM)	Cl (%ODM)	S (%ODM)	C biogene (%C)
fermentable	51	6.20	44	0.5	0.1	0.1	100
green waste	50	7.9	32	0.0	0.7	0.0	100
paper/cardboard	49	6.4	44	0.2	0.3	0.2	99
plastics	83	13.30	4	0.1	0.1	0.0	5
glass	47	10	40	3.0	0.0	0.0	98
composite packaging	59	6.7	39	2.7	0.7	0.5	60
others packaging	58	6.7	39	2.7	0.7	0.5	20
ttextiles	49	6.6	38	3.1	0.6	0.4	78
sanitary textiles	57	7.7	31	3.6	0.8	0.3	90
ferrous metals (iron)	48	6.3	44	0.5	0.7	0.1	98
non-ferrous metals (aluminium)	48	6.30	44	0.5	0.7	0.1	98
wood	49	7.6	33	0.5	1.5	0.1	100
others (inert)	48	6.3	44	0.5	0.7	0.1	98
fines (< 20 mm) (equal ½ fermentables and ½ inert)	50	6.2	44	0.5	0.4	0.1	99

Concerning heavy metals and fertilizers substances, in Table 7.4. are presented the referent values (Vogt *et al.*, 2002, Fricke *et al.*, 2002).

Table 7.4 Heavy metals and fertilizers substances

Components	Heavy metals (mg/kg DM)								Nutrients (% DM)			
	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	P	K	Mg	Ca
commingled fermentables	5.0	1.0	55.0	153.0	0.5	28.0	90	500	0.4	0.9	0.8	2.2
source separated fermentables	-	0.1	1.8	9.2	0.004	1.3	2.6	30.6	0.4	0.9	0.8	2.2
commingled green waste	-	-	-	-	-	-	-	-	0.5	1.5	0.5	4.4
source separated green waste	-	0.3	4.6	0.1	0.2	3.7	4.8	60	0.5	1.5	0.5	4.4
paper/cardboard	5.0	0.7	9.8	44.8	0.2	6.8	23	295	-	-	-	-
plastics	5.0	66.0	28.6	60.4	0.2	4.3	50	627	-	-	-	-
glass	0.0	0.0	1.3	0.0	0.0	0.0	329	82	-	-	-	-
composites packaging	5.0	1.0	36.0	68.0	0.2	7.4	30	388	-	-	-	-
others packaging	5.0	1.0	7.3	37.5	0.2	9.0	14	90	-	-	-	-
textiles	5.0	2.0	458.4	49	0.1	6.2	74	2304	-	-	-	-
sanitary textiles	5.0	0.5	27.0	23.2	0.2	11.3	10	313	-	-	-	-
ferrous metals (iron)	20.0	21.0	156.0	265.0	-	68.3	582	507	-	-	-	-
non-ferrous metals (aluminium)	20.0	21.0	156.0	265.0	-	68.3	582	507	-	-	-	-
wood	5.0	0.4	5.5	17.9	0.1	3.8	21	158	-	-	-	-
others (inert)	10.0	0.5	80.0	35.0	0.1	45.0	50	70	-	-	-	-
fines (< 20 mm) (equal ½ fermentable and ½ inert)	7.5	0.8	67.5	94	0.3	36.5	70	285	-	-	-	-

7.1.2 Description of MSW management operational units

The elements which composes MSW management system are: waste collection and transport, sorting plant, aerobic and anaerobic MBT, RDF production, RDF incineration with electricity recovery, products transportation, compost soil application, recyclables preparation, materials recycling and landfill.

7.1.2.1 Collection and transport

Waste collection is performed by municipalities. The service can be made by themselves or hiring private collection companies. MSW is temporary deposited into bins existing in the street, and collection vehicles will remove waste, living the container empty. In Table 7.5 is the requirements needed to perform the collection and transport balance provided by municipalities (Aleixo, 2010, Canta, 2010, Didelet, 2010, Gomes, 2010, Pinto, 2010, Valério, 2010). Since AMARSUL does not have BMW collection system, the approach used was to attribute the same distance and diesel fuel consumption of the municipalities that will treat BMW in the future anaerobic digestion MBT, considering a parallel route.

For MSW collection, data have been provided by most AMARSUL municipalities. In cases where data was not provided, approaches concerning waste production has been applied to estimated the distances coursed. For packaging waste, paper/cardboard waste and glass waste, the data was afforded by EGF (2009). For diesel fuel consumptions in MSW collected have been applied the average value, being applied also to BMW collection; for other waste collection data has been collected from Gomes (2009).

Table 7.5 Data requirement for collection and transport waste life cycle stage

Waste collection and transport	MSW	BMW	Packaging waste	Paper/cardboard waste	Glass waste
distance (km)	1,699,646	121,355	641,334	446,296	179,672
diesel fuel consumption (l/100 km)	49.6	49.6	65.0	94.6	78.3
references	Gomes and Rodrigues (2010); Pinto (2010); Canta (2010); Aleixo (2010); Didelet (2010); Valério (2010)		EGF (2009); Gomes (2009)		

7.1.2.2 Sorting plant

Sorting plant has the purpose to separate packaging waste, being only processed packaging waste from yellow and blue container. Glass packaging waste are only temporary stored before being transported to processing plant. The composition of packaging waste provided by EGF (2009) is in Table 7.6.

Table 7.6 Physical composition of packaging waste

Components	MSW (%)
fermentables	2.10
green and garden waste	0.35
paper/cardboard packaging	4.81
newspapers and magazines	4.07
other paper/cardboard	1.70
plastic packaging (PP, PVC, PE, PET)	36.33
other plastic packaging	3.97
plastic film	14.53
other plastic	5.97
glass packaging	3.97
other glass	0.10
composites packaging	7.77
other composites	4.94
textiles	1.01
sanitary textiles	0.58
ferrous packaging	4.97
other ferrous	0.10
non-ferrous packaging	0.10
other non-ferrous packaging	0.11
wood packaging	0.02
other wood	0.97
other packaging	0.42
others (inert)	0.11
fines (< 20 mm)	1.00

Since AMARSUL will have automated sorting plant, that was the technology applied in this operation unit. The packaging waste materials to be sorted are HDPE, LDPE, EPS, PET, mixed plastics (including plastics item in

Table 7.7), composites packaging, ferrous and non-ferrous materials. Glass can also be sorted. Auxiliary materials as lube oil and steel are also mentioned in Table 7.7. The mass balance was based on the need to fulfill Packaging Waste Directive targets, being considered the contamination maximum from 'Despacho n.º 15370/2008, de 3 de Junho'. Mass balance data in

Table 7.7 has been afforded by Rodrigues (2009) and Rodrigo and Castells (2000).

Table 7.7 Mass balance for automated MRF for 1,000 kg of packaging waste

Inputs	Quantity	Units	Outputs	Quantity	Units
packaging waste	1,000	kg	plastics	512.167	kg
lube oil	0.176	kg	refuse	241.504	kg
steel	1.200	kg	composites	93.333	kg
electricity	75,312	kJ	glass	59.513	kg
diesel fuel	1.689	kg	ferrous metals	68.413	kg
			non-ferrous metals	25.958	kg
			steel scrap	1.200	kg
			waste oil	0.176	kg

In the case of paper/cardboard waste, the sorting plant is the old compactation unit for MSW. The sorting is manual, where exists two workers which process negative sorting. The mass balance from this sorting is in Table 7.8.

Table 7.8 Mass balance for manual MRF for 1,000 Mg of paper/cardboard waste

Input	Quantity	Units	Outputs	Quantity	Units
paper/cardboard waste	1,000	kg	paper/cardboard	950	kg
diesel	0.534	kg	refuse	50	kg
steel	1.200	kg	steel scrap	1.200	kg
lube oil	0.009	kg	waste oil	0.009	kg
electric energy	19,260	kJ			

7.1.2.3 Mechanical-biological treatment: aerobic biological processes

AMARSUL aerobic MBT is located in Palmela and have processed in 2008 around 55,000 Mg of MSW. Mechanical processing is enclosing the following equipments:

- One flail mill;
- Two trammels (120 mm and 80 mm);
- Three magnetic separators;
- One eddy current separator;
- Three presses.

Also during mechanical treatment is performed manual sorting of plastics (film, HDPE, PET), and cardboard. Such material plus ferrous and non-ferrous metals are sending to recycling companies, being the rejects send to landfill. The remaining fraction will be treated biologically.

The mass balance in Table 7.9 was based on EGF data (EGF, 2009) concerning aerobic MBT working during year 2008.

Table 7.9 Mass balance for aerobic MBT for 1,000 Mg of MSW waste

Input	Quantity	Units	Output	Quantity	Units
MSW waste	1,000	kg	stabilized residue	100.473	kg
lube oil	0.106	kg	ferrous waste	13.289	kg
electricity	17,026	kJ	fines	146.700	kg
diesel	0.425	kg	plastics	7.359	kg
water	97.070	kg	glass	0.559	kg
structural material	80.169	kg	non-ferrous	0.420	kg
			waste oil	0.1056	kg
			rejects	285.962	kg

The biological treatment trough aerobiosis is conducted in a intensive composting hangar, where waste will be kept during nine weeks. The result product is a fresh compost, being the assumptions provided in Table 7.10. Data is from Vogt *et al.* (2002), Fricke and Müller (1999) and EGF (2009).

Table 7.10 Requirements for model biological treatment from aerobic MBT

Parameters for aerobic treatment (1,000 kg waste in this operation)	Quantity	Units
electricity	36,000	kJ
water	2	%
structure material	8	%
ODM decomposition rate	65	%ODM
carbon decomposition rate	65	%C
nitrogen decomposition rate	11	%N

The fresh compost is conducted to maturation phase, being firstly removed contaminants and other materials which have not decomposed, through the equipments “flop-flow” sieve and densimetric table. The compost is then sent to maturation park, where will be kept during four weeks**,** until be sent to final destination (which is, in this study, the landfill). The assumptions used to model this operation are in Table 7.11. Data is from Vogt *et al.* (2002) and EGF (2009).

In this unit do not produced waste water, since they are recirculated in composting hangar.

Table 7.11 Requirements for maturations step

Maturation step requirements (1,000 kg waste in this operation)	Quantity	Units
electricity	36,000	kJ
water	32	%
ODM decomposition rate	20	%ODM
carbon decomposition rate	20	%C
nitrogen decomposition rate	11	%N
refuses	74	% of obtained compost
Diffuse gasous emissions		
NH_4	96	% do N in air
N_2O	2	% do N in air
CH_4	3	% do C in air
TOC	2	% do C in air
N	2	% do N in air
CO_2	95	% do C in air
compost moisture obtained	40	%

Concerning emissions from biofilter, in Table 7.12 are presented the emissions resulting from this operation, based on Schwing (1999) and Frickle and Müller (1999).

Table 7.12 Factor emissions from biofilter (for 1,000 kg of MSW input in the unit)

Pollutant	Quantity	Units
formaldehyde	5.88E-05	kg
perchloroethylene	0.00058	kg
ammonia (a)	0.0611	kg
chlorine (a)	0.0187	kg
dinitrogen monoxide (a)	0.01752	kg
fluorine (a)	0.000818	kg
hydrogen chloride (a)	0.00169	kg
hydrogen sulfide (a)	2.34E-07	kg
antimony (a)	1.65E-05	kg
arsenic (a)	2.54E-06	kg
cadmium (II) ion (a)	1.61E-06	kg
chromium (a)	1.6E-05	kg
chromium (VI) (a)	1.57E-05	kg
cobalt (a)	3.14E-06	kg
copper (II) ion (a)	8.89E-06	kg

Pollutant	Quantity	Units
lead (II) ion (a)	1.58E-05	kg
manganese (a)	3.87E-05	kg
mercury (II) ion (a)	1.16E-05	kg
nickel (a)	7.98E-05	kg
thallium (a)	8.26E-06	kg
vanadium (a)	1.67E-05	kg
sulfur dioxide (a)	0.0318	kg
methane (a)	0.01515	kg
1.1.1-trichloroethane (a)	3.23E-05	kg
1.2-dichloroethane (a)	6.25E-05	kg
dichloromethane (a)	0.000715	kg
tetrachloromethane (a)	3.84E-05	kg
trans 1.2-dichloroethene (a)	6.45E-05	kg
trichloroethene (a)	0.000429	kg
trichloromethane (a)	4.22E-05	kg
vinyl chloride (a)	0.000191	kg
1.2.3-trichlorobenzene (a)	0.000133	kg
1.2.4-trichlorobenzene (a)	5.85E-05	kg
1.2-dichlorobenzene (a)	0.00055	kg
1.3.5-trichlorobenzene (a)	0.000117	kg
1.3-dichlorobenzene (a)	3.6E-05	kg
1.4-dichlorobenzene (a)	5.45E-05	kg
chlorobenzenes (a)	0.000588	kg
chlorophenols (a)	1.65E-08	kg
dioxins (unspec.) (a)	3.59E-11	kg
PCB (a)	1.25E-07	kg
PCDD. PCDF (a)	2.11E-11	kg
R 211 (a)	0.000497	kg
butanol (a)	0.00771	kg
acetaldehyde (a)	0.002265	kg
pentanal (a)	0.000141	kg
propanal (a)	0.00089	kg
cyclohexane (a)	0.000265	kg
decane (a)	0.006225	kg
heptane (a)	0.001793	kg
hexane (a)	0.000155	kg
nonane (a)	0.00206	kg

Pollutant	Quantity	Units
octane (a)	0.000793	kg
turpentine (a)	0.006122	kg
benzene (a)	0.000281	kg
ethyl benzene (a)	0.00144	kg
o-xylene (a)	0.000892	kg
styrene (a)	0.000954	kg
toluene (a)	0.001654	kg
xylene (a)	0.00332	kg
ethyl acetate (a)	0.000266	kg
acetic acid (a)	0.000809	kg
ethers. unspec. (a)	0.000202	kg
2-hexanone (a)	1.56E-05	kg
acetone (a)	0.00822	kg
acrolein (a)	3.15E-05	kg
acenaphtylene (a)	3.44E-05	kg
anthracene (a)	3.82E-07	kg
benzo(a)anthracene (a)	3.94E-08	kg
benzo(a)pyrene (a)	1.4E-08	kg
benzo[ghi]perylene (a)	1.78E-08	kg
benzo[k]fluoranthene (a)	1.4E-08	kg
chrysene (a)	2.05E-08	kg
dibenz(a)anthracene (a)	1.73E-07	kg
fluoranthene (a)	4.87E-07	kg
fluorene (a)	1.78E-06	kg
indeno[1.2.3-cd]pyrene (a)	1.4E-08	kg
naphtalene (a)	0.000244	kg
phenanthrene (a)	1.22E-05	kg
TOC (a)	0.11826	kg
HC. unspec. (w)	0.001734	kg

7.1.2.4 Mechanical-biological treatment: anaerobic biological processes

In the case of anaerobic MBT, the expected unit to be constructed at AMARSUL system will be located in Seixal municipality. The unit is composed by mechanical treatment, where recyclable waste is removed, as also combustible fraction to RDF production or to be directely burned, being the remain fraction sent to anaerobic digestion. In the unit is expected a treatment line for BMW, being this step observed forward.

Normally, the mechanical treatment has several equipments to process MSW, like flail mills, trommels, magnetic separator, eddy currents and ballistic separator. Also, is expected to exist a manual sorting. Those equipments and manual labor are capable to separate materials for recycling and high calorific fraction, being the total mass balance of the plant estimated in Table 7.13, including electricity production and wastewater treatment plant emissions. In the alternatives where RDF production was not considered, the part which would go to RDF is going to sanitary landfill, has rejects. The other situation is thar calorific fractions are directely send to incinerationplant. For all the process, data is collected from Vogt *et al.* (2002), EGF (2009), Fricke *et al.* (2002), Loll (1994,1998).

Table 7.13 Mass balance for anaerobic MBT for 1,000 Mg of MSW waste

Input	Quantity	Units	Outputs	Quantity	Units
MSW	1,000	kg	stabilized residue	138.918	kg
lube oil	0.1056	kg	ferrous waste	14.440	kg
chlorine	0.001	kg	rejects	199.804	kg
precipitants	0.366	kg	high calorific material/rejects	475.440	kg
quicklime	0.005	kg	plastics	4.044	kg
electricity	137,445.27	kJ	electric energy	202,093,4	kJ
diesel fuel	0.986	kg	waste oil	0.1056	kg
heat energy	11.593	kg	non-ferrous	2.688	kg
structural material	2.397	kg	composites	1.108	kg
water	111.440	kg	paper/cardboard	5.922	kg
			glass	1.670	kg
			Emissions		
			carbon dioxide, renewable (a)	3.201	kg
			NOx (a)	0.190	kg
			sulfur dioxide (a)	0.025	kg
			particles (a)	0.002	kg
			carbon monoxide (a)	0.269	kg
			dinitrogen monoxide (a)	0.004	kg
			hydrogen chloride (a)	0.001	kg
			hydrogen fluoride (a)	0.000	kg
			arsenic (a)	1.9E-06	kg

Input	Quantity	Units	Outputs	Quantity	Units
			beryllium (a)	2.2E-08	kg
			cadmium (a)	3.97E-07	kg
			chromium (a)	2.65E-07	kg
			mercury (a)	1.85E-06	kg
			nickel (a)	8.82E-08	kg
			methane (a)	0.006613	kg
			dichloromethane (a)	2.87E-08	kg
			tetrachloroethene (a)	3.57E-08	kg
			dioxins (unspec.) (a)	1.41E-14	kg
			hexachlorobenzene (a)	1.76E-09	kg
			PCB (a)	9.26E-09	kg
			PCDD, PCDF (a)	7.94E-15	kg
			benzene (a)	2.65E-08	kg
			ethyl benzene (a)	7.94E-07	kg
			m-xylene (a)	3E-06	kg
			o-xylene (a)	7.94E-07	kg
			toluene (a)	2.69E-06	kg
			benzo(a)pyrene (a)	2.2E-09	kg
			particles (PM10) (a)	0.002204	kg
			cadmium (w)	6.77E-06	kg
			calcium (w)	0.318146	kg
			chromium (w)	0.000348	kg
			copper (w)	0.000953	kg
			lead (w)	0.000571	kg
			magnesium (w)	0.099378	kg
			mercury (w)	3.13E-06	kg
			nickel (w)	0.000178	kg
			pottassium (w)	0.628511	kg
			zinc (w)	0.003189	kg
			ammonium (w)	0.081681	kg
			nitrate (w)	0.048408	kg
			phosphorous compounds as P (w)	0.000707	kg
			BOD-5 (w)	0.057632	kg
			COD (w)	0.792445	kg
			sewage sludge (20% DM)	17.72194	kg
			sewage, clarified	71.61557	kg
Note: (a) – emission to ar; (w) – emission to water;					

From this operation is resulted a digestate, with a several decomposed substances. The remain part will be decomposed through aerobic treatment. From that process results a fresh compost, being the parameters used to model this step in Table 7.14, from Vogt *et al.* (2002), EGF (2009), Fricke *et al.* (2002).

