Environmental Evaluation of Solid Household Waste Management – the Augustenborg Ecocity Example

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Lund 2010

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SUMMARY

Generation of solid waste is creating large environmental problems in great parts of the world. However, solid waste can also be seen as a resource and potentially constitute resources for material and energy production. The potential benefits from waste are often optimized through separation of materials at source. This has been the basic understanding for the development of the Swedish system for solid household waste management and national objectives related to this topic.

The objective of this thesis is to assess and evaluate the current performance of the Swedish solid waste management strategy from an environmental standpoint. The thesis is based on a full-scale long term investigation of solid household waste generation and management in a residential area in Malmö, southern Sweden. The study-site has been used both as an assumed standard area in which the current waste generation and household recycling behaviour could be followed and assessed in detail, as well as a trial area for introduction of new systems for property-close source-separation of household waste as well as different strategies to enhance source-separation behaviour.

Three different types of methods were used in this work; continuous weighing of disposed waste, repeated waste composition analyses and use of life cycle assessment as a tool for evaluation of environmental impacts related to changes in waste composition, recycling behavior and choices between different waste treatment alternatives. A questionnaire was also used in order to shed light on the attitudes towards currently used system for separation of solid household waste from the side of the households.

It is seen that 33 weight-% of generated household waste currently is separated for material recycling by households. However, the ratio of miss-sorting is in many cases high, reaching 32 weight-% at most in the fraction for source-separated plastic packaging. Composition studies of generated waste have shown that less than 20 weight-% consists of non-recyclable waste. Thus, the potential improvements of current recycling behaviour are large.

The environmental benefits connected to recycling of each material type respectively depend on several different factors. The environmental benefit per ton waste material from each fraction respectively (i.e. the difference between impacts related to recycling process compared to virgin material production) as well as the ratio of each waste fraction per ton total household waste is both of importance. The environmental impact will also depend on the source-separation ratio and the impacts related to an alternative treatment of each waste fraction respectively if not material recycled (i.e. landfilling, incineration etc.).

Life-cycle assessment methodology was used for the assessment of the environmental impacts connected to the current management of solid household waste generated at the study-site. Results from the assessment show that the current management strategy with source-separation of household waste as six different types of dry recyclables (packaging and newspapers) and food waste gives a net avoidance of assessed environmental impacts

(global warming, acidification, nutrient enrichment, photochemical ozone formation and stratospheric ozone depilation). Thus, when comparing to a scenario where all waste is incinerated without prior separation, net contribution to negative environmental impacts is turned to net avoidance. The results are however to a large extent dependent especially on assumptions related to the environmental profile of energy carriers used and substituted within and by the assessed systems. Optimizing source-separation behaviour would however result in a further reduction of negative impacts with more than three times compared to the current situation.

The largest potential environmental benefits, on a per ton material basis, are seen in an increased recycling of the fractions metal and plastic packaging. Metal and plastic packaging constitutes only 2.3 and 6.8 weight-% respectively of all waste generated at the study-site, while newspaper constitutes 16.6 weight-%. However, material recycling of all material belonging to the prior fractions result in avoidance of assessed environmental impacts 4.3 and 6.3 times higher compared to material recycling of all newspaper generated at the study-site. At the same time, performed waste composition analyses show that plastic and metal packaging currently are source-separated to a much lower extent compared to newspaper.

Life-cycle assessment methodology was also used for the assessment of the environmental impacts connected to management of organic household waste (food waste) generated at the study-site. It was seen that anaerobic digestion with production of biogas used as substitute of heat and petrol as car fuel is more beneficial than biogas production with use of produced biogas as substitution of heat and electricity, composting in decentralized reactors without cleaning of emissions and incineration with energy recovery (electricity and heat). Also in this case, results are to some extent dependent of the environmental profile of energy carriers produced by compared treatment-systems.

Results from the work suggest that the environmental benefits connected to material recycling of different packaging materials differ from the hierarchy induced by current national recycling objectives. Thus, the thesis raises questions regarding the basis of these objectives. Results from the work also indicate that apart from information towards households regarding solid waste recycling, also the practical arrangements of waste separation should be assessed. There is a need to further develop an infrastructure in which the whole chain of waste source-separation is made convenient and accessible for households and to investigate the effect of such developments on acceptance and participation rates.

ACKNOWLEDGEMENTS

This thesis was made possible through a large group of people. I would like to thank all of them and especially I would like to direct my thanks to the following:

My supervisor Jes la Cour Jansen for encouragement and useful comments as well as a constant cheerful attitude. Åse Dannestam at MKB-fastigheter – thank you for encouragement and many joyful moments. Roger Ekberg at VA SYD for always being there as a problem solver when needed. Stig Edner at SYSAV for many insightful comments and discussions – a living Wikipedia on the topic of Swedish waste management. My co-supervisors Eva Leire and Henrik Aspegren for your useful comments throughout my work. Henrik Aspegren also for his management of the Augustenborg project – always open for new ideas. Sanita Vukicevic at NSR for your help with tasks which are not always attractive. The laboratory staff at VA-teknik – Ylva Persson and Gertrud Persson; thank you for all your kind help and support during these years! Also my dear roomy Viveka Lidström – thank you for support and many laughs. I would also like to thank the staff at the solid waste department at VA SYD, Malmö for their encouragement, interest and help in great parts of this work. Thomas Christensen from DTU and the rest of the 3R-network – thank you for opening an arena for discussion of these topics for me. Lastly, my family and friends for support and encouragement – I hold you all very dear!

ABOUT THE PAPERS

The papers in this thesis treat a number of different topics within the area of solid waste management. The reason for this is mainly the holistic perspective that has been a red line throughout the project within which these papers has been written. Thus, the boarders of the project has not been a specific waste treatment method or waste fraction, but rather a geographical boarder, framing a residential area and the generation, collection and subsequent treatment of the solid household waste from this specific area. The thesis is based on the following papers:

Paper I Local Strategies for Efficient Management of Solid Household Waste – The Full Scale Augustenborg Experiment.
Anna Bernstad*, Jes la Cour Jansen* and Henrik Aspegren**
* Water and Environmental Engineering at the Dep. of Chem. Eng., Faculty of Eng., Lund University.
** VA SYD, Sweden.
Presented at the Sardinia Symposium, 2009, Sardinia, Italy and submitted *for publication to Waste Management and Research, March 2010.*

Paper II Influence of Information Strategies on Waste Recycling Behavior – Evaluation of a Full Scale Experiment.
Anna Bernstad*, Jes la Cour Jansen* and Henrik Aspegren**
* Water and Environmental Engineering at the Dep. of Chem. Eng., Faculty of Eng., Lund University.
** VA SYD, Sweden.
Presented at the Sardinia Symposium, 2009, Sardinia, Italy and submitted *for publication to Waste Management and Research, March 2010.*

Paper III Property close source separation of hazardous waste and waste electronic and electric equipment – a case study
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Presented at the ISWA /APESB World Congress, 2009. Lisbon, Portugal and submitted *for publication to Waste Management, May 2010.*

Paper IV Life-cycle assessment of extended property-close source-separation of solid household waste – a full scale Swedish case study
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* Water and Environmental Engineering at the Dep. of Chem. Eng., Faculty of Eng., Lund University.
Presented at the 25th International Conference on Solid Waste Technology and Management, 2010 Philadelphia, US and submitted *for publication to Waste Management and Research, May 2010.*

Paper V A life-cycle approach to management of organic household food waste – a Swedish case study
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Accepted for oral presentation at Orbit 7th International Conference, 29 of June-1 of July, 2010, Heraklion, Crete, Greece.