Table 7.14 Parameters for pre-composting aerobic treatment of digestate

Pre-composting parameters (1,000 kg input waste in this operation)	Quantity	Units
Electricity	36,000	kJ
Carbon decomposition rate	16	% C
Nitrogen decomposition rate	11	% N
Structural material	5	%

After this process, fresh compost needs to be maturated, in piles, for eleven weeks, resulting in mature compost. Also, for this operation, in Table 7.15 are presented the parameters applied, from Vogt *et al.* (2002), EGF (2009), Fricke *et al.* (2002)

Table 7.15 Requirements for post-composting process (maturation)

Post-composting parameters (1,000 kg input waste in this stage)	Quantity	Units
water	20	%
electricity	36,000	kJ
decomposition rate	50	% ODM
carbon decomposition rate	50	% C
nitrogen decomposition rate	29	% N

Concerning biogas produced, it will be used for electricity and heat production, according to expected unit. The parameters from biogas burning are in Table 7.16, from Soyez *et al.* (2000).

Table 7.16 Parameters for biogas electricity production (1 m^3)

Electricity production from biogas (1 m^3 biogas input)	Quantity	Units
carbon dioxide, renewable (a)	0.089154	kg
carbon monoxide (a)	0.007545	kg
dinitrogen monoxide (a)	0.000124	kg
hydrogen chloride (a)	1.86E-05	kg
hydrogen fluoride (a)	1.11E-05	kg
arsenic (a)	5.32E-08	kg
beryllium (a)	6.18E-10	kg

cadmium (a)	1.11E-08	kg
chromium (a)	7.42E-09	kg
mercury (a)	5.19E-08	kg
nickel (a)	2.47E-09	kg
NOx (a)	0.005319	kg
sulfur dioxide (a)	0.000693	kg
methane (a)	0.000186	kg
dichloromethane (a)	8.04E-10	kg
tetrachloroethene (a)	1E-09	kg
dioxins (unspec.) (a)	3.96E-16	kg
hexachlorobenzene (a)	4.95E-11	kg
PCB (a)	2.6E-10	kg
PCDD, PCDF (a)	2.23E-16	kg
benzene (a)	7.42E-10	kg
ethyl benzene (a)	2.23E-08	kg
m-xylene (a)	8.41E-08	kg
o-xylene (a)	2.23E-08	kg
toluene (a)	7.54E-08	kg
benzo(a)pyrene (a)	6.18E-11	kg
particles (PM10) (a)	6.18E-05	kg
electric energy	5.670	kJ
Note: (a) – air emissions		

Concerning wastewater treatment, the mass balance specified for this step is detailed in Table 7.17. Wastewater characteristics have been provided by Loll (1994, 1998) and Vogt *et al.* (2002), being the treatment process applied based on aeration tank and reverse osmosis, including sewage sludge drying through flotation (where are used precipitants) and dehydration (EGF, 2009). To model the wastewater treatment plant, data from Martinho *et al.* (2008) and Yamada and Jung (2007) have been used. The air treatment predicted and simulated has been through biofilter, where the air contamination has been provided by the average contamination obtained by den Boer *et al.* (2005), as well the efficiency cleaning.

The sewage sludge destiny will be the landfill, however, its production is below the level defined to perform the LCA (1%), based on ifeu (1994).

Table 7.17 Mass balance of wastewater treatment unit, considering 1,000 kg

Input	Quantity	Units	Outputs	Quantity	Units
wastewater	1,000	kg	cadmium (w)	5,53E-06	kg
precipitants	4,262897	kg	calcium (w)	3,708293	kg
electric energy	79200	kJ	chromium (w)	0,000284	kg
			copper (w)	0,000778	kg
			lead (w)	0,000466	kg
			magnesium (w)	1,158345	kg
			mercury (w)	2,56E-06	kg
			nickel (w)	0,000145	kg
			pottassium (w)	7,325884	kg
			zinc (w)	0,002602	kg
			ammonium (w)	0,328299	kg
			nitrate (w)	0,19698	kg
			phosphorous compounds as P (w)	0,008238	kg
			BOD-5 (w)	0,41985	kg
			COD (w)	1,399499	kg
			sewage sludge (20% DM) (wfr)	206,566	kg
			sewage, clarified	834,7472	kg

In the cases where anaerobic digestion line is predicted, the LCI provided is the one presented in Table 7.18. In this units is not recovered material to produce RDF, being the dataued to model it based on Vogt *et al.* (2002), EGF (2009), Fricke *et al.* (2002), APA (2009).

Table 7.18 Mass balance for anaerobic digestion for 1,000 Mg of BMW waste

Input	Quantity	Units	Outputs	Quantity	Units
Lube oil	0.1056	kg	Compost	335.510	kg
Chlorine	0.004	kg	Ferrous waste	2.5	kg
Precipitants	1.084	kg	Rejects	11.661	kg
Quicklime	0.015	kg	Plastics	49.95	kg
Electricity	154,210.88	kJ	Fines	6.494	kg
Diesel fuel	0.986	kg	Electric energy	763,879,46	kJ
Heat energy	34.640	kg	Waste oil	0.1056	kg
Structural material	8.863	kg	Non-ferrous	1.5	kg
Water	300.462	kg	Glass	18.998	kg
			Emissions		
			Carbon dioxide, renewable (a)	9.528	kg

Input	Quantity	Units	Outputs	Quantity	Units
			NOx (a)	0.564	kg
			Sulfur dioxide (a)	0.0736	kg
			Particles (a)	1.54E-6	kg
			carbon monoxide (a)	0.800	kg
			dinitrogen monoxide (a)	0.013	kg
			hydrogen chloride (a)	0.002	kg
			hydrogen fluoride (a)	0.001	kg
			arsenic (a)	5.64E-6	kg
			beryllium (a)	6.56E-8	kg
			Cadmium (a)	1.181E-6	kg
			chromium (a)	7.873E-7	kg
			mercury (a)	5.511E-6	kg
			nickel (a)	2.624E-7	kg
			methane (a)	0.020	kg
			dichloromethane (a)	8.529E-8	kg
			tetrachloroethene (a)	1.06E-7	kg
			dioxins (unspec.) (a)	4.20E-14	kg
			hexachlorobenzene (a)	5.25E-9	kg
			PCB (a)	2.76E-8	kg
			PCDD, PCDF (a)	2.36E-14	kg
			benzene (a)	7.87E-8	kg
			ethyl benzene (a)	2.36E-6	kg
			m-xylene (a)	8.92E-6	kg
			o-xylene (a)	2.36E-6	kg
			toluene (a)	8.00E-6	kg
			benzo(a)pyrene (a)	6.56E-9	kg
			particles (PM10) (a)	0.006	kg
			Cadmium (w)	4.35E-06	kg
			calcium (w)	0.977	kg
			chromium (w)	0.000153	kg
			copper (w)	0.000389	kg
			lead (w)	0.000210	kg
			magnesium (w)	0.284	kg
			mercury (w)	1.45E-06	kg
			nickel (w)	8.99E-5	kg
			pottassium (w)	1.901	kg
			zinc (w)	0.00145	kg

Input	Quantity	Units	Outputs	Quantity	Units
			ammonium (w)	0.164	kg
			nitrate (w)	0.0971	kg
			phosphorous compounds as P (w)	0.00167	kg
			BOD-5 (w)	0.172	kg
			COD (w)	2.359	kg
			sewage sludge (20% DM)	52.745	kg
			sewage, clarified	214.443	kg
Note: (a) – air emissions; (w) – water emissions					

The compost application into soil has also emissions into the environment, mainly into soil.

7.1.2.5 Sanitary landfill

The wastes which goes to sanitary landfill has different sources, from mixed MSW to rejects and refuses from several operational units. The emissions from the landfill can be to air, soil and water. In chapter IV has been described the assumptions used to model the biogas and leachate production. In this annex will be referred only the detailed information concerning biogas and leachate emissions.

The emissions during the biogas burning have bene provided by den Boer *et al.* (2005), being applied average values for the following pollutants presented in Table 7.19.

Table 7.19 Content of pollutants in flue gas from landfill gas engine

Pollutants	Quantity	Units
Trichloroethane	8.80E-04	mg/m^3
1,2-Dichloroethane	8.30E-04	mg/m^3
Benzene	5.20E-03	mg/m^3
Carbonmonoxide	9.83E+02	mg/m^3
Chlor (Cl-tot.)	1.10E-01	mg/m^3
Chloroform	8.30E-04	mg/m^3
Chrom	1.10E-06	mg/m^3
Dichloromethane	8.30E-04	mg/m^3
Ethylbenzene	1.80E-02	mg/m^3
Fluor(F-tot)	2.10E-02	mg/m^3
Hydrogenchloride	1.16E+01	mg/m^3

Hydrogenfluoride	1.03E+00	mg/m^3
Hydrogensulphide	8.30E-02	mg/m^3
Mercury	6.90E-08	mg/m^3
NMVOC	5.00E+01	mg/m^3
Nitrogen oxides	6.53E+02	mg/m^3
PAHs	1.40E-02	mg/m^3
Lead	8.50E-06	mg/m^3
PCB	2.70E-06	mg/m^3
Dioxins	3.29E-07	mg/m^3
PM10	1.80E+01	mg/m^3
Sulphurdioxide	1.86E+02	mg/m^3
Tetrachloroethene	3.30E-04	mg/m^3
Trichloroethene	5.00E-03	mg/m^3
Vinylchloride	2.00E-03	mg/m^3

Concerning leachate production calculation from water in waste mass has been assumed to have a residual water content of 15% weight (Schwing, 1999).

The formula used to quantify leachate production are:

- Diffuse emission dependent from precipitation:

DISW1=(15/100)(N24T1/100)*PHAA*(JNS/20)+(15/100)*(N25T1/100)*PHAB*(JNS/20)*

The assumptions considered in the previous formulas are described in Table 7.20.

Table 7.20 Parameters applied to estimate leachate production

Parameters	Values
Annual precipitation (JNS)	1,550 mm
Leachate production during phase A (N24T1)	40%
Leachate production during phase B (N25T1)	8%
Duration phase A (PHAA)	10 years
Duration phase B (PHAB)	20 years

- Diffuse emission dependent from water inside residues:

DISW2=(85/100)(N24T1/100)*(1-90/100)*PHAA*(JNS/20)+(85/100)*(N25T1/100)*(1-90/100)*PHAB*(JNS/20)*

- Diffuse emission from waste decomposition:

*SWM1=(waste quantity)*0.5*1.25*0.24+(0.24*waste quantity*30/100*30/100)*
(waste quantity)*0.5*1.25 – is the leachate from waste decomposition, calculated from organic carbon content and that each kg of carbon that degrades results in 1.25 kg of water.

(waste quantity*30/100*30/100) – is the moisture content in waste, assuming that 30% of moisture will became leachate.

From leachate produced from waste, only 24% will originate diffuse emission, only being collected 76%. 24% and 76% are round numbers for leachate collection, based in German landfills. It was also considered that leachate collection system can collect 90% of leachate produced.

At the end, leachate lost by diffusion will be equal to:

*Diffuse emissions = waste landfilled (tones) * (DISW1+DISW2)+SWM1*

Collected leachate will be given by the formulas:

- Stationary emission from landfill derived from waste mass (water infiltrated)

GESFW = (85/100)(N24T1/100)*(90/100)*PHAA*(JNS/20)+(85/100)*(N25T1/100)*(90/100)*PHAB*(JNS/20)*

- Stationary emission from leachate from waste decomposition

*SWM2 = (waste quantity)*0.5*1.25*0.76+(0.76* waste quantity*30/100*30/100)*
Total leachate collected will be:

*Leachate collected = waste landfilled*GESFW+SWM2*

The leachate treatment mass balance is the same has the one presented in Table 7.17, since the same treatment technology is to be used.

Concerning volume of waste landfilled, which translates in occupying soil area, its determination is based on that waste landfilled has density equal to 1 t/m^3. Land use is determined by the ratio of volume of waste landfilled and 20 meters of high.

7.1.2.6 RDF production

The production of RDF is still in open at national level, since the *Estratégia Nacional do CDR* does not define which will be the units that will produce RDF and where will be located. The present units existing are only for industrial non-hazardous waste. For that reason, a simple RDF production unit has been considered, where sieving, shredding, metals separation, wind or ballistic separator are applied. The first steps already occur in MBT unit, being only conducted the last stage. The efficiency of air separator has been used, from Fricke *et al.* (2003), presented in Table 7.21.

Table 7.21 Efficiency of air separator related to individual materials in high calorific waste fraction (zigzag separator)

Material	Separation efficiency (%)
paper	97.2
metals	13.6
plastics	94.6
composites	84.8
biowaste	95.9
inerts	56.3
textiles	72.1
fines	99.7
others	95.8

The mass balance of processing RDF is in Table 7.22.

Table 7.22 Mass balance for RDF plant for 1,000 Mg of RDF obtained

Input	Quantity	Units	Outputs	Quantity	Units
Lube oil	0.12	kg	RDF	1,000	kg
Electricity	90,000	kJ	Lube oil	0.12	kg
Diesel fuel	0.4185	kg	Refuse	76	kg
refuse waste	1,076	kg			

7.1.2.7 Products transportation

Products from MSW management system have to be conducted to final destination. The distances have been obtained from Google maps (2010) website and diesel consumption from transportation companies. The products will be transported by different agents. Also for units refuse, the distance to appropriate landfills has been also calculated considering the same sources. For products inside of SPV system, the technical specifications for recyclables transportation were used (described in 'Despacho n.º 15370/2008, de 3 de Junho'), being the overall data provided in Table 7.23. In case of diesel consumption, it was assumed 25 l/100 km, based on transportation companies.

Table 7.23 Technical specifications for transport AMASUL products

Products transport	Distances (km)	
	Pre-processors	Recyclers/Incineration[1]/Agriculture[2]
Ferrous metals	241.3	521.5
Non-ferrous metals	259.3	592.2
PE	0	238.6
PET	0	210.7
EPS	0	293.0
Mixed plastics	0	524.0
Paper/cardboard	339.9	811.2
Composites	210.2	1116.5
Glass	233.0	60.5
RDF[1]	0	45.4
Compost[2]	0	73.7

However, the AMARSUL product will have other destinations. In general, all the materials are send to pre-processors companies, which improves material quality through contaminants removal to send to recycling companies. The exception are plastic materials, which will be send directly from AMARSUL to recyclers. Even to calculate the distances was provided an weighted average distance from AMARSUL to the several re-processors, based on the annual quantity processed in those units.

The same procedure have been applied to gain the distance from pre-processors into recycling units. The distances from pre-processors to recycling units are described in Table 7.24.

Table 7.24 Technical specifications for transport materials to recycling units

Materials	Glass	Metals	Paper/ cardboard	EPS	PE/PP	PET	Mixed plastics	Composites
Load (tonnes)	25	20 for ferrous; 10 for non-ferrous	24	0.75	20 for film; 11 for HDPE	10	17	23

7.1.2.8 Glass recycling

The glass waste from AMARSUL system is now going to be transported into processing units, which are responsible to remove contaminants and prepare the glass to be used in glass kilns. There are two units in Portugal, being modeled the one located in Figueira da Foz, named Vidrociclo.

In this unit, contaminants are removed with magnetic separators, horizontal vibrating sieves, being the glass comminuted through hammer mills. Eddy current and infra-red optical separator are also used. The mass balance of the process is presented in Table 7.25, based on Rodrigo and Castells (2004) and APA (2009).

Table 7.25 Mass balance for glass pre-processor plant for 1,000 Mg of waste glass

Input	Quantity	Units	Outputs	Quantity	Units
Lube oil	0.176	kg	Glass	980	kg
Electricity	36,000	kJ	Waste oil	0.176	kg
Diesel fuel	2.0088	kg	Rejects	10.5	kg
Steel	1.2	kg	Steel scrap	8.7	kg
waste glass	1,000	kg	Non-ferrous	2	kg

The outputs from re-processing units are conducted to correct destination, *e.g.* waste oil will be conducted into recycling units, rejects will be landfilled and metals will be conducted into recycling units. All the distances to this destinations have been gained from Google maps (2010), being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models.

The recycling glass units existing in Portugal are Saint-Gobain Mondego, Santos Barosa, Sotancro, Gallovidro and BA Vidro. The mass balance from recycling glass into bottles are shown in Table 7.26. The sources of energy used in recycling process are electricity and natural gas, being the last fuel used for heat energy production. The emissions presented are related to the recycling process itself, being based on APA (2009), Mata (1998) and ProBas (2004).