ABREVIATONS

FOG – Fat, Oils and Grease
WEEE – Waste Electric and Electronic Equipment
LCA – Life-cycle Assessment
MHW – Municipal Household Waste
SD – Standard Deviation
CV – Coefficient of Variance
BAT – Best Available Technology
GWP – Global Warming Potential
A – Acidification
NE – Nutrient Enrichment
ODP – Ozone Depilation Potential
POF – Photochemical Ozone Formation

PREFACE

This work is based on a project initiated in 2008 by the department for solid waste management in the city of Malmö (VA SYD), Malmö Municipal Housing Company (MKB), the South-Western Scania waste management company SYSAV and Water and Environmental Engineering at the Department of Chemical Engineering at Lund University in Sweden. The aim of the project was to evaluate and develop the Swedish model for solid household waste management with focus on extended property close source-separation of several different household waste fractions. In the project, current household waste streams and waste recycling behavior were to be investigated, but a clear aim in the project was also to influence the recycling behavior both through improved possibilities of property close recycling of new waste fractions and through use of different information strategies towards households. The key perspectives in the project were economic transparency, user-friendliness and increased environmental concerns in relation to solid household waste management. Water and Environmental Engineering at the Department of Chemical Engineering at Lund University contributed to the project in order to monitor and evaluate the current status of the waste management as well as of the effects of changes in this system, taking place during the project period. Evaluations were made mainly in relation to environmental concerns and life-cycle assessments, more specifically the EASEWASTE model (Kirkeby et al., 2006), were used as a tool in the evaluation. As a concrete outcome of this strategy, the usefulness of the EASEWASTE model as a decision support tool for development of environmental benign solid waste management systems was also investigated as a part of the project.

The project was run in the form of a full scale and long-term case study in the residential area Augustenborg in the city of Malmö, Southern Sweden. A project group and a steering comity were formed consisting of VA SYD, MKB, SYSAV and Lund University. Also the waste collector entrepreneur SITA participated in the project group. Meetings with all participating agents were held on regular basis. This created a platform for direct communication between the different agents connected to different parts of the waste management chain. Problems and the often differing interests of the agents could be discussed in order to find sustainable solutions where solving of one problem does not lead to a new in a different part of the chain. Thus in correlation to the life-cycle analysis approach, a holistic approach was sought also on the organizational level of the project.

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1 INTRODUCTION

1.1 Waste management – from problem-solver to resource-provider

Municipal solid waste (MSW) has previously been seen as one of the "darker sides" of modern societies. In many parts of the world, waste and waste management is still connected to negative environmental impact and unpleasant working-conditions (Carvalho et al., 2007). However, waste management has during the last decades developed into a high-tech sector, where waste more and more is seen as a resource rather than a problem. The waste management chain is therefore in many countries becoming more and more integrated as a part of a developing recycling society.

1.1.1 Organization of solid household waste management in Sweden

Sweden has in many ways been a progressive nation in development of waste management. The Ordinance on Producer Responsibility was introduced in Sweden in 1993 for glass and cardboard packaging (SFS, 1994a). This was later followed by similar regulations also for metal, plastic and paper packaging, and newsprint in 1994 (SFS, 1994b; SFS, 2006) and later also the EU regulation of waste electronic equipment (Directive 2002/96/EC).

Management of solid household waste in Sweden is divided between several different agents. This is a result of the introduction of the producer responsibility ordinance on packaging and newspapers put in place in the 1990ies and a municipal monopoly on the management of residual waste, bulky waste and hazardous waste, the latter with the exception of batteries since the 1st of January 2009 (SFS, 2008). Households are required to source-separate producer responsibility materials and make use of developed systems for waste recycling (SFS, 1994a; SFS, 1994b; SFS, 2006). Collection and transportation is in 75% of the Swedish municipalities cared for by private entrepreneurs and in 25% by the municipality (Swedish Waste Management Association, 2009). Different entrepreneurs can also be contracted for collection of different waste fractions. Treatment facilities for packaging and newspapers are in many cases private, whereas treatment of residual waste and bulky waste often is cared for by agents owned by one or several municipalities. As a consequence of a more widespread use of property close source separation of recyclables, Swedish real estate owners also play an increasingly important role in the management of solid household waste in Sweden.

In relation to information, municipalities are responsible for providing households with correct and necessary information regarding management of household waste, including producer responsibility waste materials (SFS 1994a; SFS 1994b; SFS 2006). Producers should have an insight in the production of information regarding the latter fractions. In the case of batteries and waste electronic and electric equipment (WEEE), producers have the full responsibility of providing information to the households (SFS 2000; SFS 2008) (Figure 1).

Waste fraction	Source-separation	Collection and transportation	Treatment	Information
Food waste Residual waste	Households	Municipality (often through entrepreneurs)	Municipality (often through entrepreneurs)	Municipality
Dry recyclables		Producers	Producers	Producers
Waste Electric and Electronic Equipment (WEEE)		Producers (through the municipality)	Producers	Producers
Hazardous waste and Bulky waste		Municipality (often centralized)	Municipality	Municipality
Batteries		Producers	Producers	Producers

Figure 1. Graphic representation of division of responsibilities in the waste management chain.

Thus, the Swedish model for management of solid household waste involves many different agents. This can result in confusion amongst the user of the system – the households – and development of economically and environmentally suboptimal management solutions. As pointed out by Dahlén (2008), when the separate collection and recycling system of the producers does not work – for one reason or another, the local authorities must step in and take care of the waste (Dahlén, 2008). Residents then pay twice: first, a recycling-fee to the producers (included in the price when buying a product), and then the collection and disposal fee to the local authorities.

1.1.2 Objectives in Swedish waste management

The overall target for waste management presented by the Swedish Government in the national waste plan is that; The total quantity of waste should not increase, and the maximum possible use should be made of the resource that waste represents, while at the same time minimizing the impact on, and risk to, health and environment (SEPA, 2008). Waste management is also addressed in one of fifteen national environmental quality objectives adopted by the Swedish Parliament in 1999 (Environmental Objectives Council, 2009). These objectives are to provide a framework for waste management programs and initiatives at national, regional and local levels. The objectives state that the amount of waste being landfilled (mine waste excluded) should decrease with at least 50% till the year 2005, using 1994 as a baseline. Also, 50% of all household waste is to be material recycled, including biological treatment, in 2010 and 35% of all food waste from households, restaurants, industrial kitchens and food stores are to be biologically treated in the same year (Environmental Objectives Council, 2009). More specific objectives for recycling of some of the different fractions of solid household waste have also been stated and are presented in Table 1 (Environmental Objectives Council, 2009).

Table 1. Objectives for Swedish recycling of solid household waste.

Waste fraction	Recycling objective	Recycling level in 2009*
Newspapers	75%	89%
Metal packaging	70%	67%
Paper packaging	65%	74%
Glass packaging	70%	93.6%
Plastic packaging	70%**	30.5%
Food waste	35%	20%***

* Based on data from Swedish Waste Management Association (2009).
** Including 30% energy recovery.
*** Based on data from 2008 (Swedish Waste Management Association, 2009). Includes organic waste from industries.

National environmental objectives normally also have counterparts on a regional and a local level. The objectives should also be reflected in the municipal waste plan, which must be provided by each Swedish municipality since 1991 according to Swedish regulations (SEPA, 2008).

1.1.3 Monitoring solid household waste management in Sweden

Swedish Association of Local Authorities and Regions (2007) has pointed out that more and better indicators need to be developed on the local level concerning waste. Today, few indicators are provided by the Swedish EPA. The only indicators suggested by the Swedish Association of Local Authorities and Regions themselves in relation to waste management in Swedish municipalities is; "The total amount of produced household waste and the ratio of source-separated dry recyclables" (Swedish Association of Local Authorities and Regions, 2007). However, indicators based exclusively on the amount, volume or weight-based, of generated waste give limited information of the actual environmental impact from the generated waste. As an example; according to the Swedish EPA, there is no environmental gain in reducing the amount of waste from the wood industry, as sustainable waste management methods have been developed for this waste fraction and it is many times used to substitute fossil fuel (SEPA, 2007).

Previous studies (Parfitt and Flowerdew, 1997) have stated that there often is a gap between the academic research connected to management of solid household waste, and the local governments and authorities where the management system is being planned. There are reasons to believe that the gap between dwelling owners and the academic world is similarly large. Thus, little guidance is provided for the local actors in their work towards an achievement of national environmental objectives related to waste management and the overall goal stated in the national waste plan. As the local conditions might vary largely from municipality to municipality, especially in a demographically and geographically diverse country like Sweden, more precise and comprehensive indicators and guidance might be difficult to provide from a national level. In order to address local differences and circumstances, other tools might be useful. Life cycle assessments (LCA) have been presented as a useful decision support tool for local authorities and waste management planners (Kirkeby et al., 2006). LCA modeling tools can allow local actors to simulate the importance of different waste management options on the overall environmental impact of the waste management chain in a local context when using site-specific input data.

1.1.4 Waste collection systems

The design of collection systems for solid household waste vary largely in different parts of the world, but also between different regions within a country, and consequently also within Sweden. Recyclables can either be source-sorted by households and collected separately or disposed commingled by households for later post-sorting. The latter system is not in use in Sweden, but feasibility studies show that systems for this exist and could function also in a Swedish context (Stenmark and Sundqvist, 2008). Collection of household waste can be divided into property close collection and collection at drop-off points.

A trend in the Swedish waste management system is an expansion of extended property close source-separation of different waste fractions (Swedish Waste Management Association, 2009). Property close source-separation often includes *some* or *all* dry recyclables (e.g. glass-, paper-, plastic and metal packaging and newspapers), small batteries and, in more and more of the Swedish municipalities, also food wastes (Swedish Waste Management, 2008). Improved accessibility of source-separation possibilities has previously been seen to increase recycling behavior (McDonalld and Ball, 1998; Berger, 1997; Hage et al., 2009). The recycling of producer responsibility materials or dry recyclables has also increased in Sweden during the last years according to national statistics (Swedish Waste Management, 2008). However, national targets are still not met neither for several of the dry recyclable fractions nor for biological treatment of food waste (Swedish EPA, 2009) (Figure 2).

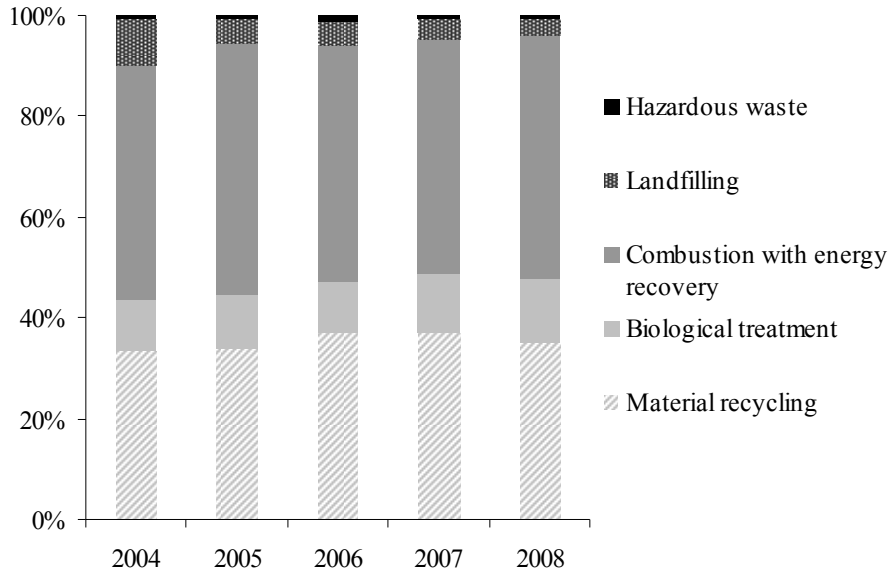


Figure 2. Development in Swedish waste treatment 2004-2008 (Swedish Waste Management Association, 2009).

Also, systems for property close source-separation of other waste fractions than packaging, newspapers and food waste have been expanding in Sweden during the last years. Different systems for property close source-separation of hazardous waste and WEEE are currently tested in several Swedish municipalities (Swedish Waste Management Association, 2009).

New systems for collection of bulky waste are also tested in some Swedish municipalities (Lindgren, 2009). The effects of extended source-separation schemes of new waste fractions and new collection systems area however seldom monitored thoroughly in relation to factor such as environmental impact, working conditions, economy or user-friendliness. Full scale and long term studies are needed in order to increase the knowledge of the effects such systems in terms of environmental and economical costs and benefits.

1.2 Aim and scope of this work

A good understanding of quantities and composition of solid household waste as well as source-separation behaviour is vital for further studies and comparisons between different waste management systems and treatment options and their economic and environmental effects and impacts. Such data is also basic input in discussions on improvement of collection systems life-cycle assessments of solid household waste management and highly relevant when estimating potential use of new investments in waste treatment faclities, such as new incinerators or anaerobic digestion of organic waste.

National statistics on recycling behavior and waste generation are commonly used as input both in life-cycle assessments and feasibility studies for new waste treatment strategies (Kärrman et al., 2005; Borgshed et al., 2003). However, national averages can be associated with high levels of uncertainties and circumstances that can disturb waste stream analyses, such as illegal waste dumping or back yard composting (Dahlén, 2008). Focusing on a well-defined and smaller area improves the possibility of a detailed monitoring of waste streams as sources of error and uncertainties can be minimized. This strategy can also be suitable for monitoring of seasonal variations in waste generation and composition. However, gained results might not be applicable to other geographical areas even in the close surrounding as they might be closely connected to factors such as dwelling-type, socio-economic status etc. (se chapter 4.2 for further discussion on this).