Table 7.26 Mass balance for glass recycling plant for 1,000 Mg of glass waste

Inputs	Quantity	Units	Outputs	Quantity	Units
lube oil	3	kg	carbon dioxide. fossil (a)	8.395246	kg
steel	1	kg	carbon monoxide (a)	0.006118	kg
electric energy	752,000	kJ	dinitrogen monoxide (a)	0.005008	kg
heat energy	4,378,000	kJ	hydrogen chloride (a)	0.00457	kg
glass (wfr)	1,000	kg	hydrogen fluoride (a)	0.00048	kg
water (process)	3281.6	kg	cadmium (a)	0.002	kg
			lead (a)	0.133	kg
			zinc (a)	0.024	kg
			NOx (a)	0.144841	kg
			sulfur dioxide (a)	0.295116	kg
			VOC, unspec. (a)	0.00241	kg
			particles (a)	0.146212	kg
			suspended solids (w)	2.0231	kg
			BOD-5 (w)	0.00104	kg
			COD (w)	0.00756	kg
			oil, detergents (w)	0.03742	kg
			bottles	1,000	kg
			steel scrap (wfr)	1	kg
			waste oil (wfr)	3	kg
			sewage, unspec.	3281.6	kg

7.1.2.9 Ferrous and non-ferrous metals recycling

The ferrous and non-ferrous metals obtained from the different infrastructures is now conducted to processing units, where a more refined sorting will occur. There exists few processing units for ferrous and non-ferrous metals, being the ones for non-ferrous: Ambitrena, Batistas-reciclagem de Sucatas, Recifemetal, Constantino, Riometais and Sucatas do Ramil. For ferrous metals the same units receive it, plus Gar company. The mass balance of this units are presented in Table 7.27 based from Rodrigo and Castells (2000).

Table 7.27 Mass balance for ferrous and non-ferrous metals pre-processor plant for 1,000 Mg of specific waste

Input	Quantity	Units	Outputs	Quantity	Units
lube oil	0.176	kg	processed metal	950	kg
electricity	36,000	kJ	waste oil	0.176	kg
diesel fuel	2.0088	kg	rejects	50	kg
steel	1.2	kg	steel scrap	1.2	kg
waste metal	1,000	kg			

Most ferrous and non-ferrous metals recycling occurs in Spain (Fileira Metal, 2010). For non-ferrous metals, the units considered belongs to companies Recial (Portugal), Alcoa (Spain) and Alcan (Spain). Since it was not able to calculate specific emissions from recycling units, it has been considered data from UMBERTO software (ETH Zurich, 2008), Boustead, 2000), being the mass balance presented in Table 7.28 and Table 2.9. The outputs from re-processing units and recycling units are conducted to correct destination, e.g. waste oil will be conducted into recycling units and rejects, slags and ashes will be landfilled. All the distances to this destinations have been gained from Google maps (2010), being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models. The refuses from processing plants are send to Portuguese industrial non-hazardous waste landfills.

Table 7.28 Mass balance for ferrous recycling plant for 1,000 Mg of ferrous waste

Input	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	665,638.9	kJ	ammonia (a)	1.71E-05	kg
KEA (nuclear)	635,161.9	kJ	carbon dioxide, fossil (a)	329.4764	kg
KEA, fossil total	4,373,712	kJ	carbon monoxide (a)	0.055194	kg
KEA, unspec.	192.0189	kJ	dinitrogen monoxide (a)	0.00012	kg
brown coal (r)	120.4642	kg	hydrogen chloride (a)	0.036759	kg
hard coal (r)	59.45398	kg	hydrogen fluoride (a)	0.005339	kg
crude oil (r)	23.86339	kg	arsenic (a)	2.45E-05	kg
natural gas (r)	17.98449	m^3	cadmium (a)	0.000212	kg
limestone (r)	0.026388	kg	chromium (a)	0.005364	kg
iron (wfr)	1,000	kg	cobalt (a)	7.64E-05	kg
cooling water	86.89576	kg	copper (a)	0.000918	kg
water, unspec.	0.337581	kg	lead (a)	0.004227	kg
			manganese (a)	0.011455	kg
			nickel (a)	0.000268	kg
			zinc (a)	0.009182	kg
			NOx (a)	0.944747	kg
			sulfur dioxide (a)	1.635354	kg
			methane (a)	0.659167	kg
			NMVOC, unspec. (a)	0.226289	kg
			VOC, unspec. (a)	0.009014	kg
			particles (a)	0.28246	kg
			steel	909.0909	kg
			slags and ash	91.03742	kg
			sewage (cooling water)	85.513	kg
			sewage unspec.	0.094617	kg
Note: (r) – resource; (wfr) – waste for recovery; (a) – emissions to air					

The outputs from re-processing units and recycling units are conducted to correct destination, *e.g.* waste oil will be conducted into recycling units and rejects, slags and ashes will be landfilled. All the distances to this destinations have been gained from Google maps (2010), being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models.

Table 7.29 Mass balance for non-ferrous recycling plant for 1,000 Mg of non-ferrous waste

Inputs	Quantity	Units	Outpus	Quantity	Units
chlorine	1.548887	kg	ammonia (a)	0.019361	kg
hydrochloric acid	0.193611	kg	carbon dioxide. fossil (a)	666.9894	kg
nitrogen	1.742498	kg	carbon monoxide (a)	0.251694	kg
salts. inorg.	13.26234	kg	chlorides (a)	0.048403	kg
sodium hydroxide	1.548887	kg	chlorine (a)	0.000474	kg
sulfuric acid	7.744434	kg	dinitrogen monoxide (a)	0.001355	kg
KEA (hydro)	193,519.7	kJ	hydrogen chloride (a)	0.028074	kg
KEA (nuclear)	1,223,973	kJ	hydrogen fluoride (a)	0.005905	kg
KEA. fossil total	10,074,176	kJ	hydrogen sulfide (a)	0.002711	kg
lime	7.163601	kg	nitrogen (a)	2.420136	kg
alloying additives	75.50823	kg	NOx (a)	0.89061	kg
brown coal (r)	39.01258	kg	phosphine (a)	0.000503	kg
hard coal (r)	35.81801	kg	sulfur dioxide (a)	1.548887	kg
crude oil (r)	23.71733	kg	VOC (hydrocarbons) (a)	2.032914	kg
natural gas (r)	238.441	m^3	particles (a)	0.203291	kg
fuel oil. light	0.002323	kg	aluminium ingot	968.0542	kg
aluminium scrap. reprocessed (wfr)	1,000	kg	slags and ash	25.26621	kg
water (process)	7.744434	kg	aluminium oxide	115.1985	kg
			scrap (iron)	2.129719	kg
			waste. unspec.	0.077444	kg
Note: (r) – resource; (wfr) – waste for recovery; (a) – emissions to air					

7.1.2.10 PET recycling

The PET separated in AMARSUL infrastructures are directly send to recycler plant. The only plant existing in Portugal is located in Portalegre (called Artenius), being the other in Spain (called Extremadura Torrepet). The unit processes PET to produce PET film. It also produces a granulate which can be used to produce tubes. The mass balance is in Table 7.30, from APA (2009) and ProBas (2004).

The outputs from re-processing units are conducted to correct destination, *e.g.* waste oil will be conducted into recycling units, rejects and sewage sludge will be landfilled and metals will be conducted into recycling units. All the distances to this destinations have been gained from Google maps (2010) being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models.

Table 7.30 Mass balance for PET recycling plant for 1,000 Mg of PET waste

Inputs	Quantity	Units	Outputs	Quantity	Units
water	2,259.542	kg	PE granulate	59.542	kg
sodium hydroxide	7.634	kg	PE, amorphous	763.359	kg
electricity	2,671,755.7	kJ	rejects	142.748	kg
quicklime	7.634	kg	sewage sludge	19.847	kg
fatty alcohols	3.053	kg	steel scrap	6.870	kg
PET waste	1,000	kg	sewage, clarified	2,259.542	kg
Film production					
PET amorphous	1,000	kg	PET film	1000	kg
electricity	1,638,000	kJ			

7.1.2.11 PE recycling

Like happens with PET, also PE (LDPE and HDPE) are directly send to recycling units. Several recycling PE units exists in Portugal, being calculated the average distance from AMARSUL to those units. The units which recycle PE are: Ambiente, Sirplaste, Micronipol, FAP, IRP, Trinoplás and Grijótubos.

The common recycling process produces a regranulate, which can be used to produce tubes, being the mass balance presented in Table 7.31, based on Arena *et al.* (2003).

For this study the product chosen has been tubes. Also to calculate the distance between recyclers and PE tubes producers were obtained from GoogleMaps©, diesel consumption also of 25 liters/100 km, being a cargo of 24 tonnes.

The outputs from re-processing units are conducted to correct destination, *e.g.* waste oil will be conducted into recycling units, rejects and sewage sludge will be landfilled and metals will be conducted into recycling units. All the distances to this destinations have been gained from Google maps (2010) being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models.

Table 7.31 Mass balance for PE recycling plant for 1,000 Mg of PE waste

Input	Quantity	Units	Output	Quantity	Units
sodium hydroxide	294.0147	kg	regranulate (PE)	700.035	kg
fatty alcohols	0.910046	kg	rejects	285.9643	kg
zinc stearate	0.021001	kg	sewage sludge	94.57473	kg
precipitants	1.120056	kg	steel scrap	0.070004	kg
steel	0.070004	kg	sewage clarified	355.6178	kg
limestone	1.19006	kg			
HDPE	14.0007	kg			
electric energy	161,288.1	kJ			
PE/PP (wfr)	1,000	kg			
water (process)	355.6178	kg			
Pipes production					
regranulate (PE)	1,000	kg	PE tube	1,000	
water	19,444	kg	sewage	19,444	
electricity	1,800,000	kJ			

7.1.2.12 EPS recycling

Mitromar, Plastimar, Internoplaste, Contraven and Petibol are the companies which recycles EPS. The recycling unit considered simply commingle the EPS waste, being produced EPS balls which can have different applications, such as soil lightener. The mass balance of EPS recycling is in Table 7.32 (Silva, 2010). The rejects are landfilled, being the transport (25 l/100 km, 24 t load) and final destination modeled.

Table 7.32 Mass balance for EPS recycling plant for 1,000 Mg of PS waste

Input	Quantity	Units	Output	Quantity	Units
PS waste	1,000	kg	EPS	933	kg
electricity	382.500	kJ	rejects	67	kg

7.1.2.13 Mixed plastics recycling

Mixed plastics recycling are also recycled in Portugal, by one company (Extruplás) and in Spain, the Ligeplas. The result from recycling is wood plastic, used to produce outdoor furniture. In Table 7.33 is presented the mass balance, being the data applied from Alves, 2010.

Table 7.33 Mass balance for mixed plastics recycling plant for 1,000 Mg of mixed plastics waste

Input	Quantity	Units	Output	Quantity	Units
mixed plastics	1,000	kg	plastic wood	980.392	kg
electricity	10,975.392	kJ	rejects	14.706	kg
			steel scrap	4.902	kg

The outputs from recycling unit is conducted to correct destination, *e.g.* rejects will be landfilled and metals will be conducted into recycling units. All the distances to this destinations have been gained from Google maps (2010) being considered the same diesel consumption (25 l/100 km) and load of 24 tonnes. It has also been modeled the treatment and landfill models.

7.1.2.14 Paper/cardboard recycling

Paper/cardboard are sending to processers industries which removes contaminants to be recycled in paper recycling industries. The mass balance of pre-processors are presented in Table 7.34, based on Rodrigo and Castells (2000).

Table 7.34 Mass balance for paper/cardboard waste pre-processor plant for 1,000 Mg of paper/cardboard waste

Input	Quantity	Units	Outputs	Quantity	Units
lube oil	0.0088	kg	wastepaper	971.2	kg
electricity	19,260	kJ	waste oil	0.0088	kg
diesel fuel	0.53568	kg	rejects	28.8	kg
steel	1.2	kg	steel scrap	1.2	kg
paper/cardboard waste	1,000	kg			

Then, paper/cardboard will be send to recycling plants in Portugal. The considered units which processed this material can cardboard, like companies CEMOPOL, Fábrica de Papel da Lapa, fábrica de Papel de Ponte Redonda, Portucel Viana, Prado Karton, Papeleira Portuguesa and Fábrica de Papel e Cartão da Zarrinha. For simplification, it was considered that paper/cardboard waste has been applied to produce brown kraftliner, since the major units in Portugal produces this product. The mass balance of paper/cardboard recycling is in Table 7.35 (APA, 2009, ProBas, 2004).

Table 7.35 Mass balance for wastepaper/cardboard recycling plant for 1,000 Mg of paper/cardboard waste

Inputs	Quantity	Units	Outputs	Quantity	Units
paper/cardboard	1,000	kg	brown kraftliner	1,000	kg
electricity	1,803,850	kJ	rejects	26.5	kg
starch	1.3	kg	sewage	1,100	kg
sodium hydroxide	7.8	kg	steam	1,500	kg
sulfuric acid	13.2	kg			
quicklime	82	kg			
water	2,600	kg			
heat energy	3,030,470	kJ			

7.1.2.15 Composites recycling

Composites waste are send to the same re-processers industries as paper/cardboard waste. After the bales have being sorted, composite packaging waste is send to Spain to be recycled, once Portugal does not have such units. The mass balance of the process is presented in Table 7.36 (the mass balance does only includes the recycling process), being the data from Stora Enso (2008).

Table 7.36 Mass balance of recycling composites to produce 1,000 kg of solid bleached board

Inpus	Quantity	Units	Outputs	Quantity	Units
latex	15.16109	kg	suspended solids (w)	44	kg
starch	13.26595	kg	chemical oxygen demand (COD) (w)	1	kg
steel	1.48	kg	solid bleached board (300g)	1,000	kg
limestone	140.8718	kg	electric energy	7,885,933	kJ
electric energy	2,052,000	kJ	hazardous waste	0.28	kg
heat energy	20,106,557	kJ	sewage sludge	287.9	kg
composites (paper, cardboards)	1,091.598	kg	waste, unspecified	1.49	kg
water (process)	8,700	kg	wood	0.2	kg
			steel scrap	1.48	kg
			sewage, clarified	5,400	kg
Note: (w) – emission to water					

7.1.2.16 High calorific fraction and RDF burning

For this study was considered that RDF would be burned in incineration plant, for energy recovery. The assumption was that RDF would not have quality to be used in cement plants. The incineration plantwould be the one located in São João da Talha.

Concerning the incineration, the LCI from RDF burning are presented in Table 7.37, from UBA (1999), Achernbosch and Richers (1997,1999), Schäfl (1995), being similar to the high calorific fraction direct burning mass balance.

Table 7.37 Mass balance for RDF burning for 1,000 Mg of RDF

Inputs	Quantity	Units	Outputs	Quantity	Units
ammonium hydroxide	0.000103	kg	ammonia (a)	1.54E-06	kg
precipitants	0.000925	kg	carbon dioxide. fossil (a)	0.000624	kg
calcium hydroxide	0.02124	kg	carbon dioxide. renewable (a)	0.000557	kg
coke	0.000257	kg	carbon monoxide (a)	5.14E-06	kg
RDF	1,000	kg	dinitrogen monoxide (a)	1.18E-07	kg
water (boiler feed)	20	kg	hydrogen chloride (a)	0.001004	kg
water (process)	180	kg	hydrogen fluoride (a)	0.000449	kg
			antimony (a)	4.5E-07	kg
			arsenic (a)	1.84E-07	kg
			cadmium (a)	1.27E-05	kg
			chromium (a)	6.08E-07	kg
			cobalt (a)	4.5E-07	kg
			copper (a)	1.46E-06	kg
			lead (a)	2.15E-06	kg
			manganese (a)	3.35E-06	kg
			mercury (a)	5.95E-07	kg
			nickel (a)	2.44E-07	kg
			thallium (a)	2.2E-07	kg
			tin (a)	1.34E-06	kg
			vanadium (a)	2.25E-07	kg
			zinc (a)	6.9E-06	kg
			NOx (a)	3.08E-05	kg
			sulfur dioxide (a)	0.002491	kg
			chlorobenzenes (a)	2.57E-11	kg
			chlorophenols (a)	5.14E-11	kg
			PCB (a)	2.57E-13	kg
			PCDD. PCDF (a)	2.57E-15	kg
			benzo(a)pyrene (a)	3.6E-13	kg
			TOC (a)	5.14E-07	kg
			exhaust gas. dry (a)	0.513599	Nm^3
			particles (a)	5.14E-07	kg
			electric energy	1604979	kJ

Inputs	Quantity	Units	Outputs	Quantity	Units
			flue gas cleaning residue	9.021531	kg
			slags and ash	6.810409	kg
			ashes and slags	106.6964	kg
			gypsum (flue gas clean.)	1.92E-06	kg
			scrap (iron)	1.377632	kg
			sewage (boiler elutriation)	20	kg
			steam	474	kg
Note: (a) – air emissions					

7.1.2.17 Compost application

The compost is to be used for agriculture purpose. The agriculture fields considered to use compost are in distance range of 75 km. The load to be transported is 24 tonnes and diesel consumption of 25 l/100 km.

Besides the benefits of compost application (nutrient source), heavy metals contained will also be applied in to soil and will occur the release of ammonia and N_2O, being the air emissions factors in Table 7.38, from den Boer *et al.* (2004).

Table 7.38 Air emissions factors from landspreading of compost

Parameter	Value	Units
ammonia	37	37% of NH4-N in compost
ammonia	4	4% of non NH4-N in compost
N_2O	1	1% of total N in compost

7.1.2.18 Auxiliary materials

The application of common modules like electricity, transport and heat/vapor production is based on the use of several LCI modules existing in UMBERTO library. According to REN (2008), the national mix includes 15% of electricity from Spain. For that reason is also necessary to include Spain mix for that year. In cases where units where located in Spain, it was used the mix from Spain, but also from Portugal and France.

Concerning natural gas, it was not possible to obtain the specific data for raw Algerian natural gas (before treatment). For that reason it has been used raw natural gas from Norway, since both have similar properties.

Concerning raw materials/processes it is also important to mention several measures which have been taken during the LCI. Road transportation, sanitary landfills, lube oil production, lube oil treatment, heat production from fuels, sodium hydroxide production, steel production, sulfuric acid production and limestone production where processes update to Portugal reference data, to be more adequate at national reality.

Some emission factors from road transportation have been updated by EMEP/CORINAIR data with emission factors from Portugal, for the indicated vehicles. Also emissions from heat/vapor energy were used EMEP/CORINAIR.

Concerning fuels or sources of energy, a brief comment is made to decisions made, which justifies their use. Concerning Portugal electricity, the energetic mix consists in percentage distribution of primary source in producing electricity at national net. Since such production is changeable, due to hydraulic field, the chosen mix in this study is referent to 2007. A detailed information of the references used in each auxiliary processes are presented in Table 7.39.

Table 7.39 References used in auxiliary processes

Material/process	Geography	Year	Technology	Reference
Electricity mix Portugal	Portugal	2000	Balance includes electricity production, fuels reservation, extraction, transport and previews chains from secundary materials	Fritsche *et al.* (1994)
Electricity mix Spain	Spain	2000		Fritsche *et al.* (1994)
Electricity mix France	França	2000		Fritsche *et al.* (1994)
Diesel productiom	Germany	early 90s	Crude oil extraction, transportation and refining	Frischknecht *et al.* (1996), Fritsche (2001), MEI (2008)
Light fuel oil	Germany	early 90s	Oil extraction, transport and production	Frischknecht *et al.* (1996), Fritsche (2001), MEI (2008)
Natural gas	Noruega	1994	Extraction, and transport of natural gas from Norway to Germany	Fritsche (1994), Frischknecht *et al.* (1994), MEI (2008)
Biomass	Germany	early 90s	Prodution of pine trunk wood, including energy and operating recources connected with wood production in forestry	ifeu (1994)
Diesel engine	Generic	early 90s	Diesel-powered engine without emission reduction	Fritsche *et al.* (2001), MEI (2008)
Heat production, by	Europe	early 90s	Heat production unit, 25 MW. The utilisation ratio	Fritsche *et al.* (2001), MEI

fueloil			is 85%	(2008)
Heat production by natural gas	Europe	early 90s	Heat production unit, 10 MW. The utilisation ratio is 90%	Fritsche *et al.* (2001), MEI (2008)
Heat production by natural gas	Europe	early 90s	Heat production unit, turbine,50 MW. The utilisation ratio is 35%	Fritsche *et al.* (2001), MEI (2008)
Electricity production by natural gas	Europe	early 90s	Gas turbine power station, 50 MW and an efficiency of 35 %	Fritsche *et al.* (2001), MEI (2008)
Heat production by biomass	Europe	early 90s	Heat production by wood chips. The utilisation ratio is 80%.	Fritsche *et al.* (2001), MEI (2008), EMEP/CORINAIR (2007, EMEP/EEA, 2009)
Sodium hydroxide	Europe	1994	Production of sodium hydroxide right from the removal of the raw materials from the natural resources, including the associated processes.	PlasticsEurope, 1994
Steel	Germany	90s	Production of liquid raw steel via the basic oxygen furnase method	Corradini and Köhler (1999), Rentz *et al.* (1996), ETH Zurich (1998), Fritsche (2000)
Lube oil	Europe and Portugal	90s and 2000	Production of lube oil	Martinho and Pires (2009)
Limestone	Germany	90s	Mining/quarrying and dressing of limestone	Patyk and Reinhardt (1997), BUWAL (1998)
Sulfuric acid	Germany	90s	From extraction into processes, including six different methods of producing sulfuric acid	Patyk and Reinhardt (1997)
Landfill for industrial waste (hazardous and non-hazardous)	Germany	1990-1997	-	Weber (1990a,b), Rettenberger (1997), Rettenberger and Schneider (1997), Heyer and Stegmann (1997), BUWAL (1996), Umweltbundesamt *et al.* (1997), Förstner *et al.* (1997), Kersten *et al.* (1995), Simon (1995), Regener *et al.* (1997), Hirschmann and Förstner (1997)
Road transport	Germany, Portugal	1997, 1999	Vehicle transportation	Borken *et al.* (1999), Knörr *et al.* (1997), Schmidt *et al.* (1998), EMEP/EEA (2009)

7.1.2.19 Substituted materials

Concerning substituted life cycles, the referent mass balances are also presented in Table 7.40-Table 7.53. The data used to model the substitute the following process has been based on: APA (2009), BUWAL (1998), ProBas (2004), Mata (1998), ifeu (1994), Patyk and Reinhardt (1997), PlasticsEurope (1994), Corradini and Köhler (1999), Rentz *et al.* (1996), ETH Zurich (1998), Fritsche (2000), Boustead (2000), Bannick *et al.* (2001), EVD (2001), Hydro Agri (2003), and ifu and ifeu (2001).

Table 7.40 Virgin aluminum production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
aluminium hydroxide	12.5	kg	ammonia (a)	0.023	kg
Árgon	1.5	kg	carbon dioxide. fossil (a)	10,634	kg
Chlorine	0.1	kg	carbon monoxide (a)	96	kg
Nitrogen	0.68	kg	dinitrogen monoxide (a)	0.0032	kg
sulfuric acid	31.4	kg	hydrogen chloride (a)	1.4	kg
oils. unspec.	0.08	kg	hydrogen fluoride (a)	0.75	kg
KEA (hydro)	35,212,235	kJ	mercury (a)	0.00022	kg
KEA (nuclear)	28,014,545	kJ	metals. unspec. (excl. Hg) (a)	0.5	kg
KEA. fossil total	1.12E+08	kJ	NOx (a)	27	kg
explosives	0.4	kg	sulfur dioxide (a)	71.6	kg
cast iron	4.3	kg	methane. fossil (a)	20	kg
steel bars	5.1	kg	NMVOC (HC excl. PAH) (a)	9.9	kg
aluminium oxide	0.4	kg	perfluoroethane (a)	0.028	kg
calcium fluoride	27.1	kg	perfluoromethane (a)	0.252	kg
Cryolite	1.6	kg	benzo(a)pyrene (a)	0.0032	kg
Soda	1.4	kg	PAH not B(a)P. unspec. (a)	0.0968	kg
acid. unspec.	8.7	kg	particles (a)	27	kg
carbon blocks	7.5	kg	dissolved solids (w)	1.7	kg
fibre materials	0.11	kg	acids as H(+) (w)	0.018	kg
packaging materials	1.8	kg	chloride (w)	56	kg
alloying additives	10.8	kg	cyanide (w)	0.00063	kg
collar / ramming paste	6.5	kg	mercury (w)	7E-06	kg
fluxing salts	0.4	kg	metals (excl. Hg) (w)	8.6	kg
organic components. unspec.	1.1	kg	pottassium (w)	0.089	kg
refractory material	15.2	kg	sodium (w)	9.3	kg
salts. unspec.	88.8	kg	ammonium (w)	0.06	kg
brown coal (r)	1328	kg	nitrate (w)	0.24	kg
hard coal (r)	1464	kg	phosphate (w)	0.17	kg
crude oil (r)	1369	kg	sulfate (w)	34	kg
natural gas (r)	485.7143	m^3	HC excl. PAH (w)	0.016	kg
bauxite (r)	4,111	kg	oil (w)	1.13	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
limestone (r)	159.4	kg	PAH. unspec. (w)	0.014	kg
potassium chloride (r)	3	kg	phenols (w)	0.0061	kg
cooling water	15200	kg	suspended solids (w)	4.4	kg
water (process)	870	kg	BOD-5 (w)	0.0024	kg
			COD (w)	0.23	kg
			fluorine (total) (a)	1.24	kg
			aluminium ingot	1000	kg
			bauxite residue (wfd)	1286	kg
			carbon waste (wfd)	3.9	kg
			dross fines (wfd)	0.11	kg
			filter dust (wfd)	2	kg
			hazardous waste (wfd)	0.32	kg
			Na as Na2O (wfd)	14.8	kg
			refractories (wfd)	7	kg
			sodium oxalate (wfd)	6	kg
			soot (wfd)	1.2	kg
			SPL carbon (wfd)	7.7	kg
			SPL refractory (wfd)	15.2	kg
			tar waste (wfd)	0.41	kg
			waste. unspecified (wfd)	64	kg
			carbon (wfr)	18.4	kg
			carbon for fuel (wfr)	3	kg
			crushed bath sold (wfr)	4.1	kg
			aluminium oxide (wfr)	0.4	kg
			steel scrap (wfr)	7	kg
			swarf/turnings (wfr)	0.84	kg
			sand (wfd)	98	kg
			skimmings and dross (wfr)	18.6	kg
			SPL carbon (wfr)	6.6	kg
			SPL refr. bricks (wfr)	1.7	kg
			sewage (cooling water)	15,200	kg

Table 7.41 Virgin brown kraftliner production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
sulfur dioxide	2.678	kg	ammonia (a)	0.063446	kg
KEA (hydro)	92829.95	kJ	carbon dioxide. fossil (a)	731.1468	kg
KEA (nuclear)	1650181	kJ	carbon dioxide. unspec. (a)	6.16	kg
KEA. fossil total	9582845	kJ	carbon monoxide (a)	0.456179	kg
KEA. renewable. others	21924334	kJ	dinitrogen monoxide (a)	0.019274	kg
KEA. unspec.	3596.631	kJ	hydrogen chloride (a)	0.050978	kg
carbon dioxide. fossil (a)	2265.83	kg	hydrogen fluoride (a)	0.007022	kg
pesticides. unspec.	0.089699	kg	hydrogen sulfide (a)	0.008	kg
packaging waste (wfr)	242	kg	arsenic (a)	1.67E-06	kg
land use C2 (FRG)	27.97321	m^2	Cadmium (a)	9.83E-07	kg
land use C3 (FRG)	314.6986	m^2	chromium (a)	2.77E-06	kg
land use C4 (FRG)	181.8258	m^2	nickel (a)	6.25E-05	kg
land use C5 (FRG)	102.5684	m^2	NOx (a)	2.214431	kg
aluminium sulfate	20	kg	sulfur dioxide (a)	2.834755	kg
Air	12.525	kg	methane (a)	1.272316	kg
auxiliary materials (soda production)	0.06	kg	NMVOC from diesel emission (a)	0.01535	kg
process materials (paper prod.)	5.8	kg	PCDD. PCDF (a)	1.47E-11	kg
brown coal (r)	165.4418	kg	perfluoromethane (a)	7.54E-10	kg
hard coal (r)	56.58717	kg	methylene oxide (a)	0.001795	kg
crude oil (r)	100.0778	kg	benzene (a)	0.000666	kg
natural gas (r)	74.39282	m^3	benzo(a)pyrene (a)	6.09E-08	kg
limestone (r)	37.78169	kg	PAH not B(a)P. unspec. (a)	2.68E-06	kg
potassium carbonate (r)	2.347387	kg	PAH. unspec. (a)	1.47E-07	kg
rock phosphate (r)	0.903512	kg	NMVOC. unspec. (a)	0.076215	kg
sodium chloride (r)	23.25	kg	VOC. unspec. (a)	0.003385	kg
sulfur (r)	0.060531	kg	particles (a)	0.417701	kg
Coke	1.185	kg	particles (small) (a)	0.011152	kg
commercial fertilizer (wfr)	104.1241	kg	Ca/Mg hydroxide (w)	0.39	kg
cooling water	11202.48	kg	calcium sulfate (w)	0.1575	kg
water (process)	11165.04	kg	chloride (w)	14.25	kg
			Fe/Al oxide (w)	0.075	kg
			hydroxide (w)	0.069	kg
			limestone (w)	1.17	kg
			calcium (w)	5.67	kg
			sodium (w)	2.865	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			ammonium as N (w)	0.0105	kg
			nitrate (w)	0.537975	kg
			nitrogen compounds as N (w)	3.73E-09	kg
			phosphorous compounds as P (w)	0.000426	kg
			salts. inorganic (w)	2.07E-08	kg
			sand (w)	0.1575	kg
			sulfate (w)	0.13755	kg
			suspended solids (w)	0.27	kg
			AOX (w)	4.14E-12	kg
			BOD-5 (w)	6.2E-06	kg
			BOD-7 (w)	0.5	kg
			COD (w)	5.3	kg
			kraftliner. brown	1000	kg
			hazardous waste (wfd)	0.014176	kg
			industrial waste (wfd)	30	kg
			sewage sludge (wfd)	0.001657	kg
			slags and ash (wfd)	10.91783	kg
			waste (soda production) (wfd)	0.33	kg
			ashes and slags (wfr)	6.21991	kg
			gypsum (flue gas clean.) (wfr)	3.986474	kg
			protein (wfr)	0.68	kg
			waste. unspec.	0.328223	kg
			sewage (cooling water)	10746.64	kg
			sewage (process)	11229.69	kg

Table 7.42 Virgin solid bleached board production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
barium chloride	0.20475	kg	ammonia (a)	0.083734	kg
hydrogen chloride	0.6825	kg	carbon dioxide, fossil (a)	575.8335	kg
Nitrogen	0.197069	kg	carbon dioxide, unspec. (a)	42.6991	kg
sodium chromate	0.0819	kg	carbon disulfide (a)	1.77E-09	kg
tetrasodiumpyrophosphate	0.0022	kg	carbon monoxide (a)	3.268204	kg
Xylene	0.008778	kg	chlorine (a)	2.52E-07	kg
KEA (hydro)	1658816	kJ	dinitrogen monoxide (a)	0.032162	kg
KEA (nuclear)	3431606	kJ	fluorine (a)	3.72E-08	kg
KEA, fossil total	7790890	kJ	hydrogen (a)	0.009007	kg
KEA, others	10227.34	kJ	hydrogen chloride (a)	0.009636	kg
KEA, renewable	1013.697	kJ	hydrogen cyanide (a)	1.19E-11	kg
KEA, renewable. others	37657656	kJ	hydrogen fluoride (a)	0.000469	kg
KEA, unspec.	22205.27	kJ	hydrogen sulfide (a)	0.011002	kg
carbon dioxide, unspec. (a)	5113.778	kg	arsenic (a)	4.45E-06	kg
cyclohexylpyrrolidone	0.00594	kg	Cadmium (a)	1.85E-06	kg
cyclosol 63	0.015158	kg	chromium (a)	1.15E-05	kg
ethylanthraquinone	0.0077	kg	lead (a)	1.51E-09	kg
pesticides, unspec.	0.094655	kg	mercury (a)	1.22E-05	kg
trioctyl phosphate	0.00594	kg	metals, unspec. (a)	3.27E-05	kg
land use C2 (NORD)	26.56508	m^2	nickel (a)	0.000116	kg
land use C3 (NORD)	298.8571	m^2	NOx (a)	2.900283	kg
land use C4 (NORD)	863.3651	m^2	sulfur dioxide (a)	2.329219	kg
land use C5 (NORD)	194.8106	m^2	sulfuric acid (a)	1.41E-11	kg
Sulfur	5.71131	kg	methane (a)	0.414244	kg
Air	128.2914	kg	methane, fossil (a)	0.57454	kg
feedstocks. diverse	0.22605	kg	NMVOC from diesel emission (a)	0.025795	kg
brown coal (r)	23.24812	kg	NMVOC, aromat., unspec. (a)	0.001108	kg
hard coal (r)	49.93108	kg	PCDD, PCDF (a)	1.91E-11	kg
crude oil (r)	113.836	kg	NMVOC. chlor., unspec. (a)	1.6E-06	kg
natural gas (r)	45.39242	m^3	NMVOC. fluor., unspec. (a)	3.46E-06	kg
biomass. unspec. (r)	113.1	kg	perfluoroethane (a)	1.43E-08	kg
barite (r)	0.004631	kg	perfluoromethane (a)	1.54E-07	kg
bauxite (r)	0.004908	kg	aldehydes. unspec. (a)	5.77E-09	kg
bentonite (r)	0.001083	kg	methylene oxide (a)	0.002698	kg
limestone (r)	34.91446	kg	benzene (a)	0.001828	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
potassium carbonate (r)	2.309829	kg	ethane thiol (a)	4.96E-07	kg
potassium chloride (r)	0.3087	kg	benzo(a)pyrene (a)	4.42E-07	kg
rock phosphate (r)	0.889055	kg	PAH. unspec. (a)	3.28E-05	kg
sodium chloride (r)	26.58423	kg	NMVOC, unspec. (a)	0.083081	kg
iron (Fe) (r)	0.006169	kg	VOC (hydrocarbons) (a)	0.000463	kg
sulfur (r)	7.135153	kg	VOC, unspec. (a)	0.032308	kg
soil (r)	427.44	kg	particles (a)	0.958393	kg
commercial fertilizer (wfr)	102.4581	kg	particles (small) (a)	0.022464	kg
cooling water	986.8575	kg	dissolved solids (w)	0.22526	kg
water (process)	105134.2	kg	acids as H(+) (w)	0.001501	kg
			Ca/Mg hydroxide (w)	0.002626	kg
			calcium sulfate (w)	0.001061	kg
			carbonate (w)	0.002286	kg
			chlorate (w)	0.1	kg
			chloride (w)	0.838006	kg
			chlorine, dissolved (w)	2.38E-05	kg
			cyanide (w)	8.08E-08	kg
			Fe/Al oxide (w)	0.000505	kg
			fluoride (w)	1.35E-06	kg
			hydroxide (w)	0.000465	kg
			limestone (w)	0.007879	kg
			aluminium (w)	0.000508	kg
			arsenic (w)	1.91E-09	kg
			calcium (w)	0.039377	kg
			chromium (VI) oxide (w)	1.73E-10	kg
			copper (w)	1.12E-06	kg
			iron (w)	5.03E-07	kg
			lead (w)	5.25E-09	kg
			magnesium (w)	2.38E-05	kg
			mercury (w)	1.84E-08	kg
			metals, unspec. (w)	0.002341	kg
			nickel (w)	1.11E-06	kg
			pottassium (w)	0.009142	kg
			sodium (w)	0.592062	kg
			zinc (w)	2.43E-07	kg
			ammonium (w)	3.98E-05	kg
			ammonium as N (w)	7.07E-05	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			nitrate (w)	0.529382	kg
			nitrogen compounds as N (w)	8.36E-05	kg
			nitrogen compounds. unspec. (w)	5.98E-05	kg
			phosphate (w)	3.21E-06	kg
			phosphorous compounds as P (w)	0.000423	kg
			sand (w)	0.001061	kg
			sulfate (w)	0.037939	kg
			sulfur (w)	3.06E-06	kg
			detergents. oil (w)	0.000301	kg
			dissolved organics (w)	0.000152	kg
			chlorinated, org. compounds., unspec. (w)	7.12E-05	kg
			HC. unspec. (w)	0.000518	kg
			phenols (w)	4.99E-05	kg
			organic compounds (w)	9.09E-06	kg
			suspended solids (w)	0.165129	kg
			AOX (w)	0.2	kg
			BOD-5 (w)	2.700266	kg
			COD (w)	28.60457	kg
			solid bleached board (300g)	1000	kg
			hazardous waste (wfd)	0.038199	kg
			industrial waste (wfd)	72.22365	kg
			mineral waste (wfd)	291.1072	kg
			plastics, unspec. (wfd)	0.005472	kg
			slags and ash (wfd)	0.509853	kg
			waste for incineration (wfd)	0.019693	kg
			waste, inert (chemical industry) (wfd)	0.064448	kg
			ashes and slags (wfr)	8.15E-09	kg
			gypsum (flue gas clean.) (wfr)	0.106621	kg
			metals, unspec. (wfr)	0.000129	kg
			waste, unspec.	2.288752	kg
			sewage (cooling water)	987.1092	kg
			sewage (process)	105104.6	kg