The aim of this project was to gain more profound knowledge of the current performance and possible improvements of the Swedish model for solid household waste management. In addition of gaining knowledge of current waste generation and recycling behavior, the aim was to develop the current recycling system by introduction new source-separation systems for several waste fractions; food waste, hazardous waste as well as waste electric and electronic equipment and to monitor and evaluate the effects of these developments from an environmental and users perspective. In short, the different aims with study can be described as follows:

- Investigate quantity and quality of current waste generation and recycling behaviour.
- Investigate the effect of different information strategies in relation to recycling-behaviour of solid household waste.
- Investigate the effect of introduction new source-separation systems for several waste fractions; food waste, hazardous waste as well as waste electric and electronic waste (WEEE).
- Investigate the environmental effects of changes in recycling behavior through systems analyses.

- Investigate the environmental effects of different treatment alternatives for disposed organic household waste in the study site and to identify hot-spots, i.e. the processes contributing most significantly from the assessed management system.

The focus of the work is a case study in the shape of a long-term and full scale experiment in a residential area in southern Sweden. Using results from the case study, a detailed quantitative and qualitative description of the waste streams is gained (paper I). The work contains long-term monitoring of the introduction of systems for property close source-separation waste fractions that had not previously been source-separated in the study area, including systems for source-separation of hazardous waste as well as WEEE (paper II). Monitoring and evaluation of the influence of oral information in relation to increased and improved recycling behavior of food waste and producer responsibility materials were also performed (paper III). The environmental effects of these behavioral changes are assessed though systems analyses (paper IV). The work also contains a comparative analyze of the environmental performance connected to different types of systems for management of food waste using life-cycle assessment methodology (paper V).

1.3 Study site and case study approach

The case study approach has previously been described as a qualitative approach, method or strategy and is compared with participatory inquiry, interviewing, participant observation and interpretative analysis (Vidish, 2000), grounded theory or action research (Denzin and Lincoln, 2000) or as a method were both qualitative and quantitative analyses can be used to reflect qualitative and quantitative qualities of the studied phenomenon (Åsberg, 2001). In this study, the case study approach is understood as a means of gaining deeper insight in a general phenomenon through the use of qualitative and/or quantitative analyses of data collected within a smaller area or set of individuals. The main difference between a case study approach compared to many other research methods is with this understanding therefore not a matter of qualitative or quantitative approaches. Rather, in a case study, essentially *all* individuals connected to a certain area, group or action (e.g. the *case*), are assessed, whereas, using many other approaches, researchers are struggling to analyze a subset of individuals, supposedly representative for a larger group and through this approach say something about the general phenomenon. However, results from a case study can be said to be applicable only to the specific case.

The study site used in this case study, the residential area of Augustenborg in the city of Malmö, consists of 1631 households in 37 multifamily buildings. By the end of the 1990's, a property close source-separation system for solid household waste was introduced in the area. Since then, waste disposal and collection is concentrated to 13 recycling buildings distributed over the area (Figures 3 and 4). The recycling buildings were provided with locks and each resident had access to only one specific building with the use of an electronic key. Household waste could be source-separated in nine different fractions; glass- (clear and colored), paper-, plastic- and metal packaging, newspapers, batteries and residual waste. Organic household waste could be disposed in decentralized compost reactors, one in each building. Since the summer of 2008, the service in the recycling buildings was extended to provide possibilities of source-separation of waste electronic and electrical equiptment (WEEE) and hazardous waste (HW). Compost reactors were disassembled and a collection system for organic food waste was installed. Households

were encouraged to dispose food waste in paper bags in a special perforated plastic vessel in their kitchen for later disposal in bins in recycling buildings, pretreatment and production of biogas. In the summer 2009, a system for property-close source-separation of fat, oils and grease (FOG) was introduced in 210 of the households at the study-site. However, results from this experiment are not presented in this work. The site was considered to be well suited for a case study as it was geographically well defined and provided good possibilities of before- and after follow ups of the changes in the recycling system introduced in the area during the first years of the experiment.

Figure 3. Exterior of one of 13 recycling buildings at the study-site.

Figure 4. Interior of one of 13 recycling buildings at the study-site.

It has previously been stated that the effects of various kinds of incentives related to recycling of solid household waste seldom are investigated in long-term studies (Schultz et al., 1995). At the same time, knowledge of the persistency of an incentive-program is vital in order to be relevant to public policy making. It has also been lifted that many studies related to recycling behavior focuses on merely *one* type of indicator to reflect the successfulness of an intervention activity, which has been seen as a limitation (Schultz et al., 1995). Often, the chosen indicator is participation rate (percent of households reporting to participate in household waste recycling), but also indicators such as the quantity of source-separated recyclables in comparison to non-separated residuals or the quality of source-separated fractions are used (Schultz el al., 1995). Different interventions techniques could however have different impacts and effect different indicators. Thus, a focus on merely one of them could therefore lead to misleading conclusions. As stated by de Young (1993), different intervention techniques can also be complementary and together play an important role in changing conservation behavior. Another limiting factor in earlier research project in this field is that they often focus on the grade to which households *say* they participate in waste recycling – with little possibilities of controlling actual recycling behaviour. The design of this case study was set to tackle some of the above-mentioned limitation. In order to monitor effects from different types of information strategies regarding household waste separation, households at the study-site were divided into three different groups (Figure 5). Different types of information strategies were used in each group. Both self-reported and actual source-separation of recyclables was studied. Both participation rate and the ratio and quality of source-separation were examined. Continuous weighing and repeated waste composition analyses from the study are throughout a period of 24 months provides information regarding the long-term effects of used information strategies.

Figure 5. The study site with indication of different information strategies. 1. Written and oral information regarding recycling of all waste fractions – 1,001 flats; 2. Written information about recycling of food waste - 210 flats; and 3. Written and oral information regarding recycling of food waste - 420 flats.

The experiment also had an organizational level, as it connected all parts of the waste management chain; from the real estate owner with a direct link to the actual households in the area, via the entrepreneur responsible for collection and transportation of waste and the municipal waste department with the responsibility of information towards households and lastly to the waste treatment company, responsible for further treatment of different waste fractions (the bulk of the collected household waste). Recyclers of dry recyclables (packaging and newspapers) were not included in the project in the same way. This allowed a large part of the many different agents involved in the Swedish household waste management chain to discuss problems and search compromises in line with the objectives of each agent. Continuous reports from the monitoring formed a base for further development of the recycling and waste management system as well as basis for production of information provided to households in the area. However, this multi-stakeholder approach and process and the effects and outcomes of it has not been studied and evaluated on an academic level.

1.4 Definitions

The term *waste management* is in this work used to describe the waste treatment chain in its full extent – from the disposal to the recovery and final disposal. Thus, it includes both households' recycling-behavior, collection, transport and further treatment of household waste. The term *household waste* is in this work used for all waste generated by private persons in their households. *Bulky waste* is used as a term for waste that is not to be disposed as "normal" waste due to its large volume. The term includes for example furniture, bulky WEEE such as TV-sets[1], lager lamps etc. and metal objects such as bikes. If not explicitly stated, the term *household waste* does not include bulky waste. *Dry recyclables* is the term used to describe newsprint and packaging materials (made of paper, glass, plastic and metal) covered by the Swedish Ordinance on Producer Responsibility, with established separate collection and recycling systems. *Recyclables* is used as a common name for dry recyclables, food waste, waste electronic and electric

[1] Management systems for fridges and refrigerators is not discussed in this work as they are not managed by households themselves but by the facility manager and property owner at the study site.

equipment (WEEE) and hazardous waste (HW). The expression *property-close collection* is used to describe curbside collection at single-family houses, as well as collection from the premises of multi-family dwellings. *Property-close source-separation* is used to describe waste collection systems were households can separate their dry recyclables and some times also food waste close to the residence. This system should be distinguished from residential areas where households are directed to *recycling stations* – referring to a non-manned collection points where households can discard dry recyclables and often also batteries. A *recycling center* is a manned facility where households and in some cases also smaller enterprises can bring and discard a variety of household waste, including WEEE, hazardous waste, bulky waste and garden waste. In the case of households, this service is always free of charge. The term *infrastructure* is in this work used to describe the physical equipment used for disposal, collection, transpiration and treatment of solid household waste.

2 METHODS

Methods used for monitoring and evaluation of the current system for waste disposal at the study site and of introduced changes in this system consisted of; continuous weighing of disposed waste, repeated waste composition analyses, questionnaires as well as use of life cycle assessment as a tool for evaluation of environmental impacts related to changes in waste composition, recycling behavior and choices between different waste treatment alternatives. Strengths and weaknesses of used methods are presented and discussed in the chapter "Methodological discussion".

2.1 Weighing of waste

Weights were collected from all household waste disposed in the Augustenborg area during the study period (i.e. 24 months). A balance was installed in the waste collection vehicle in order to collect weight from glass- (clear and colored), paper-, plastic- and metal packaging, newspapers, food waste and residual waste when collected. Weights were registered automatically through a tag with a code on each waste container. Thus, disposed waste in each recycling building was recorded separately for later follow up. In cases when tags were missing or weights for other reasons not were collected, a template weight was calculated based on weights from other waste-containers from the same fraction and same collection date.

Weights were also collected from disposed WEEE, hazardous waste (including different types of light bulbs and batteries) and bulky waste. In addition, weights were collected from waste incorrectly disposed in the area. This waste mostly consisted of bulky waste and larger WEEE-material left in basements, outdoors or on the floor in recycling buildings. This material was taken care of by the property owner and stored in a special building, awaiting transportation to treatment facilities. No weighs were collected from organic waste disposed in compost reactors before the changed collection system for food waste. The average weekly amount of waste disposed in composts was therefore calculated on the basis of records of produced compost (Graham, 2003).

$$OW=((C-SM)*0.5)/52$$

OW = Organic Waste (average amount produced per week as wet weight in kg)
C = Compost (average amount produced in each reactor per year as wet weight in kg)
SM = Structure Material - wood chips (average amount used in each reactor per year in kg)

2.2 Waste composition analyses

Waste composition analyses were carried out at four different occasions during the trial. These occasions where spread out over the year in order to investigate possible seasonal changes in waste generation and composition. Samples used for waste composition analyses where collected from three different recycling buildings, representing 23% of the recycling buildings in the area and 20% of the households in the area. All waste disposed in vessels for dry recyclables and food waste and half of the vessels for residual waste (randomly selected) were analyzed. This approach is recommended by SWA-tool

(European Commission, 2004)[2]. Thus, errors related to sub-sampling were eliminated in the case of dry recyclables and strongly reduced in the case of residual waste. The same waste composition analyze method and analyze team was used at all four occasions. The method is based on the Nordtest Method NT ENVIR 001 (Nordtest, 1995) and developed in cooperation with Luleå University of Technology (Ohlsson, 1998). The used method for waste composition analyses is thoroughly described by Dahlén et al. (Dahlén et al., 2008). Source-separated hazardous waste, WEEE and batteries were not included in these analyses. Both residual waste, food waste and dry recyclables were diverted into 8 main categories and 17 sub-categories (Table 2). This makes it possible to monitor missorted material in source-separation fractions not only by mass, but also on a qualitative basis.

Table 2. Used categories in conducted waste composition analyses (based on Dahlén et al., 2008).

Main category	**Sub-category**
Biodegradables	Food waste
	Garden waste
Paper	Newspaper
	Paper packaging
	Corrugated cardboard
	Other paper
Plastics	Plastic packaging – solid
	Plastic packaging – soft
	Other plastics
Glass	Glass packaging colored
	Glass packaging clear
	Other glass
Metal	Metal packaging
	Other metal
Hazardous waste	Non*
WEEE	Non *
Others	Textiles
	Wood
	Dipers, hygiene products
	All other

* See section 2.3 for further analyses of these fractions

2.3 Waste composition analyses of bulky waste, source-separated WEEE and hazardous waste

In the case of bulky waste (including bulky WEEE, metal waste and other bulky waste disposed in mobile waste containers[3]) and WEEE (source-separated in WEEE-cages in recycling buildings), a total weight of all disposed material within these fractions was collected when disposed waste was transported from the area. However, no separate weighing of each piece of disposed bulky waste or WEEE was performed to give information about the different types of materials disposed within these fractions.

[2] The SWA-tool recommends that all waste disposed during a certain time frame should be analyzed, without further sub-sampling.

[3] Each month, households at the study site were given the possibility to dispose bulky waste, including bulky WEEE and metal objects in a mobile container in the area during one afternoon for three hours.

Qualitative monitoring of source-separated WEEE was made in a total of fifteen filled WEEE-cages. All source-separated WEEE was registered and template weights (FRN, 2009; Huisman et al., 2008; SWICO Recycling Guarantee, 2006) were used to define the composition of bulky WEEE items. Smaller WEEE items were weighted individually. All items (but light bulbs and low energy light bulbs) observed in the waste composition analyses of source-separated WEEE were registered individually. Light bulbs and low energy light bulbs were registered aggregated on a weight basis and template weights were used in the definition of the total quantity of disposed waste from these fractions. No qualitative analyses were made on bulky waste incorrectly disposed inside recycling buildings in the area. However, the total weight from this fraction was recorded.

In the case of hazardous waste, non source-separated as well as source-separated items were classified into 12 different categories (Table 3) and weights were collected from each one of these. Also quantitative and qualitative analyses of non source-separated WEEE observed through waste composition analyses were performed. Items were classified into 8 different categories (Table 3) and weighing of each peace of incorrectly disposed item was performed.

Table 3. Used categories in waste composition analyses for non-source-separated hazardous waste and WEEE.

Main category	**Sub-category**
Hazardous waste	Paint (water based)
	Paint (non water based)
	Aerosols
	Solvents
	Cleaning agents
	Other chemicals
	Make-up
	Lighters
	Pharmaceuticals
	Thermometers
	Fire-extinguisher/Car batteries
	Motor oil
WEEE	Larger electric household devices*
	Electric toys
	Cables
	Smaller electric devices**
	Light bulbs
	Low energy light bulbs
	Fluorescent lamps
	Un-used EEE

* Coffee machines, larger loud-speakers, Hi-Fi equipment etc.
** Cellular phones, alarm clocks, telephones etc.

Figure 6. Cabinet for disposal of HW. Light bulbs, low energy bulbs and fluorescent lamps are disposed in open vessels on the side of the cabinet.

Figure 7. Cage for disposal of WEEE.