Table 7.43 Virgin EPS production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
Nitrogen	198.1295	kg	ammonia (a)	8.91E-06	kg
Oxygen	1.663672	kg	asbestos (a)	1.17E-11	kg
phosphate (as P2O5)	4.3E-08	kg	carbon dioxide. fossil (a)	2713.063	kg
KEA (hydro)	173430.1	kJ	carbon dioxide. renewable (a)	0.052958	kg
KEA (nuclear)	2139735	kJ	carbon disulfide (a)	3.94E-06	kg
KEA. fossil total	77711545	kJ	carbon monoxide (a)	5.603525	kg
KEA. others	296036.5	kJ	chlorine (a)	0.000905	kg
KEA. renewable. others	72446.53	kJ	dinitrogen monoxide (a)	1.93E-05	kg
KEA. unspec.	392.2908	kJ	fluorine (a)	4.26E-05	kg
carbon dioxide. renewable (a)	0.052958	kg	hydrogen (a)	0.052074	kg
Sulfur	0.001743	kg	hydrogen chloride (a)	0.05056	kg
Air	457.1474	kg	hydrogen cyanide (a)	1.99E-15	kg
brown coal (r)	0.041089	kg	hydrogen fluoride (a)	0.001886	kg
hard coal (r)	153.7271	kg	hydrogen sulfide (a)	5.3E-05	kg
crude oil (r)	1023.855	kg	antimony (a)	1.3E-07	kg
natural gas (r)	853.2254	m^3	arsenic (a)	9.5E-06	kg
wood (r)	0.049041	kg	Cadmium (a)	1.18E-06	kg
barite (r)	0.001785	kg	chromium (a)	0.001581	kg
bauxite (r)	0.641909	kg	copper (a)	7.9E-05	kg
bentonite (r)	0.088045	kg	lead (a)	0.000258	kg
calcium sulfate (r)	0.008881	kg	mercury (a)	1.83E-06	kg
chalk (r)	5.12E-28	kg	metals. unspec. (a)	0.002721	kg
clay (r)	0.000107	kg	nickel (a)	0.002874	kg
dolomite (r)	0.019272	kg	selenium (a)	3.29E-08	kg
feldspar (r)	2.5E-13	kg	silver (Ag) (a)	9.49E-07	kg
fluorite (r)	0.012545	kg	zinc (a)	3.75E-05	kg
granite (r)	5.42E-10	kg	NOx (a)	5.320642	kg
gravel (r)	0.005813	kg	sulfur dioxide (a)	7.319361	kg
limestone (r)	0.639503	kg	sulfuric acid (a)	5.84E-12	kg
olivine (r)	0.014783	kg	methane (a)	30.39522	kg
potassium chloride (r)	0.005338	kg	NMVOC. aromat.. unspec. (a)	0.03129	kg
quartz (SiO2) (r)	1.74E-18	kg	dichloroethane (a)	3E-06	kg
rutil (r)	7.35E-28	kg	dichloromethane (a)	2.58E-06	kg
sand (r)	0.352388	kg	vinyl chloride (a)	1.02E-05	kg
shale (r)	0.025144	kg	PCDD. PCDF (a)	8.54E-38	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
sodium chloride (r)	1.951178	kg	NMVOC. chlor.. unspec. (a)	0.000487	kg
sodium nitrate (NaNO3) (RiL)	2.63E-06	kg	R 22 (a)	1.49E-05	kg
talcum (r)	9.44E-29	kg	NMVOC. fluor.. unspec. (a)	1.58E-05	kg
chromium (Cr) (r)	2.17E-05	kg	aldehydes. unspec. (a)	1.27E-10	kg
copper (Cu) (r)	0.175354	kg	ethene (a)	0.006847	kg
ferromanganese (r)	0.001432	kg	propene (a)	0.005071	kg
iron (Fe) (r)	1.575679	kg	benzene (a)	0.022648	kg
lead (Pb) (r)	0.009916	kg	ethyl benzene (a)	0.040283	kg
magnesium (Mg) (r)	8.76E-07	kg	styrene (a)	0.074503	kg
mercury (Hg) (r)	7.14E-06	kg	toluene (a)	0.004253	kg
nickel (Ni) (r)	0.0142	kg	xylene (a)	0.001048	kg
zinc (Zn) (r)	0.092216	kg	ethane thiol (a)	1.73E-05	kg
sulfur (r)	0.145718	kg	PAH. unspec. (a)	0.002874	kg
peat (r)	0.118312	kg	NMVOC. unspec. (a)	0.045837	kg
biomass (kg)	7.170997	kg	VOC. unspec. (a)	2.952957	kg
industrial waste (wfd)	2.097123	kg	particles (PM10) (a)	0.90149	kg
cooling water	131384.2	kg	dissolved solids (w)	0.213042	kg
Wasser (Prozess) (Trinkwasser)	1193.217	kg	acids as H(+) (w)	0.005378	kg
water (process)	7981.635	kg	bromate (w)	2.77E-06	kg
			carbonate (w)	0.115698	kg
			chlorate (w)	0.000481	kg
			chloride (w)	0.343786	kg
			chlorine. dissolved (w)	1.78E-05	kg
			cyanide (w)	1.5E-07	kg
			fluoride (w)	0.000337	kg
			aluminium (w)	0.001052	kg
			arsenic (w)	8.36E-07	kg
			Cadmium (w)	9.76E-08	kg
			calcium (w)	0.01262	kg
			chromium (VI) (w)	1.62E-08	kg
			copper (w)	0.00174	kg
			iron (w)	0.000157	kg
			lead (w)	3.16E-06	kg
			magnesium (w)	0.000163	kg
			manganese (w)	1.04E-06	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			mercury (w)	1.93E-07	kg
			metals. unspec. (w)	0.112647	kg
			molybdenum (w)	2.13E-05	kg
			nickel (w)	0.00155	kg
			pottassium (w)	0.000326	kg
			sodium (w)	0.210349	kg
			strontium (w)	3.95E-08	kg
			zinc (w)	4.94E-05	kg
			ammonium (w)	0.012796	kg
			nitrate (w)	0.006987	kg
			nitrogen compounds. unspec. (w)	0.003511	kg
			phosphorous compounds as P (w)	0.003363	kg
			sulfate (w)	0.403714	kg
			sulfite (w)	0.001221	kg
			sulfur (w)	0.000169	kg
			detergents. oil (w)	0.024126	kg
			dissolved organics (w)	0.008931	kg
			1.2-dichloroethane (w)	4.68E-08	kg
			vinyl chloride (w)	1.9E-07	kg
			PCDD. PCDF (w)	9.92E-10	kg
			chlorinated. org. compounds.. unspec. (w)	2.41E-05	kg
			benzene (w)	0.001026	kg
			HC. unspec. (w)	0.015617	kg
			phenols (w)	0.000498	kg
			organic compounds (w)	2.58E-06	kg
			organo-silicon compounds (w)	1.24E-16	kg
			organo-tin compounds (w)	1.3E-09	kg
			suspended solids (w)	0.272139	kg
			AOX (w)	3.36E-08	kg
			BOD-5 (w)	0.048539	kg
			COD (w)	0.384532	kg
			TOC (w)	0.041934	kg
			polystyrene (GGPS)	1000	kg
			Abfälle (Grubenverfüllung) (AzB)	45.03447	kg
			Abfälle (ungeregelte Chemikalien) (AzB)	2.910292	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			hazardous waste (wfd)	4.082713	kg
			industrial waste (wfd)	1.833692	kg
			mineral waste (wfd)	9.530745	kg
			paper. cardboard (wfd)	0.022333	kg
			plastics. unspec. (wfd)	0.070205	kg
			slags and ash (wfd)	10.17719	kg
			waste for incineration (wfd)	24.60824	kg
			waste. inert (chemical industry) (wfd)	2.692885	kg
			waste. unspecified (wfd)	2.014928	kg
			metals. unspec. (wfr)	0.095455	kg
			mixed valuable materials (wfr)	0.31547	kg
			plastic containers (wfr)	0.000973	kg
			sewage (cooling water)	131384.2	kg
			sewage (process)	9174.852	kg

Table 7.44 Fertilizer Ca prodution – 1000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	249.9092	kJ	ammonia (a)	0.010487	kg
KEA (nuclear)	274702	kJ	carbon dioxide, fossil (a)	285.562	kg
KEA. fossil total	27070210	kJ	carbon monoxide (a)	5.609012	kg
KEA. others	243283.6	kJ	dinitrogen monoxide (a)	0.189357	kg
KEA. renewable	306526.6	kJ	fluorine (a)	3.35E-09	kg
KEA. renewable. others	4485.059	kJ	hydrogen chloride (a)	0.046002	kg
KEA. unspec.	0.996134	kJ	hydrogen fluoride (a)	0.003531	kg
brown coal (r)	27.3041	kg	hydrogen sulfide (a)	3.52E-07	kg
hard coal (r)	37.34524	kg	antimony (a)	4.53E-08	kg
crude oil (r)	618.0778	kg	arsenic (a)	1.32E-05	kg
natural gas (r)	31.27614	m^3	beryllium (a)	5.42E-08	kg
uranium (r)	0.000256	kg	Cadmium (a)	3.2E-05	kg
limestone (r)	2645.224	kg	chromium (a)	1.61E-05	kg
sand (r)	0.00973	kg	cobalt (a)	3.93E-08	kg
sodium chloride (r)	0.002767	kg	copper (a)	2.03E-07	kg
sulfur (r)	0.001807	kg	lead (a)	5.88E-07	kg
cooling water	8844.474	kg	manganese (a)	1.97E-07	kg
water (boiler feed)	2127.201	kg	mercury (a)	5.79E-07	kg
water (process)	3.695733	kg	nickel (a)	0.001299	kg
water. unspec.	733.599	kg	selenium (a)	3.16E-06	kg
			thallium (a)	8.41E-09	kg
			tin (a)	1.52E-07	kg
			uranium (a)	1.55E-07	kg
			vanadium (a)	4.35E-08	kg
			zinc (a)	1.48E-06	kg
			NOx (a)	18.73163	kg
			radionuclides. total (a)	15622413	Bq
			sulfur (a)	2.35E-08	kg
			sulfur dioxide (a)	2.360828	kg
			methane (a)	0.958628	kg
			NMVOC (hydrocarbons) (a)	0.00443	kg
			chlorobenzenes (a)	2.68E-15	kg
			chlorophenols (a)	5.36E-15	kg
			PCB (a)	2.68E-17	kg
			PCDD. PCDF (a)	4.63E-11	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			perfluoroethane (a)	1.36E-08	kg
			perfluoromethane (a)	1.01E-07	kg
			methylene oxide (a)	0.114912	kg
			hexane (a)	4.25E-06	kg
			benzene (a)	0.03168	kg
			benzo(a)pyrene (a)	4.11E-06	kg
			PAH not B(a)P. unspec. (a)	2.5E-07	kg
			NMVOC. unspec. (a)	0.890991	kg
			particles (>PM10) (a)	0.132422	kg
			particles (a)	139.1995	kg
			particles (PM10) (a)	0.208748	kg
			particles (small) (a)	0.531678	kg
			waste heat (a)	220745.4	kJ
			Cadmium (s)	0.00042	kg
			chromium (s)	0.015944	kg
			copper (s)	0.011888	kg
			lead (s)	0.008531	kg
			mercury (s)	1.4E-05	kg
			nickel (s)	0.006713	kg
			zinc (s)	0.064336	kg
			boron (w)	2.93E-09	kg
			chloride (w)	0.001965	kg
			chlorine (w)	3.16E-05	kg
			cyanide (w)	2.76E-09	kg
			fluoride (w)	4.77E-06	kg
			fluorine (w)	2.94E-07	kg
			aluminium (w)	8.12E-07	kg
			antimony (w)	3.62E-11	kg
			arsenic (w)	1.01E-07	kg
			barium (w)	4.8E-07	kg
			beryllium (w)	6.32E-09	kg
			Cadmium (w)	2.54E-08	kg
			chromium (w)	4.41E-07	kg
			cobalt (w)	1.52E-09	kg
			copper (w)	1.02E-07	kg
			lead (w)	1.08E-05	kg
			manganese (w)	2.11E-05	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			mercury (w)	4.58E-10	kg
			molybdenum (w)	2.63E-06	kg
			nickel (w)	2.18E-07	kg
			selenium (w)	7.86E-07	kg
			tin (w)	1.54E-09	kg
			uranium (w)	3.49E-06	kg
			vanadium (w)	2.03E-06	kg
			zinc (w)	6.27E-07	kg
			ammonium (w)	5.19E-05	kg
			nitrate (w)	5.87E-06	kg
			nitric acid (w)	4.94E-07	kg
			nitrogen compounds as N (w)	1.12E-06	kg
			phosphorous compounds as P (w)	6.65E-08	kg
			radionuclides. total (w)	93021.35	kBq
			salts. inorganic (w)	1.17E-06	kg
			sulfate (w)	0.021976	kg
			sulfide (w)	3.21E-08	kg
			PCB (w)	3.55E-12	kg
			HC. unspec. (w)	3.33E-08	kg
			oil (w)	1.23E-12	kg
			benzo(a)pyrene (w)	2.17E-13	kg
			PAH excl. B(a)P (w)	1.95E-13	kg
			phenols (w)	4.06E-09	kg
			waste heat (w)	48755.94	kJ
			AOX (w)	5.03E-09	kg
			BOD-5 (w)	1.39E-06	kg
			COD (w)	1.42E-05	kg
			TOC (w)	4.96E-05	kg
			landfill volume	0.000155	m^3
			nucl.waste. high-radioactive final disp.	1.52E-07	m^3
			nucl.waste. low-radioactive. final disp.	2.3E-07	m^3
			nucl.waste. med-radioactive. final disp.	4.23E-08	m^3
			calcium (Ca) (r)	1000	kg
			hazardous waste (wfd)	0.000132	kg
			mineral waste (wfd)	329.1256	kg
			radioactive waste (high-radioactive) (wfd)	1.91E-08	m^3

Inputs	Quantity	Units	Outputs	Quantity	Units
			sewage sludge (wfd)	0.008618	kg
			slags and ash (wfd)	0.083046	kg
			waste. unspecified (wfd)	0.069858	kg
			ashes (fluidized-bed incinerator) (wfr)	0.008326	kg
			ashes and slags (wfr)	10.41522	kg
			coarse ashes (wfr)	0.012445	kg
			filter dust (wfr)	0.083258	kg
			gypsum (flue gas clean.) (wfr)	1.222151	kg
			melting chamber granulate (wfr)	0.104345	kg
			sodium sulphate (wfr)	0.002731	kg
			waste. unspec.	0.009527	kg
			seepage water. collected	0.000854	kg
			seepage water. diffuse	0.004203	kg
			sewage (boiler elutriation)	2127.201	kg
			sewage (cooling water)	3465.015	kg
			sewage (process)	0.396341	kg
			sewage. clarified	0.035433	kg
			sewage. unspec.	0.005458	kg
			steam	5391.109	kg

Table 7.45 Fertilizer K prodution – 1000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	238786.8	kJ	ammonia (a)	0.008056	kg
KEA (nuclear)	1564399	kJ	carbon dioxide. fossil (a)	2256.383	kg
KEA. fossil total	35157961	kJ	carbon monoxide (a)	2.122975	kg
brown coal (r)	173.1381	kg	dinitrogen monoxide (a)	0.186004	kg
hard coal (r)	122.3674	kg	hydrogen chloride (a)	0.224699	kg
crude oil (r)	220.2544	kg	arsenic (a)	2.77E-06	kg
natural gas (r)	624.2549	m^3	Cadmium (a)	6.94E-06	kg
potassium carbonate (r)	31818.18	kg	chromium (a)	3.45E-06	kg
			nickel (a)	0.000281	kg
			NOx (a)	7.43206	kg
			sulfur dioxide (a)	1.155804	kg
			methane (a)	4.265504	kg
			PCDD. PCDF (a)	7.68E-10	kg
			methylene oxide (a)	0.049425	kg
			benzene (a)	0.013673	kg
			benzo(a)pyrene (a)	1.5E-06	kg
			NMVOC. unspec. (a)	0.546463	kg
			particles (a)	2.586408	kg
			particles (small) (a)	0.258403	kg
			Cadmium (s)	0.000303	kg
			chromium (s)	0.010606	kg
			copper (s)	0.008788	kg
			lead (s)	0.001515	kg
			mercury (s)	6.06E-05	kg
			nickel (s)	0.004545	kg
			zinc (s)	0.011212	kg
			K-fertlilizer	1000	kg