2.3.1 Applied indicators for evaluation

Four different indicators were chosen in order to monitor and evaluate waste flows and current status of the recycling scheme in the area. All indicators but the participation rate were calculated on a wet weight basis.

- The specific waste generation [kg household^{-1}, year^{-1}] was determined based on weekly average from 24 months weighing of waste.

- The source sorting ratio [weight-%], defined as the weight of collected source-sorted recyclable material in relation to the sum of the same material – sorted, incorrectly sorted and unsorted – in disposed waste.

- Ratio of miss-sorted waste [weight-%], defined as the weight of incorrectly source-sorted recyclable materials in relation to all waste disposed in the particular fraction.

- Participation rate [%], defined as the self-reported participation in recycling of food waste.

2.4 Questionnaire

A questionnaire was sent to all households in the area 16 months after the beginning of the experiment. The questionnaire focused on the source-separation of food waste, hazardous waste and WEEE. It contained questions regarding the extent of participation in the recycling scheme and the reasons for not participating (if they were not). The questionnaire also contained questions regarding accessibility of waste recycling as well as attitudes towards environmentally related issues in general and wastes recycling in particular.

2.5 Life-cycle assessment methodology

The life-cycle assessment (LCA) tool was originally developed for assessment of product systems, but has shown to be amenable also to the study of processes, such as waste

management systems. Life-cycle assessments of different types of solid waste management alternatives are commonly used as decision support tools and a number of different software assessment programs have been developed over the years; the IWM-model (White et al., 1995; McDougall et al., 2001), ORWARE (Dalemo et al., 1998) and EASEWASTE (Kirkeby et al., 2006) to mention some. Though these models allow advanced management system comparisons, there is commonly a lack of knowledge in some of the very basic input data in the modeling, such as a well-documented description of the waste composition, source-separation ratio and ratio of miss-sorted material in recycling-fractions. Data gained from a thorough monitoring of a full-scale case study is therefore valuable in the environmental evaluation of the system.

According to the LCA standards, ISO 14040, developed in 1997, the LCA consists of four separable phases:

- Definition of the goal and scope of the study (ISO 14041, 1998).
- Inventory analysis of inputs and outputs from all processes that form part of the product's or system's life-cycle (ISO14041, 1998).
- Impact assessment, where results from the inventory are used to prepare environmental impact and resource consumption profiles for the system (ISO 14042, 2000a).
- Interpretation of the impact profile and resource consumption according to the defined goal and scope of the study. This phase includes sensitivity analysis of key elements in the assessment (ISO 14043, 2000b).

Above stated phases are to bee seen as theoretically separable and subsequently following each other. In practice, the LCA is an iterative procedure, where information gathered in one phase can make the user realize the need for a modification of earlier phases (Hauschild, 2006). LCA-methodology is aiming towards a holistic approach on the assessment of the system in question, where the aim is to include all relevant environmental impacts. However, only five different types of environmental impacts are addressed in this work: global warming potential (GWP), acidification (A), nutrient enrichment (NE), photochemical ozone formation (POF) and stratospheric ozone depilation potential (ODP). These categories were chosen as they are environmentally relevant and internationally accepted, in accordance with ISO 14042 recommendations (ISO, 2000a) and believed to be relevant for identification of key issues on a global/regional scale (Lindfors et al., 1995).

2.5.1 Description of the EASEWASTE model

Life-cycle assessments were in this study performed using the EASEWASTE-software developed at DTU, Denmark (Kirkeby et al., 2006) (Figure 8). The model is a framework where the user to a large extent is free to define input data for assessment of a specific waste management system. Large part of the input data can be user defined and thereby site-specific[4] and provide reliable assessments of well defined waste management systems.

[4] When using the word "site-specific" in this work, it refers to the input data and not a specific geographical area to which the environmental impacts of the management systems are allocated. The latter is the use of the same term by Hauschild and Potting (2004).

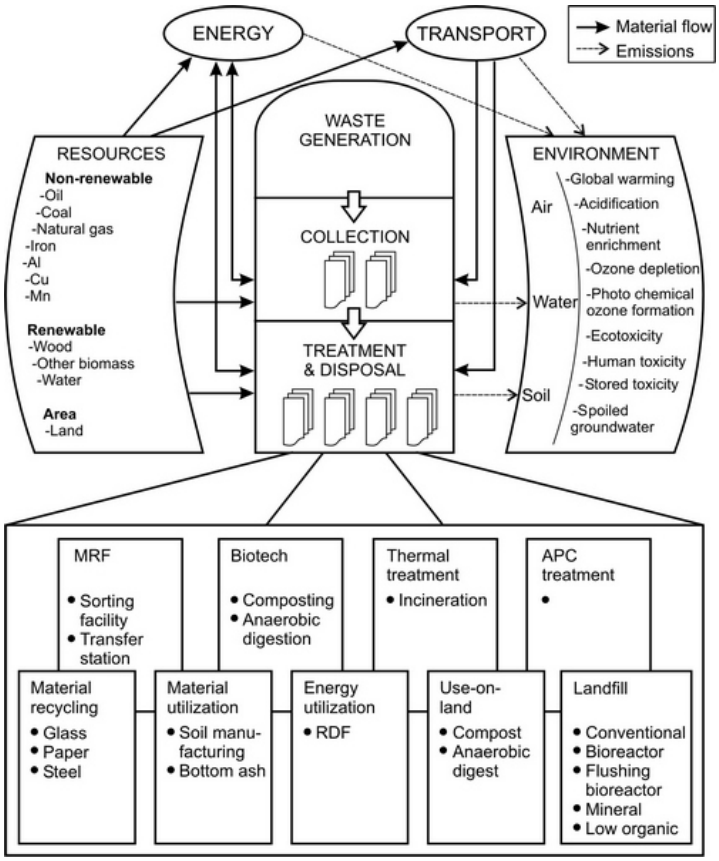


Figure 8. Graphic model of the EASEWASTE LCA-tool (Kirkeby et al., 2006).

In EASEWASTE, resources as well as emissions to air, water and soil are calculated and aggregated into different types of environmental impacts, using standardized impact-factors, such as CO_2-equivalents for green house gas emissions. The EASEWASTE-model is built on the EDIP 1997 method (Wenzel et al., 1997) in which results are given at four different levels:

- life-cycle inventory
- characterization of impacts
- normalized impact profile
- weighed impact profile which applies political reduction targets

Use of normalized impact profiles gives an idea of the size of the LCA-results, as the numeric results are related into average yearly contribution to environmental impact from one person in a specific geographical region. This helps to interpret the result from the LCA and is done by using person equivalents (PE). The person in question can be either an average Swede, German, Dane as well as European or global citizen. In the present study, environmental impacts with relevance on a global scale (i.e. GWP and ODP) were presented as PE (global citizen) and impacts with relevance on a regional scale (i.e. A, NE and POF) were presented as PE (EU-15 citizen), in both cases with 2003 as reference year (Standdorf et al., 2005).

2.5.2 Functional unit and system boundaries

The functional unit is a definition of the service provide by the product or process examined in an LCA. In a waste management system, the functional unit is typically the collection and further management of one ton of waste or the total amount of waste or waste of a particular character disposed by a certain amount of households during a chosen time frame. In this work, the latter definition was chosen, as it gives more room for investigation of household waste recycling behavior.

A clear definition of system boundaries in a LCA is essential for the reliability, transparency and quality of the assessment. The system boundaries in the EASEWASTE model are defined by the waste management system from waste generation and source-separation to final disposal of the waste residuals, where they become inert and no longer contributing to further environmental impacts (Kirkeby et al., 2006). The model includes the processes waste collection and transportation, treatment and final disposal. The model also includes the impact of any external energy and material displaced by goods produced within the system. System expansion is used to represent lesser need for production of virgin material and energy due to material and energy recovery. Avoided external productions and non-waste based processes are included as avoided resource consumptions and emissions (Kirkeby et al., 2006). Resulting environmental impacts can be assessed by applying different life cycle assessment methods. In this case, the EDIP 1997 method (Wenzel et al., 1997) is used.

It is important to define system boundaries also in time. When comparing different waste management techniques it is vital to address the fact that some processes can have the overall environmental impact (both negative and positive) during a short time period, whereas with other techniques the impact needs to be considered during a much longer time-period. An example can be used to describe this. Material recycling can result in use of electricity and auxiliary materials during the actual processing of the recycled waste and benefits can be seen directly as recycled materials substitute's virgin materials. Contrary to this, environmentally hazardous emissions of gas and leachate can be produced from the same material during several decades if the material was instead to be disposed on a landfill. Gas extraction from a landfills with municipal waste may continue during a period of 30–40 years (Barlaz et al., 2003; Kjeldsen and Christophersen, 2001), while leachate from a landfill may continue for thousands of years (Astrup et al., 2006). Thus, time frames where this is taken into consideration must be established when comparing different treatment techniques.

2.5.3 Used input data

The life-cycle analyses in this work are based on site-specific data to an the widest possible extent. However, given the limited time and resource for data-collection, data from previous research findings were also used as supplement. Datasets describing energy/resource use and emissions related to material recovery, including biological treatment, are to a large extent site-specific and based on environmental reports and personal communication with waste recovery/treatment facilities. Datasets describing energy and resource use and emissions related to external processes such as production of virgin materials (plastics, cardboard etc.) were collected from producers and branch organizations. Datasets describing energy and resource use and emissions related to production and use of gasoline or fuel oil, were collected from LCA-databases such as

SimaPro, ECOINVENT and literature. These datasets illustrate environmental impacts connected to average production methods and are in some cases also describing processes within a specific region, such as Europe. In these cases, the most likely geographical region was chosen.

3 METHODOLOGIES DISCUSSION AND UNCERTAINTIES

3.1 Sources of errors related to monitoring of waste streams

In performing waste composition analyses, there is a risk of committing several types of sampling errors. Pierre Gy in Pitard (1993) list seven different types of such errors; *long-range heterogeneity fluctuation error* (representativeness of a sample for a larger area), *periodic heterogeneity fluctuation error, fundamental error* (the size of the sample is to small to account for differences in physical character in the analyzed material), *grouping and segregation error* (the sample can be misleading due to uneven distribution of variation in the material), *increment delimitation error* (incorrect division of material into sub-samples other than the ones were the mass center is allocated), *increment extraction error* (material from a sample is lost during sub-sampling and analyzing) and *preparation error* (changes occurring during the analyze period is affecting the sample) (Pitard, 1993). Other possible sources of error are lacking information regarding waste dumping, yard composting and a lack of control over who has access to waste disposal in a specific area.

Long-range heterogeneity fluctuation error

Some of the earlier mentioned sources of sampling errors, such as *grouping and segregation* and *increment delimitation errors* are minimized as no physical sub-sampling were performed before the waste composition analyses performed in this work. However, an underlying assumption in the work is that the initial sub-sample of roughly 20% of the examined households is representative for the study-site as a whole. The risk of committing long-range heterogeneity fluctuation errors can be addressed through further division of an area into sub-areas and samples from each sub-area for analyze. These samples could either be aggregated into one sample or analyzed individually.

Periodic heterogeneity fluctuation error

Seasonal variations could affect both waste composition and total waste generation. In order to reduce the risk of committing such errors, waste composition analyses can be performed continuously over the year, as was done in this work. It has also been recommended that each analyzed sample should cover at least one full week, since the waste generation during weekends is different compared to weekdays (Dahlén, 2008). This was done also in the present work.

Fundamental errors

The amount of waste analyzed in the waste composition analyses must be large enough to account for the heterogeneity in the disposed waste. So, how can it be determined what is large enough? The question is even trickier to answer when we are investigating an area which is target for changes in the recycling system and some differences in source-separation ratio and ratio of missorted material could be expected. However, the total amount of waste of different fractions can be expected to remain rather stable, although the manners of disposal might vary over time. This can be investigated in order determine the representativeness of the samples.

Previous studies have developed both absolute and relative criteria's for sample sizes when performing waste composition analyses. The amount of analyzed material should be above 450-950 kg according to Petersen (2004) and above 5% of the population or 100-200 households according to Nordtest (1995) for assessment of specific waste generation and composition analysis respectively. Some fractions are, relatively seen, more sensitive to fundamental errors than others. If the number of objects belonging to a specific waste category is small, the relative impact on the estimation of occurrence of waste belonging to a specific category and the risk of committing fundamental errors increases. Hazardous waste and WEEE can be used as examples. Samples in waste composition analyzes performed in this work consisted of more than 1640 kg household waste, more than 320 (100 for sub-samples), equal to more than 20% of the households in the area, the coefficient of variance between analyzed sub-samples was still large for these fractions over the whole study-period; 29-90% for hazardous waste and 51-74% in the case of WEEE, compared to 14% for food waste and 9% in the case of cardboard-packaging (paper I; paper III).

In order to reduce the risk of committing fundamental errors one can either increase the sample size or increase the homogeneity of the waste. If the latter result in a determination of the waste composition on a chemical basis, it is not recommendable for determination of source-separation ratio and ratio of miss-sorted material in the type of recycling system investigated in this work. Increased sample sizes are therefore a more viable option. Thus, in order to determine the total amount and source-separation ratio of less abundant waste fractions such as WEEE and hazardous waste in solid household waste, recommendations presented by Petersen and Nordtest (above) might not be valid.

The design of this study made it possible to make a comparison between results from waste composition analyses performed in waste from 20% of the households in the area and results from weekly weighing of waste disposed in the area (Table 4). The coherence (C) between results from the two methods was determined:

$$C\ (\%) = W_{WCA} / W_{W} * 100$$

W_{WCA} = Weight (waste composition analyses, kg household^{-1} week^{1}
W_{W} = Weight (weighing of disposed waste, kg household^{-1} week^{1}

Thus, a coherence of 100% implies an absolute coherence between the two methods for monitoring of waste streams.

Table 4 Coherence between results from weighing of waste and waste composition analyses.

Fraction	*Based on weekly weighing of waste (n=104)*		*Based on waste composition analyses*		*Coherence (%)*
	Average (kg household^{-1} week^{1})	*CV (%)*	*Average (kg household^{-1} week^{1})*[2]	*CV (%)*	
Food waste[1]	0.7	57.0	0.8	14.5	86
Dry recyclables	3.0	15.2	2.7	13.5	113
Residual waste	7.8	11.0	7.6	10.3	103
WEEE and hazardous waste	0.2	-	0.2	-	
Total	11.8	9.1	10.9	9.5	108

[1] Not including food waste disposed in compost in the first waste composition analysis.
[2] Based on four waste composition analyses in the case of metal, plastic, paper packaging and food waste, three waste composition analyses in the case of and newspaper and two in the case of glass packaging.