Table 7.46 Fertilizer Mg prodution – 1000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	1178.127	kJ	ammonia (a)	0.00345	kg
KEA (nuclear)	709140.6	kJ	carbon dioxide. fossil (a)	957.8294	kg
KEA. fossil total	8317889	kJ	carbon monoxide (a)	8.5295	kg
KEA. renewable. others	21272.42	kJ	dinitrogen monoxide (a)	0.079041	kg
brown coal (r)	127.9941	kg	fluorine (a)	1.59E-08	kg
hard coal (r)	25.04969	kg	hydrogen chloride (a)	0.024895	kg
crude oil (r)	134.4086	kg	hydrogen fluoride (a)	0.000508	kg
natural gas (r)	32.56729	m^3	hydrogen sulfide (a)	8.29E-07	kg
uranium (r)	0.001214	kg	antimony (a)	2.15E-07	kg
limestone (r)	3391.877	kg	arsenic (a)	4.94E-06	kg
sand (r)	0.046147	kg	beryllium (a)	2.57E-07	kg
sodium chloride (r)	0.013124	kg	Cadmium (a)	7.5E-06	kg
sulfur (r)	0.008572	kg	chromium (a)	4.55E-06	kg
cooling water	39932.96	kg	cobalt (a)	1.86E-07	kg
water (boiler feed)	10089.21	kg	copper (a)	9.62E-07	kg
water (process)	17.52868	kg	lead (a)	2.79E-06	kg
water. unspec.	0.771363	kg	manganese (a)	9.35E-07	kg
			mercury (a)	2.75E-06	kg
			nickel (a)	0.000306	kg
			selenium (a)	1.5E-05	kg
			thallium (a)	3.99E-08	kg
			tin (a)	7.23E-07	kg
			uranium (a)	7.37E-07	kg
			vanadium (a)	2.06E-07	kg
			zinc (a)	7.02E-06	kg
			NOx (a)	4.339704	kg
			radionuclides. total (a)	74096352	Bq
			sulfur (a)	1.11E-07	kg
			sulfur dioxide (a)	0.458429	kg
			methane (a)	0.598683	kg
			NMVOC (hydrocarbons) (a)	0.021012	kg
			chlorobenzenes (a)	1.27E-14	kg
			chlorophenols (a)	2.54E-14	kg
			PCB (a)	1.27E-16	kg
			PCDD. PCDF (a)	6.93E-11	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			perfluoroethane (a)	4.46E-09	kg
			perfluoromethane (a)	5.81E-11	kg
			methylene oxide (a)	0.026063	kg
			hexane (a)	2.01E-05	kg
			benzene (a)	0.007244	kg
			benzo(a)pyrene (a)	9.17E-07	kg
			PAH not B(a)P. unspec. (a)	1.19E-06	kg
			NMVOC. unspec. (a)	0.195058	kg
			particles (>PM10) (a)	0.628072	kg
			particles (a)	0.020085	kg
			particles (PM10) (a)	0.990083	kg
			particles (small) (a)	0.115714	kg
			waste heat (a)	1046985	kJ
			Cadmium (s)	0.000166	kg
			chromium (s)	0.011443	kg
			copper (s)	0.004975	kg
			lead (s)	0.001658	kg
			mercury (s)	1.66E-05	kg
			nickel (s)	0.001658	kg
			zinc (s)	0.015755	kg
			boron (w)	1.39E-08	kg
			chloride (w)	0.00932	kg
			chlorine (w)	0.00015	kg
			cyanide (w)	1.31E-08	kg
			fluoride (w)	2.26E-05	kg
			fluorine (w)	1.39E-06	kg
			aluminium (w)	3.85E-06	kg
			antimony (w)	1.72E-10	kg
			arsenic (w)	4.79E-07	kg
			barium (w)	2.28E-06	kg
			beryllium (w)	3E-08	kg
			Cadmium (w)	1.21E-07	kg
			chromium (w)	2.09E-06	kg
			cobalt (w)	7.22E-09	kg
			copper (w)	4.84E-07	kg
			lead (w)	5.12E-05	kg
			manganese (w)	0.0001	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			mercury (w)	2.17E-09	kg
			molybdenum (w)	1.25E-05	kg
			nickel (w)	1.04E-06	kg
			selenium (w)	3.73E-06	kg
			tin (w)	7.32E-09	kg
			uranium (w)	1.65E-05	kg
			vanadium (w)	9.63E-06	kg
			zinc (w)	2.97E-06	kg
			ammonium (w)	0.000246	kg
			nitrate (w)	2.78E-05	kg
			nitric acid (w)	2.34E-06	kg
			nitrogen compounds as N (w)	5.21E-06	kg
			phosphorous compounds as P (w)	3.16E-07	kg
			radionuclides. total (w)	441195.8	kBq
			salts. inorganic (w)	3.36E-08	kg
			sulfate (w)	0.10423	kg
			sulfide (w)	1.52E-07	kg
			PCB (w)	1.68E-11	kg
			HC. unspec. (w)	1.58E-07	kg
			oil (w)	5.84E-12	kg
			benzo(a)pyrene (w)	1.03E-12	kg
			PAH excl. B(a)P (w)	9.25E-13	kg
			phenols (w)	1.92E-08	kg
			waste heat (w)	231247.1	kJ
			AOX (w)	2.37E-08	kg
			BOD-5 (w)	5.7E-06	kg
			COD (w)	4.54E-05	kg
			TOC (w)	0.000235	kg
			landfill volume	0.000735	m^3
			nucl.waste. high-radioactive final disp.	7.19E-07	m^3
			nucl.waste. low-radioactive. final disp.	1.09E-06	m^3
			nucl.waste. med-radioactive. final disp.	2.01E-07	m^3
			magnesium (Mg) (r)	1000	kg
			hazardous waste (wfd)	0.000496	kg
			mineral waste (wfd)	1108.707	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			sewage sludge (wfd)	0.001949	kg
			slags and ash (wfd)	0.261658	kg
			waste. unspecified (wfd)	0.331334	kg
			ashes (fluidized-bed incinerator) (wfr)	0.039489	kg
			coarse ashes (wfr)	0.059027	kg
			filter dust (wfr)	0.39489	kg
			gypsum (flue gas clean.) (wfr)	1.43872	kg
			melting chamber granulate (wfr)	0.494903	kg
			sodium sulphate (wfr)	0.012952	kg
			seepage water. collected	0.004051	kg
			seepage water. diffuse	0.019935	kg
			sewage (boiler elutriation)	10089.21	kg
			sewage (cooling water)	14418.45	kg
			sewage (process)	1.879824	kg
			sewage. clarified	0.168058	kg
			steam	25569.77	kg

Table 7.47 Fertilizer N prodution – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	164526.8	kJ	ammonia (a)	46.20651	kg
KEA (nuclear)	960616.9	kJ	carbon dioxide. fossil (a)	13303.59	kg
KEA. fossil total	2.27E+08	kJ	carbon monoxide (a)	10.86392	kg
KEA. others	730443.2	kJ	dinitrogen monoxide (a)	34.29695	kg
KEA. renewable	920326.4	kJ	hydrogen chloride (a)	0.184759	kg
KEA. unspec.	0.788506	kJ	hydrogen fluoride (a)	0.010282	kg
brown coal (r)	33.40651	kg	hydrogen sulfide (a)	5.31E-07	kg
hard coal (r)	194.139	kg	arsenic (a)	1.26E-05	kg
crude oil (r)	846.018	kg	cadmium (a)	3.16E-05	kg
natural gas (r)	5506.985	m^3	chromium (a)	1.57E-05	kg
limestone (r)	2037.048	kg	nickel (a)	0.001281	kg
cooling water	1275.857	kg	NOx (a)	43.2773	kg
water. unspec.	580.5633	kg	sulfur dioxide (a)	8.797493	kg
			methane (a)	17.12539	kg
			PCDD. PCDF (a)	1.22E-09	kg
			perfluoroethane (a)	3.81E-08	kg
			perfluoromethane (a)	3.03E-07	kg
			methylene oxide (a)	0.132835	kg
			benzene (a)	0.038182	kg
			benzo(a)pyrene (a)	4.73E-06	kg
			NMVOC. unspec. (a)	2.238844	kg
			particles (a)	112.7525	kg
			particles (small) (a)	0.614712	kg
			cadmium (s)	1.111111	kg
			chromium (s)	32.22222	kg
			copper (s)	14.81481	kg
			lead (s)	79.25926	kg
			mercury (s)	0.074074	kg
			nickel (s)	14.07407	kg
			zinc (s)	141.8519	kg
			nitrogen compounds as N (w)	6.55E-08	kg
			salts. inorganic (w)	3.49E-06	kg
			AOX (w)	8.63E-11	kg
			BOD-5 (w)	5.55E-07	kg
			COD (w)	1.39E-05	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			N-fertilizer (as N)	1000	kg
			calcium (Ca) (r)	444.4444	kg
			magnesium (Mg) (r)	88.88889	kg
			hazardous waste (wfd)	2.17E-05	kg
			mineral waste (wfd)	286.3324	kg
			radioactive waste (high-radioactive) (wfd)	5.74E-08	m^3
			sewage sludge (wfd)	0.024641	kg
			slags and ash (wfd)	0.022068	kg
			ashes and slags (wfr)	31.2703	kg
			gypsum (flue gas clean.) (wfr)	2.758467	kg
			waste. unspec.	0.028604	kg
			sewage (cooling water)	1275.857	kg
			sewage. unspec.	0.00432	kg

Table 7.48 Fertilizer P production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	867854.3	kJ	ammonia (a)	0.049512	kg
KEA (nuclear)	12909389	kJ	carbon dioxide. fossil (a)	22344.39	kg
KEA. fossil total	3.17E+08	kJ	carbon monoxide (a)	19.37365	kg
KEA. others	18440038	kJ	dinitrogen monoxide (a)	0.957095	kg
KEA. renewable	23233640	kJ	hydrogen chloride (a)	3.11907	kg
Sulfur	2531.25	kg	hydrogen fluoride (a)	0.259562	kg
brown coal (r)	238.0255	kg	hydrogen sulfide (a)	1.34E-05	kg
hard coal (r)	2594.287	kg	arsenic (a)	1.3E-05	kg
crude oil (r)	1994.658	kg	cadmium (a)	3.27E-05	kg
natural gas (r)	4803.776	m^3	chromium (a)	1.62E-05	kg
rock phosphate (r)	50750	kg	nickel (a)	0.001324	kg
sulfur (r)	850	kg	NOx (a)	92.56168	kg
cooling water	32206.26	kg	sulfur dioxide (a)	96.64373	kg
			methane (a)	44.14795	kg
			PCDD. PCDF (a)	7.74E-10	kg
			perfluoroethane (a)	9.63E-07	kg
			perfluoromethane (a)	7.66E-06	kg
			methylene oxide (a)	0.216809	kg
			benzene (a)	0.058243	kg
			benzo(a)pyrene (a)	7.22E-06	kg
			NMVOC. unspec. (a)	5.082136	kg
			particles (a)	14.9293	kg
			particles (small) (a)	2.123041	kg
			cadmium (s)	0.135	kg
			chromium (s)	1.425	kg
			copper (s)	0.215	kg
			lead (s)	0.23125	kg
			mercury (s)	0.00025	kg
			nickel (s)	0.36	kg
			zinc (s)	2.95	kg
			nitrogen compounds as N (w)	1.65E-06	kg
			salts. inorganic (w)	8.81E-05	kg
			AOX (w)	2.18E-09	kg
			BOD-5 (w)	1.4E-05	kg
			COD (w)	0.000351	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			P-fertilizer	1000	kg
			mineral waste (wfd)	7228.462	kg
			radioactive waste (high-radioactive) (wfd)	1.45E-06	m^3
			sewage sludge (wfd)	0.622065	kg
			ashes and slags (wfr)	789.4122	kg
			gypsum (flue gas clean.) (wfr)	69.63563	kg
			waste, unspec.	0.7221	kg
			sewage (cooling water)	32206.26	kg

Table 7.49 Plastic wood production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA. fossil total	1443081	kJ	ammonia (a)	0.000578	kg
KEA. renewable. others	13329333	kJ	carbon dioxide. fossil (a)	106.7086	kg
pesticides. unspec.	0.002587	kg	carbon monoxide (a)	0.251528	kg
limestone	0.173333	kg	dinitrogen monoxide (a)	0.009715	kg
materials. unspec.	166.6667	kg	hydrogen chloride (a)	3.51E-05	kg
oil (running material)	0.234667	kg	hydrogen fluoride (a)	4.51E-07	kg
crude oil (r)	36.06466	kg	arsenic (a)	7.6E-07	kg
wood (r)	1025.333	kg	cadmium (a)	1.9E-06	kg
electric energy	96096000	kJ	chromium (a)	9.46E-07	kg
			nickel (a)	7.72E-05	kg
			NOx (a)	1.085467	kg
			sulfur dioxide (a)	0.094833	kg
			methane (a)	0.027831	kg
			PCDD. PCDF (a)	2.06E-12	kg
			methylene oxide (a)	0.006487	kg
			benzene (a)	0.001804	kg
			benzo(a)pyrene (a)	2.44E-07	kg
			NMVOC, unspec. (a)	0.389171	kg
			particles (a)	0.010272	kg
			particles (small) (a)	0.030074	kg
			wood, unspec.	1000	kg

Table 7.50 Ferrous production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
KEA (hydro)	64354.95	kJ	sulfuric acid	2.947994	kg
KEA (nuclear)	1144345	kJ	ammonia (a)	0.006225	kg
KEA. fossil total	21172415	kJ	carbon dioxide. fossil (a)	2400.733	kg
KEA. others	2.048945	kJ	carbon monoxide (a)	28.59138	kg
KEA. renewable	0.790975	kJ	chlorine (a)	9.7E-12	kg
KEA. unspec.	4.713451	kJ	dinitrogen monoxide (a)	0.031266	kg
Air	0.003361	kg	fluorine (a)	1.23E-11	kg
flux materials (steel prod.)	71.1417	kg	hydrogen chloride (a)	0.082348	kg
cutting oil	0.002517	kg	hydrogen cyanide (a)	0.000557	kg
brown coal (r)	115.2708	kg	hydrogen fluoride (a)	0.01572	kg
hard coal (r)	413.3376	kg	hydrogen sulfide (a)	0.085395	kg
crude oil (r)	162.8157	kg	arsenic (a)	0.000222	kg
natural gas (r)	40.88701	m^3	cadmium (a)	0.000409	kg
bentonite (r)	1.577838	kg	chromium (a)	0.000289	kg
iron ore (r)	1849.153	kg	cobalt (a)	0.000244	kg
limestone (r)	161.9388	kg	copper (a)	0.0011	kg
sodium chloride (r)	0.005979	kg	lead (a)	0.00939	kg
cooling water	7342.928	kg	manganese (a)	0.064096	kg
water (process)	1679.516	kg	mercury (a)	8.81E-05	kg
water. unspec.	62754.72	kg	metals. unspec. (a)	1.65E-05	kg
ground water	2481.383	kg	nickel (a)	0.009326	kg
			selenium (a)	0.004055	kg
			vanadium (a)	0.00017	kg
			zinc (a)	0.034884	kg
			NOx (a)	7.296058	kg
			sulfur dioxide (a)	7.325926	kg
			methane (a)	6.201367	kg
			NMVOC (HC excl. benzene) (a)	0.000272	kg
			NMVOC. aromat.. unspec. (a)	4.28E-07	kg
			PCDD. PCDF (a)	4.16E-09	kg
			NMVOC. chlor.. unspec. (a)	2.22E-09	kg
			methylene oxide (a)	0.013461	kg
			benzene (a)	0.003354	kg
			toluene (a)	4.35E-07	kg
			xylene (a)	7.77E-07	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			ethane thiol (a)	9.21E-15	kg
			benzo(a)pyrene (a)	0.000167	kg
			PAH not B(a)P. unspec. (a)	1.88E-06	kg
			PAH. unspec. (a)	0.000488	kg
			NMVOC. unspec. (a)	0.474924	kg
			VOC (hydrocarbons) (a)	4.48E-05	kg
			VOC. unspec. (a)	0.00219	kg
			particles (a)	13.98319	kg
			particles (small) (a)	0.315786	kg
			waste heat (a)	10845.66	kJ
			dissolved solids (w)	8.29E-07	kg
			chloride (w)	0.092287	kg
			cyanide (w)	0.005327	kg
			fluoride (w)	0.238337	kg
			antimony (w)	8.17E-05	kg
			arsenic (w)	9.87E-05	kg
			cadmium (w)	7.33E-05	kg
			chromium (w)	0.003601	kg
			copper (w)	0.000678	kg
			iron (w)	0.045482	kg
			lead (w)	0.025281	kg
			mercury (w)	5.99E-05	kg
			metals. unspec. (w)	5.73E-07	kg
			nickel (w)	0.001664	kg
			selenium (w)	0.000162	kg
			sodium (w)	3.41E-05	kg
			zinc (w)	0.013314	kg
			ammonium (w)	0.002878	kg
			ammonium as N (w)	0.019492	kg
			nitrate as N (w)	0.002878	kg
			nitrogen compounds as N (w)	2.86E-05	kg
			nitrogen compounds. unspec. (w)	3.41E-09	kg
			salts. inorganic (w)	1.45E-08	kg
			sulfate (w)	0.137739	kg
			sulfide (w)	0.000165	kg
			detergents. oil (w)	7.87E-09	kg
			dissolved organics (w)	1.6E-09	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			chlorinated. org. compounds.. unspec. (w)	6.4E-12	kg
			HC. aromat.. unspec. (w)	0.000388	kg
			HC. unspec. (w)	1.74E-05	kg
			oil (w)	2.19E-05	kg
			PAH. unspec. (w)	4.9E-07	kg
			phenols (w)	0.007887	kg
			suspended solids (w)	6.97E-05	kg
			undissolved solids (w)	0.476298	kg
			AOX (w)	5.57E-06	kg
			BOD-5 (w)	0.125468	kg
			COD (w)	0.154703	kg
			crude steel. liquid (BOF)	1000	kg
			ferrous waste (wfd)	64.65092	kg
			hazardous waste (wfd)	0.009972	kg
			industrial waste (wfd)	0.001738	kg
			mineral waste (wfd)	0.00293	kg
			sewage sludge (wfd)	0.001163	kg
			slags and ash (wfd)	7.669935	kg
			waste oil (wfd)	0.002498	kg
			waste. inert (chemical industry) (wfd)	1.46E-05	kg
			waste. unspecified (wfd)	4.23E-05	kg
			ashes and slags (wfr)	112.7773	kg
			dusts (steel prod.) (wfr)	19	kg
			gypsum (flue gas clean.) (wfr)	2.798371	kg
			clearing residue (steel production) (wfr)	7.787598	kg
			sludge (blast furnace) (wfr)	3.025932	kg
			top gas dust (wfr)	15.56591	kg
			top gas sludge (wfr)	224.7172	kg
			waste. unspec.	0.0198	kg
			condensate	1.26999	kg
			sewage (cooling water)	7022.562	kg
			sewage (process)	11.69025	kg
			sewage. clarified	1.430252	kg
			sewage. unspec.	66831.62	kg
			steam	0.103371	kg

Table 7.51 PE production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
Nitrogen	169.9842	kg	ammonia (a)	8.05E-07	kg
Oxygen	0.030624	kg	asbestos (a)	3.53E-12	kg
phosphate (as P2O5)	3.52E-06	kg	carbon dioxide. fossil (a)	1980.098	kg
KEA (hydro)	625972.9	kJ	carbon dioxide. renewable (a)	43.14216	kg
KEA (nuclear)	3890064	kJ	carbon disulfide (a)	1.5E-08	kg
KEA. fossil total	73408914	kJ	carbon monoxide (a)	12.86688	kg
KEA. others	290709.9	kJ	chlorine (a)	1.4E-07	kg
KEA. renewable. others	618019.7	kJ	dinitrogen monoxide (a)	4.03E-09	kg
KEA. unspec.	4429.914	kJ	fluorine (a)	9.96E-08	kg
carbon dioxide. renewable (a)	43.14216	kg	hydrogen (a)	0.067221	kg
Sulfur	0.000232	kg	hydrogen chloride (a)	0.095355	kg
Air	260.7068	kg	hydrogen cyanide (a)	4.92E-16	kg
brown coal (r)	0.003065	kg	hydrogen fluoride (a)	0.003089	kg
hard coal (r)	162.139	kg	hydrogen sulfide (a)	0.00012	kg
crude oil (r)	945.72	kg	antimony (a)	2.25E-08	kg
natural gas (r)	758.9707	m^3	arsenic (a)	1.24E-07	kg
wood (r)	47.06383	kg	cadmium (a)	1.13E-07	kg
barite (r)	0.00021	kg	chromium (a)	1.26E-06	kg
bauxite (r)	0.010872	kg	copper (a)	2.64E-09	kg
bentonite (r)	0.03435	kg	lead (a)	1.82E-06	kg
calcium sulfate (r)	0.003327	kg	mercury (a)	3.96E-06	kg
chalk (r)	3.68E-30	kg	metals. unspec. (a)	0.002891	kg
clay (r)	5.5E-07	kg	nickel (a)	1.27E-06	kg
dolomite (r)	0.019448	kg	selenium (a)	2.39E-16	kg
feldspar (r)	6.19E-14	kg	silver (Ag) (a)	6.9E-15	kg
fluorite (r)	0.000424	kg	zinc (a)	1.78E-06	kg
granite (r)	2.26E-11	kg	NOx (a)	4.492029	kg
gravel (r)	0.005871	kg	sulfur dioxide (a)	6.229184	kg
limestone (r)	0.444404	kg	sulfuric acid (a)	1.86E-12	kg
olivine (r)	0.014928	kg	methane (a)	20.05984	kg
potassium chloride (r)	1.47E-05	kg	NMVOC. aromat.. unspec. (a)	0.086528	kg
quartz (SiO2) (r)	1.26E-26	kg	dichloroethane (a)	1.5E-07	kg
rutil (r)	1.61E-06	kg	dichloromethane (a)	5.79E-11	kg
sand (r)	0.08438	kg	vinyl chloride (a)	2.5E-06	kg
shale (r)	0.00942	kg	PCDD. PCDF (a)	4.32E-29	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
sodium chloride (r)	0.365234	kg	NMVOC. chlor.. unspec. (a)	1.03E-06	kg
sodium nitrate (NaNO3) (RiL)	4.49E-07	kg	NMVOC. fluor.. unspec. (a)	0.001338	kg
talcum (r)	1.26E-23	kg	aldehydes. unspec. (a)	2.47E-08	kg
chromium (Cr) (r)	0.002745	kg	ethene (a)	0.001631	kg
copper (Cu) (r)	3.43E-06	kg	propene (a)	0.001208	kg
ferromanganese (r)	0.001445	kg	benzene (a)	4.45E-07	kg
iron (Fe) (r)	1.59115	kg	ethyl benzene (a)	9.54E-08	kg
lead (Pb) (r)	0.00251	kg	styrene (a)	6.5E-11	kg
magnesium (Mg) (r)	1.5E-07	kg	toluene (a)	2.45E-07	kg
mercury (Hg) (r)	7.17E-07	kg	xylene (a)	1.59E-07	kg
nickel (Ni) (r)	3E-07	kg	ethylene oxide (a)	1.49E-07	kg
zinc (Zn) (r)	0.015753	kg	ethane thiol (a)	2.39E-05	kg
sulfur (r)	0.055691	kg	PAH. unspec. (a)	1.27E-06	kg
peat (r)	1.926111	kg	NMVOC. unspec. (a)	0.151465	kg
biomass (kg)	18.63349	kg	VOC. unspec. (a)	4.659357	kg
industrial waste (wfd)	17.93835	kg	particles (PM10) (a)	0.794943	kg
cooling water	54223.87	kg	dissolved solids (w)	0.021611	kg
Kühlwasser (Trinkwasser)	161.2018	kg	acids as H(+) (w)	0.00292	kg
Wasser (Prozess) (Trinkwasser)	1865.962	kg	bromate (w)	5.6E-07	kg
water (process)	1716.681	kg	carbonate (w)	0.029106	kg
			chlorate (w)	0.000101	kg
			chloride (w)	0.157818	kg
			chlorine. dissolved (w)	1.08E-06	kg
			cyanide (w)	1.51E-07	kg
			fluoride (w)	6.75E-06	kg
			aluminium (w)	0.000561	kg
			arsenic (w)	2E-07	kg
			cadmium (w)	1.58E-08	kg
			calcium (w)	0.002903	kg
			chromium (VI) (w)	2.24E-09	kg
			copper (w)	0.000157	kg
			iron (w)	4.86E-05	kg
			lead (w)	2.07E-06	kg
			magnesium (w)	1.07E-06	kg
			manganese (w)	1.81E-07	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			mercury (w)	2.21E-07	kg
			metals. unspec. (w)	0.007479	kg
			nickel (w)	3.79E-07	kg
			pottassium (w)	0.000679	kg
			sodium (w)	0.077693	kg
			strontium (w)	1.07E-08	kg
			zinc (w)	0.000134	kg
			ammonium (w)	0.004056	kg
			nitrate (w)	0.002333	kg
			nitrogen compounds. unspec. (w)	0.001528	kg
			phosphorous compounds as P (w)	0.000477	kg
			sulfate (w)	0.832874	kg
			sulfur (w)	8.37E-06	kg
			detergents. oil (w)	0.006293	kg
			dissolved organics (w)	0.010164	kg
			1.2-dichloroethane (w)	2.49E-09	kg
			vinyl chloride (w)	4.55E-08	kg
			PCDD. PCDF (w)	9.85E-07	kg
			chlorinated. org. compounds.. unspec. (w)	5.84E-06	kg
			benzene (w)	3.98E-12	kg
			HC. unspec. (w)	0.004531	kg
			phenols (w)	0.001878	kg
			organic compounds (w)	6.85E-05	kg
			organo-silicon compounds (w)	3.04E-17	kg
			organo-tin compounds (w)	7.54E-08	kg
			suspended solids (w)	0.328562	kg
			AOX (w)	4.24E-09	kg
			BOD-5 (w)	0.022624	kg
			COD (w)	0.205974	kg
			TOC (w)	0.011156	kg
			PE tube	1000	kg
			Abfälle (Grubenverfüllung) (AzB)	31.46495	kg
			Abfälle (ungeregelte Chemikalien) (AzB)	3.321556	kg
			hazardous waste (wfd)	2.525398	kg
			mineral waste (wfd)	1.396079	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			paper. cardboard (wfd)	0.929821	kg
			plastics. unspec. (wfd)	4.867426	kg
			slags and ash (wfd)	20.25448	kg
			waste for incineration (wfd)	0.887211	kg
			waste. inert (chemical industry) (wfd)	0.724559	kg
			waste. unspecified (wfd)	2.33172	kg
			metals. unspec. (wfr)	0.120308	kg
			mixed valuable materials (wfr)	4.509397	kg
			plastic containers (wfr)	5.2E-09	kg
			sewage (cooling water)	54385.08	kg
			sewage (process)	3582.643	kg

Table 7.52 PET production

Inputs	Quantity	Units	Outputa	Quantity	Units
Nitrogen	265.7156	kg	ammonia (a)	1.39E-06	kg
Oxygen	0.007156	kg	asbestos (a)	7.38E-11	kg
phosphate (as P2O5)	0.017599	kg	carbon dioxide. fossil (a)	4425.287	kg
KEA (hydro)	524555.9	kJ	carbon dioxide. renewable (a)	0.008629	kg
KEA (nuclear)	8777541	kJ	carbon disulfide (a)	3.65E-08	kg
KEA. fossil total	91613273	kJ	carbon monoxide (a)	11.52657	kg
KEA. others	453943.9	kJ	chlorine (a)	3.3E-07	kg
KEA. renewable. others	465023.2	kJ	dinitrogen monoxide (a)	4.43E-08	kg
KEA. unspec.	3140.982	kJ	fluorine (a)	2.32E-08	kg
carbon dioxide. renewable (a)	0.008629	kg	hydrogen (a)	0.232117	kg
Sulfur	3.97E-05	kg	hydrogen chloride (a)	0.279152	kg
Air	4075.804	kg	hydrogen cyanide (a)	1.18E-15	kg
brown coal (r)	0.019457	kg	hydrogen fluoride (a)	0.010492	kg
hard coal (r)	503.1333	kg	hydrogen sulfide (a)	1.31E-05	kg
crude oil (r)	763.8042	kg	antimony (a)	3.43E-11	kg
natural gas (r)	1131.448	m^3	arsenic (a)	3.58E-08	kg
wood (r)	0.009074	kg	cadmium (a)	1.03E-07	kg
barite (r)	0.000149	kg	chromium (a)	0.003489	kg
bauxite (r)	0.002445	kg	copper (a)	1.03E-08	kg
bentonite (r)	0.069991	kg	lead (a)	4.35E-07	kg
calcium sulfate (r)	0.006971	kg	mercury (a)	5.28E-06	kg
chalk (r)	1.86E-30	kg	metals. unspec. (a)	0.004511	kg
clay (r)	1.5E-05	kg	nickel (a)	0.006344	kg
dolomite (r)	0.004821	kg	selenium (a)	1.16E-12	kg
feldspar (r)	1.49E-13	kg	silver (Ag) (a)	3.34E-11	kg
fluorite (r)	0.000848	kg	zinc (a)	2.83E-07	kg
granite (r)	1.13E-11	kg	NOx (a)	10.85315	kg
gravel (r)	0.001452	kg	sulfur dioxide (a)	14.81909	kg
limestone (r)	0.285055	kg	sulfuric acid (a)	2.55E-11	kg
olivine (r)	0.003692	kg	methane (a)	41.34672	kg
potassium chloride (r)	0.000549	kg	NMVOC. aromat.. unspec. (a)	0.280571	kg
quartz (SiO2) (r)	6.12E-23	kg	dichloroethane (a)	5.6E-08	kg
rutil (r)	2.67E-30	kg	dichloromethane (a)	7.34E-10	kg
sand (r)	0.249398	kg	vinyl chloride (a)	1.23E-06	kg
shale (r)	0.019734	kg	PCDD. PCDF (a)	2.12E-28	kg

Inputs	Quantity	Units	Outputa	Quantity	Units
sodium chloride (r)	1.667156	kg	NMVOC. chlor.. unspec. (a)	2.9E-06	kg
sodium nitrate (NaNO3) (RiL)	4.33E-15	kg	NMVOC. fluor.. unspec. (a)	5.65E-06	kg
talcum (r)	3.06E-22	kg	aldehydes. unspec. (a)	3.56E-12	kg
chromium (Cr) (r)	3.32E-09	kg	ethene (a)	0.001628	kg
copper (Cu) (r)	6.25E-06	kg	propene (a)	0.001206	kg
ferromanganese (r)	0.000357	kg	benzene (a)	0.002225	kg
iron (Fe) (r)	0.393564	kg	ethyl benzene (a)	0.000477	kg
lead (Pb) (r)	0.001297	kg	styrene (a)	3.21E-07	kg
magnesium (Mg) (r)	1.44E-15	kg	toluene (a)	0.001224	kg
mercury (Hg) (r)	2.77E-06	kg	xylene (a)	0.000794	kg
nickel (Ni) (r)	7.05E-10	kg	ethylene oxide (a)	0.000746	kg
zinc (Zn) (r)	4.61E-05	kg	ethane thiol (a)	1.23E-05	kg
sulfur (r)	0.113016	kg	PAH. unspec. (a)	0.006351	kg
peat (r)	0.14542	kg	NMVOC. unspec. (a)	0.894692	kg
biomass (kg)	45.3415	kg	VOC. unspec. (a)	8.777475	kg
industrial waste (wfd)	29.77331	kg	particles (PM10) (a)	2.414426	kg
cooling water	91239.26	kg	dissolved solids (w)	0.13317	kg
Wasser (Prozess) (Trinkwasser)	3431.684	kg	acids as H(+) (w)	0.052482	kg
water (process)	1504.941	kg	bromate (w)	3.8E-07	kg
			carbonate (w)	0.078729	kg
			chlorate (w)	0.000408	kg
			chloride (w)	0.216208	kg
			chlorine. dissolved (w)	1.03E-06	kg
			cyanide (w)	3.2E-08	kg
			fluoride (w)	1.41E-06	kg
			aluminium (w)	0.000755	kg
			arsenic (w)	3.08E-07	kg
			cadmium (w)	7.36E-11	kg
			calcium (w)	0.000143	kg
			chromium (VI) (w)	3.68E-09	kg
			copper (w)	8.39E-05	kg
			iron (w)	1.71E-05	kg
			lead (w)	2.52E-07	kg
			magnesium (w)	4.25E-07	kg
			manganese (w)	3.48E-10	kg

Inputs	Quantity	Units	Outputa	Quantity	Units
			mercury (w)	1.42E-07	kg
			metals. unspec. (w)	0.020209	kg
			nickel (w)	1.61E-06	kg
			pottassium (w)	1.72E-05	kg
			sodium (w)	0.2174	kg
			strontium (w)	4.46E-08	kg
			zinc (w)	4.83E-05	kg
			ammonium (w)	0.003867	kg
			nitrate (w)	0.011797	kg
			nitrogen compounds. unspec. (w)	0.002355	kg
			phosphorous compounds as P (w)	0.000161	kg
			sulfate (w)	0.348388	kg
			sulfur (w)	7.34E-09	kg
			detergents. oil (w)	0.019443	kg
			dissolved organics (w)	0.014314	kg
			1.2-dichloroethane (w)	1.26E-09	kg
			vinyl chloride (w)	2.28E-08	kg
			PCDD. PCDF (w)	3.23E-12	kg
			chlorinated. org. compounds.. unspec. (w)	2.43E-05	kg
			benzene (w)	7.62E-18	kg
			HC. unspec. (w)	0.07328	kg
			phenols (w)	0.00068	kg
			organic compounds (w)	0.339115	kg
			organo-silicon compounds (w)	4.1E-17	kg
			organo-tin compounds (w)	4.05E-12	kg
			suspended solids (w)	0.360481	kg
			AOX (w)	2.21E-09	kg
			BOD-5 (w)	1.034358	kg
			COD (w)	1.408732	kg
			TOC (w)	0.02292	kg
			PET film	1000	kg
			Abfälle (Grubenverfüllung) (AzB)	97.94187	kg
			Abfälle (ungeregelte Chemikalien) (AzB)	8.30406	kg
			hazardous waste (wfd)	2.69679	kg
			mineral waste (wfd)	0.450556	kg

Inputs	Quantity	Units	Outputa	Quantity	Units
			paper. cardboard (wfd)	3.58E-08	kg
			plastics. unspec. (wfd)	8.389658	kg
			slags and ash (wfd)	44.16755	kg
			waste for incineration (wfd)	36.37716	kg
			waste. inert (chemical industry) (wfd)	1.985231	kg
			waste. unspecified (wfd)	1.842345	kg
			metals. unspec. (wfr)	6.77E-06	kg
			mixed valuable materials (wfr)	0.174315	kg
			plastic containers (wfr)	2.63E-09	kg
			sewage (cooling water)	91239.26	kg
			sewage (process)	4936.625	kg

Table 7.53 Virgin glass production – 1,000 kg

Inputs	Quantity	Units	Outputs	Quantity	Units
Chlorine	4.52E-06	kg	salts. inorg.	1.19E-08	kg
sodium sulfate	4	kg	sulfuric acid	0.002952	kg
KEA (hydro)	731.1667	kJ	ammonia (a)	0.019182	kg
KEA (nuclear)	109743.7	kJ	carbon dioxide. fossil (a)	4157.028	kg
KEA. fossil total	54040055	kJ	carbon monoxide (a)	8.850383	kg
KEA. others	209849.3	kJ	chlorine (a)	9.71E-15	kg
KEA. renewable	264400.7	kJ	dinitrogen monoxide (a)	0.413587	kg
KEA. unspec.	0.392331	kJ	fluorine (a)	1.23E-14	kg
land use C1 (FRG)	0.152172	m^2	hydrogen chloride (a)	0.106482	kg
land use C7 (FRG)	9.11E-05	m^2	hydrogen cyanide (a)	5.58E-07	kg
quicklime	0.001605	kg	hydrogen fluoride (a)	0.010395	kg
Air	3.36E-06	kg	hydrogen sulfide (a)	8.56E-05	kg
flux materials (steel prod.)	0.071227	kg	arsenic (a)	2.41E-05	kg
cutting oil	2.52E-06	kg	cadmium (a)	0.00084	kg
crude oil	1.127113	kg	chromium (a)	3.55E-05	kg
brown coal (r)	0.50872	kg	cobalt (a)	3.21E-07	kg
hard coal (r)	27.88518	kg	copper (a)	2.02E-06	kg
crude oil (r)	1146.485	kg	lead (a)	0.041214	kg
natural gas (r)	220.8756	m^3	manganese (a)	7.57E-05	kg
bauxite (r)	0.000864	kg	mercury (a)	9.24E-08	kg
bentonite (r)	0.00158	kg	metals. unspec. (a)	1.65E-08	kg
gravel (r)	845.4	kg	nickel (a)	0.002435	kg
iron ore (r)	1.65624	kg	selenium (a)	4.06E-06	kg
limestone (r)	378.6908	kg	tin (a)	7.56E-08	kg
rock salt (r)	263.81	kg	vanadium (a)	1.91E-07	kg
sodium chloride (r)	5.99E-06	kg	zinc (a)	0.007344	kg
iron (Fe) (r)	0.000378	kg	NOx (a)	37.38796	kg
electric energy	7727.196	kJ	sulfur dioxide (a)	8.528444	kg
Coke	11.92204	kg	methane (a)	1.532258	kg
heat energy	1304.578	kJ	NMVOC (HC excl. benzene) (a)	2.72E-07	kg
scrap (iron) (wfr)	0.195132	kg	NMVOC. aromat.. unspec. (a)	4.29E-10	kg
cooling water	374.0095	kg	PCDD. PCDF (a)	6.65E-11	kg
water (process)	13920.78	kg	NMVOC. chlor.. unspec. (a)	2.23E-12	kg
water. unspec.	206.6722	kg	perfluoroethane (a)	1.1E-08	kg
ground water	2.48436	kg	perfluoromethane (a)	8.72E-08	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			methylene oxide (a)	0.215077	kg
			benzene (a)	0.059273	kg
			toluene (a)	4.35E-10	kg
			xylene (a)	7.78E-10	kg
			ethane thiol (a)	9.22E-18	kg
			benzo(a)pyrene (a)	7.87E-06	kg
			PAH not B(a)P. unspec. (a)	1.9E-09	kg
			PAH. unspec. (a)	4.89E-07	kg
			NMVOC. unspec. (a)	1.673196	kg
			VOC (hydrocarbons) (a)	0.03726	kg
			VOC. unspec. (a)	1.12E-05	kg
			exhaust gas. dry (standard conditions) (a)	0.921974	m^3
			particles (>PM10) (a)	6.44E-06	kg
			particles (a)	29.86368	kg
			particles (PM10) (a)	6.44E-06	kg
			particles (small) (a)	0.997401	kg
			waste heat (a)	10.85868	kJ
			dissolved solids (w)	8.3E-10	kg
			acids as H(+) (w)	2.21E-09	kg
			chloride (w)	0.00038	kg
			cyanide (w)	5.33E-06	kg
			fluoride (w)	0.000239	kg
			antimony (w)	8.18E-08	kg
			arsenic (w)	9.91E-08	kg
			cadmium (w)	9.11E-06	kg
			calcium (w)	2.26E-05	kg
			chromium (w)	1.87E-05	kg
			copper (w)	1.57E-05	kg
			iron (w)	0.000329	kg
			lead (w)	7.95E-05	kg
			mercury (w)	6.04E-08	kg
			metals. unspec. (w)	5.74E-10	kg
			nickel (w)	1.67E-06	kg
			selenium (w)	1.62E-07	kg
			sodium (w)	3.41E-08	kg
			zinc (w)	1.33E-05	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			ammonium (w)	4.31E-06	kg
			ammonium as N (w)	1.95E-05	kg
			nitrate as N (w)	2.88E-06	kg
			nitrogen compounds as N (w)	5.53E-06	kg
			nitrogen compounds. unspec. (w)	3.41E-12	kg
			salts. inorganic (w)	161.69	kg
			sulfate (w)	0.000138	kg
			sulfide (w)	1.65E-07	kg
			detergents. oil (w)	7.88E-12	kg
			dissolved organics (w)	1.6E-12	kg
			chlorinated. org. compounds.. unspec. (w)	6.41E-15	kg
			HC. aromat.. unspec. (w)	3.88E-07	kg
			HC. unspec. (w)	1.74E-08	kg
			oil (w)	2.19E-08	kg
			PAH. unspec. (w)	4.91E-10	kg
			phenols (w)	7.9E-06	kg
			suspended solids (w)	6.98E-08	kg
			undissolved solids (w)	0.000477	kg
			ammonia (fw)	2.7E-06	kg
			nitrate (fw)	2.7E-06	kg
			AOX (w)	0.000964	kg
			BOD-5 (w)	1.057235	kg
			COD (w)	2.55065	kg
			landfill volume	0.001823	m^3
			steel	0.910182	kg
			bottles	1000	kg
			heat energy	1.809201	kJ
			ferrous waste (wfd)	0.064729	kg
			hazardous waste (wfd)	1.55E-05	kg
			industrial waste (wfd)	1.74E-06	kg
			mineral waste (wfd)	82.28405	kg
			radioactive waste (high-radioactive) (wfd)	1.65E-08	m^3
			sewage sludge (wfd)	0.085553	kg
			slags and ash (wfd)	0.104754	kg
			waste (soda production) (wfd)	8.1696	kg

Inputs	Quantity	Units	Outputs	Quantity	Units
			waste oil (wfd)	2.5E-06	kg
			waste. inert (chemical industry) (wfd)	1.46E-08	kg
			waste. unspecified (wfd)	0.000184	kg
			ashes and slags (wfr)	9.096725	kg
			dusts (steel prod.) (wfr)	0.019023	kg
			gypsum (flue gas clean.) (wfr)	0.795443	kg
			clearing residue (steel production) (wfr)	0.007797	kg
			sludge (blast furnace) (wfr)	0.00303	kg
			top gas dust (wfr)	0.015585	kg
			top gas sludge (wfr)	0.224987	kg
			waste. unspec.	0.008249	kg
			seepage water. collected	0.018823	kg
			seepage water. diffuse	0.005782	kg
			condensate	0.001272	kg
			sewage (cooling water)	373.6874	kg
			sewage (process)	0.011704	kg
			sewage. clarified	3279.665	kg
			sewage. unspec.	66.91298	kg
			steam	0.095739	kg

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ANNEX II – DATA COLLECTION FOR MCDM

8 ANNEX II – DATA COLLECTION FOR MCDM

To perform the MCDM, several data has been collected from different sources. mentioned in Chapter V. To clarify how the results have been reached, following sub-chapters will describe the formulas applied to operational units described in Chapter IV.

8.1 ENVIRONMENTAL CRITERIA

For environmental criteria results from LCA have been used, being already shown in Chapter IV. The calculation of net energy has been explained in Chapter V.

8.2 ECONOMIC CRITERIA

The sub-criteria belonging to economic criteria are investment costs. operational costs and operational revenues. To show how data have been processed and how sub-criteria have been constructed, in the next sub-chapters will be explained according to the operational units described in Chapter IV.

8.2.1 Investment cost criteria

The investment cost I_c of a scenario is calculated based on:

$$Ic = I_{collection\ and\ transportation} + I_{transfer\ station} + I_{ecocenter} + I_{sorting\ plant} + I_{aerobic\ MBT} + I_{anaerobic\ MBT\ or\ AD\ MBW} + I_{landfill}$$

RDF plant is not owned by AMARSUL, so in scenarios where RDF production is considered. the investment cost is null.

8.2.1.1 Investment cost of waste collection and transport

The investment needed to perform waste collection and transport focus on collection vehicles and containers. The formula applied is:

$$I_{collection\ and\ transportation} = I_{vehicles} + I_{containers}$$

The formula has been applied to mix MSW, BMW, packaging, paper/cardboard and glass waste collection. The data used to perform the calculation is on Table 8.1.

Table 8.1 Data for collection and transport

Waste collection and transport	Mixed MSW	BMW	Packaging waste	Paper/Cardboard waste	Glass waste
Vehicle prices (€/unit)	103,465	103,465	103,465	103,465	103,465
Containers prices (€/unit)	172	172	415	415	415

To calculate the number of vehicles and of containers, the following formulas have been used:

Number of vehicles = ((quantity of waste per day/waste density)/number of days when waste was produced)/(quantity collected per day of collection)

The number of containers was obtained by the formula:

Number of containers = Number of habitations/number of containers in AMARSUL

The formula has been applied for mix MSW and BMW. For packaging waste, paper/cardboard and glass, the data collected in SPV (2010) have been used.

8.2.1.2 Investment cost of transfer station

The investment cost of transfer station was collected from bibliography, mentioned in Chapter V. The value used was 1.19 millions €/unit.

8.2.1.3 Investment cost of ecocenter

The investment cost of ecocenter was collected from bibliography, mentioned in Chapter V. The value applied was 0.20 millions €/unit.

8.2.1.4 Investment cost of automated sorting plant for packaging waste

For sorting plant, the investment cost have been collected from bibliography, being the value used 13.2 millions €/unit.

8.2.1.5 Investment cost of sorting plant of paper/cardboard waste

For sorting plant, the investment cost have been collected from bibliography, being the value used 11 millions €/unit.

8.2.1.6 Investment cost of aerobic MBT plant

For aerobic MBT plant, the investment cost has been obtained through the formula:

$I_{aerobic\ MBT} = 0.0015 * (Quantity\ of\ waste)^{0.8}$,

where 7,500 tones/year ≤ quantity of waste ≤ 250,000 tones/year.

8.2.1.7 Investment cost of anaerobic MBT plant

For anaerobic MBT plant, the investment cost has been obtained through the formula:

$I_{anaerobic\ MBT} = 0.0025 * (Quantity\ of\ waste)^{0.8}$,

where 7,500 tones/year ≤ quantity of waste ≤ 250,000 tones/year.

8.2.1.8 Investment cost of anaerobic MBT plant

For anaerobic MBT plant, the investment cost has been obtained through the formula:

$I_{anaerobic\ digestion} = 0.0345 * (Quantity\ of\ waste)^{0.55}$,

where 2,500 tones/year ≤ quantity of waste ≤ 100,000 tones/year.

8.2.1.9 Investment cost of sanitary landfill

For sanitary landfill, the investment cost has been obtained through the formula:

$I_{anaerobic\ MBT} = 0.0035 * (Quantity\ of\ waste)^{0.7}$,

where 60,000 tones/year ≤ quantity of waste ≤ 1,500,000 tones/year.

8.2.2 Operational cost criteria

The operational cost Oc of a scenario is calculated based on:

$Oc = O_{collection\ and\ transportation} + O_{transfer\ station} + O_{ecocenter} + O_{sorting\ plant} + O_{aerobic\ MBT} + O_{anaerobic\ MBT\ or\ AD\ MBW} + O_{landfill}$

8.2.3 Operational cost of waste collection and transport

The operational cost for waste collection and transport can be calculated based on the formula:

$O_{collection\ and\ transportation} = O_{vehicles} + O_{containers} + O_{fuel\ and\ salaries}$

$O_{vehicles}$ can be calculated through:

$O_{vehicles}$ = *Number of vehicles*Price vehicle*annuity+ maintenance + residual value*

Number of vehicle and price vehicle has been presented in Table 8.1. Annuity considered is 6% (based on data provided by municipalities), and maintenance and repair is 14% of investment cost (Gomes *et al.*, 2008). Residual value considered is 15% (Gomes *et al.*, 2008).

For containers, the formula is:

$O_{containers}$ = *Number of containers*Price container*annuity + maintenance + residual value*

Number of containers and price of containers have been described previously. Annuity considered is 0.019% (based on data provided by municipalities), and maintenance and repair is 35% of investment cost (Gomes *et al.*, 2008). Residual value considered is null (Gomes *et al.*, 2008).

$O_{\text{fuel and salaries}}$ can be calculated through:

$O_{\text{fuel and salaries}}$ = *Number of operators*salaries + diesel fuel consumption*price diesel*distance*

According to municipalities, the number of operators by shift is three. The average annual salary per operator is 8,760 €. The average price of diesel is 1.122 €/liter. The distance has been determined during LCA, being referred in Annex I, as well the diesel fuel consumption.

The formula has been applied for all types of waste collection modeled in AMARSUL system.

8.2.3.1 Operational cost of transfer station and ecocenters

The operational costs involving these units are minimal concerning when compared with the other operational units. Sometimes these units share the same operators, since the units are together; or can also be associated to other infrastructures, like sanitary landfills. For that reason, operational cost has not been considered in this study.

8.2.3.2 Operational cost of automated sorting plant

For sorting plant, the operational cost have been collected from bibliography, being the value used 218 €/t of waste.

8.2.3.3 Operational cost of paper/cardboard sorting plant

For this sorting plant, the operational cost have been collected from bibliography, being the value used 65 €/t of waste.

8.2.3.4 Operational cost aerobic MBT plant

For aerobic MBT plant, the operational cost has been obtained through the formula:

$I_{aerobic\ MBT}$ *(€/t input waste) = (4,000 * (Quantity of waste)$^{-0.4}$)*Quantity of waste + Annuity*Investment cost*1E^{6}*

where 7,500 tones/year ≤ quantity of waste ≤ 250,000 tones/year.

The annuity (a_c) has been obtained by the formula:

$a_c = (i_c*(1+i_c)^n)/((1+i_c)^n - 1)$, *where*

i_c is the social tax, being considered 7% and n the unit lifetime (20 years).

8.2.3.5 Operational cost anaerobic MBT plant

For anaerobic MBT plant, the operational cost has been obtained through the formula:

$I_{anaerobic\ MBT}$ *(€/t input waste) = (5,000 * (Quantity of waste)$^{-0.4}$)*Quantity of waste + Annuity*Investment cost*1E6*

where 7,500 tones/year ≤ quantity of waste ≤ 250,000 tones/year.

The annuity (a_c) has been obtained by the formula:

$a_c = (i_c*(1+i_c)^n)/((1+i_c)^n - 1)$, *where*

i_c is the social tax, being considered 7% and n the unit lifetime (20 years).

8.2.3.6 Operational cost anaerobic digestion plant

For anaerobic digestion plant, which represents the extra operation line in AMARSUL predicted AD MBT, the operational cost has been obtained through the formula:

*$I_{anaerobic\ digestion}$ (€/t input waste) = (17,000 * (Quantity of waste)$^{-0.6}$)*Quantity of waste + Annuity*Investment cost*1E6*

where 2,500 tones/year ≤ quantity of waste ≤ 100,000 tones/year.

The annuity (a_c) has been obtained by the formula:

$a_c = (i_c*(1+i_c)^n)/((1+i_c)^n - 1)$, *where*

i_c is the social tax. being considered 7% and n the unit lifetime (20 years).

8.2.3.7 Operational cost sanitary landfill

For sanitary landfill, the operational cost has been obtained through the formula:

*$I_{sanitary\ landfill}$ (€/t input waste) = (150 * (Quantity of waste)$^{-0.3}$)*Quantity of waste + Annuity*Investment cost*1E6*

where 60,000 tones/year ≤ quantity of waste ≤ 1,500,000 tones/year.

The annuity (a_c) has been obtained by the formula:

$a_c = (i_c*(1+i_c)^n)/((1+i_c)^n - 1)$, *where*

i_c is the social tax, being considered 7% and n the unit lifetime (20 years).

8.2.4 Operational revenues criteria

For products obtained, in Table 8.2 are presented the different values applied to products obtained from waste management systems alternatives. The values from recyclable material have been based on Sociedade Ponto Verde, applying the criteria defined by the amount collected/inhabitant. As is shown, there are products which do not have a market value. For those materials, the benefit of MSW management system is to divert them from landfill, saving the cost of landfilling it.

Table 8.2 Selling for MSW management system products

Products	Values
Glass from sorting plant (€/t)	48
Paper/Cardboard from sorting plant (€/t)	135
Composites from sorting plant (€/t)	823
Plastics form sorting plant (€/t)	876
Mixed plastics from sorting plant (€/t)	530
Ferrous metals from sorting plant (€/t)	688
Non-ferrous metals from sorting plant (€/t)	1.283
Electricity from AD MBT and AD unit (€/kWh)	0.1254
Electricity from landfill (€/kWh)	0.11123
Glass from MBT plant (€/t)	5
Paper/Cardboard from MBT plant (€/t)	5
Composites from MBT plant (€/t)	0 (avoided cost of landfilling)
Plastic film and HDPE from MBT plant (€/t)	275
PET from MBT plant (€/t)	180
Mixed plastics from MBT plant (€/t)	0 (avoided cost of landfilling)
Ferrous metals from MBT plant (€/t)	15
Non-ferrous metals from MBT plant (€/t)	35
RDF (€/t)	0 (avoided cost of landfilling)
Compost (€/t)	2.5

8.3 SOCIAL CRITERIA

Social criteria used in this study has been referred and described in Chapter V. Here will be explained the procedure to construct those criteria.

8.3.1 Economic sufficiency

To determine economic sufficiency has been applied the following formula:

Economic sufficiency (%) = Revenues from billing system/total cost

In the two scenarios (water billing system with waste charge and PAYT), the total cost is the amount need to manage MSW system. In PAYT, the amount to be charged to population is the same as the total cost system. In case of water billing system, the amount charged to population is dependent from water consumption and charges established by municipalities, being presented in Table 8.3.

Table 8.3 Economic sufficiency calculation parameters

Municipalities	Annual waste charge (€/water user)	Annual water consumption (m^3/user)	Annual waste charge (€/t waste)	Annual water consumption (m^3)	Total revenue (€)
Alcochete	18.15	762	0.02	6,324,106	150,627
Almada	36.00	382	0.09	37,876,614	3,565,440
Barreiro	24.34	368	0.07	15,545,224	1,026,807
Moita	51.00	514	0.10	17,370,855	1,725,177
Montijo	12.00	547	0.02	14,007,819	307,080
Palmela	24.00	567	0.04	17,939,232	759,840
Seixal	27.60	322	0.09	24,621,632	2,110,158
Sesimbra	18.54	359	0.05	10,773,735	555,755
Setúbal	47.98	489	0.10	30,506,829	2,991,409

8.3.2 Fee

The determination of the fee has been applied the following formula:

Fee (€/t) = Total cost – Total benefits + Waste Management tax

Total cost represents all operational costs plus interest rate, being considered 5% of operational costs (according to AMARSUL, 2009). The Waste Management tax is applied annually to waste landfilled, being the amount 2€/t.

8.3.3 Odor

The calculation of odor impact has been obtained from LCA performed in Chapter IV.

8.4 TECHNICAL CRITERION

The technical criterion used to represent it has been landfill deviation rate. The determination of landfill deviation rate for each alternative studied has been made based on LCI, being determined as:

Landfill deviation rate (%) = Waste avoided/total waste production