Table 4 indicates that there is an incoherence of 7% between the total amount of waste when monitored through waste composition analyses and through weekly weighing of waste. In the case of food waste, values based on waste composition analyses were higher compared to weight-data, whereas the relation is reversed for dry recyclables and residual waste. Results imply that chosen recycling buildings were representative to the overall waste composition at the study site. There can be several explanations to this observation. The waste composition analyses were performed on waste from 20% of the households at the study site. These households might not be representative for the area as a whole, although no reason for this could be thought of. Another explanation could be evaporation from moist waste fractions during the waste composition analyses. This would however create a situation where the amount of food waste would be higher according to waste composition analyses compared to weighing – thus opposite to what is seen in Table 4. It was seen that the variation (demonstrated as the coefficient of variance, i.e. the standard deviation divided by the average) was high in the case of food waste. A large variation would increase the risk of performing a waste composition analyze whit non-representative amounts of source-separated food waste. This would however not explain the incoherence amongst dry recyclables. No similar studies were found where investigations of household waste generation were made both through weighing of disposed waste and waste composition analyses and it was therefore not possible to relate incoherence found in the present study with earlier findings. As incoherence between the two methods where less than 15% in all assessed waste fractions, the difference is regarded as of minor importance.

*Increment extraction error*s

Increment extraction errors have in earlier studies been discussed in relation to sub-sampling, as this procedure can result in a loss of particles and subsequently influence results from the analysis (Dahlén, 2008). However, similar effects can arise even without sub-sampling due to losses of different types of contaminations and humidity in analyzed material. Previous studies have shown a potential weight reduction of 35% for metal packaging and newspaper, 40-45% for plastic and paper packaging and 5% for glass packaging if the material is cleaned from contamination (mostly consisting of food waste) and air dried at room temperature (Swedish Waste Management Association, 2005). In this study, only visible food rests were removed from packaging before weighing,

analyzed waste was not dried and no correction factors were used. As weight gains related to moister and dirt can be assumed to be more common on non source-separated waste, this creates a risk of overestimation of the recycling potential at the same time as the source-separation ratio and ratio of miss-sorted materials in dry recyclables can have been underestimated. Assuming that the weight losses presented above are valid for non-source-separated waste but not for source-separated waste could have an effect on the source-separation ratio. The overall source-separation ratio amongst dry recyclables would increase with 13.6% and reach an average of 68% (paper I). In the case of the ratio of miss-sorted material in food waste, an overestimation is however more likely, as the weight of miss-sorted paper, cardboard, plastics etc. is increased by the source-separated and wet food waste.

Diversion of food waste from residual waste can result in a reduced weight of non-source-separated dry recyclables such as newspaper and paper packages in residual waste as the source-separation will decrease the humidity in the residual waste and thereby also the possibility of these fractions gaining weight. This could be misinterpreted as an increased source-separation behavior of dry recyclables in areas where source-separation of food waste has been introduced, when in fact only the determination of the source-separation ratio of these fractions has become more accurate.

There are also other methodological difficulties connected to the evaluation of different source-separation schemes for food waste. If food waste is source-separated in paper bags, households can be advised to leave wet food waste such as potato peels etc. to lose some of the water content before disposed in paper bags and households were also encouraged to use a special vessel to increase air circulation around the bag and reduce risks for anaerobic condition and production of mould. Moister is also lost after paper bags are disposed in bins for food waste in recycling buildings. Thus, water and thereby weight is lost from source sorted food waste at three occasions in the waste collection chain and there can be a need to adjust results from waste composition analyses in order not to reach misleading conclusions regarding source-separation ratio and ratio of miss-sorted material in source-separated food waste. An assessment of this impact show a weight-loss of 14% in food waste source-separated in paper bags compared to disposal of food waste in plastic bags. It was however observed in the waste composition analyses that a large part of the households adds a sheet of newspaper or tissues in the paper bags fro food waste. This would naturally decrease some of the evaporation from the food waste as the paper would absorb some of the moister-content. The importance of this absorption was not included in the assessment (paper I).

Preparation error

Preparation error can be defined as problems arising when there is a lack of control of the sampling situation – for example changes in collection routines due to holidays which lead to miss-matches and lacking representativity of samples. One of the main aims of the project was to monitor and evaluate changes in the source-separation system in the area occurring during the experiment period. Thus, such changes are not to be regarded as preparation errors. However, other changes might have influenced the results. One example is the constant change of people living in the area. The turnover rate at the study site was determined to around 500 persons per year, corresponding to more than 15% of the total population in the area (City of Malmö, 2007). Previous experiences of waste recycling,

level of environmental awareness, language skills and similar amongst in- and out-moving households might have influenced the results from waste composition analyses and weighing of waste during the study period. The turn-over might also have been unequally spread in the area and thereby affect the comparison between sub-groups receiving different types of information regarding waste recycling (paper IV).

Other sources of error

The risk for errors due to waste dumping, yard composting and lacking control over who has access to waste disposal at the study-site are minimized through the design of the study, as the site is well known in terms of waste dumping and occurrence of home composting. The fact that only households with the correct keys have access to the different recycling buildings increases the possibility of evaluating information strategies used in the area. However, there is a possibility that households associated to smaller enterprises could make use of the recycling buildings in order to dispose material generated within the enterprise. Also, there is a risk that tenants who have not received oral information regarding source-separation of different waste fractions will receive the same type of oral information from their neighbors.

3.2 Uncertainties connected to weighing of waste

Due to technical problems, weight from disposed waste was in some cases not collected. This was mainly handled with the use of template weights (described in chapter 2). Template weights were used in 15% of the cases.

3.3 Uncertainties connected to questionnaires and focus groups

The representativity aspect is often a difficulty when using questionnaires as well as focus groups. There is a clear risk that the persons replying to a questionnaire or involving in a focus group have a larger interest in the question at stake than the general public and their opinions or habits might therefore not be representative for the larger population. In an attempt to avoid this situation in the present work, a lottery was arranged where all households participating in the questionnaire had the possibility to receive a small gift. The aim with the lottery was to encourage also less engaged households to participate and reply to the questionnaire. There have been no possibilities to assess the influence of the lottery and to what extent this contributed to a larger representativity.

3.4 Limitations related to life-cycle assessment methodology

The broad system perspective makes LCA a powerful tool for environmental comparison of different options for waste management and the tool has gained wide acceptance as a part of waste management planning and policy-making. However, as pointed out by Ekvall et al., (2007) it is highly important for both users of the tool and those who take part of the results to know that the environmental information it generates is neither complete nor absolutely objective or accurate; a life-cycle assessment always requires a drastic simplification of the complex reality (Ekvall et al., 2007). Thus, an LCA is always connected to a large amount of uncertainties (USEPA, 1995; Björklund, 2000; Ekvall, 2007) and although this has been recognized long since, there is still a lack of consensus regarding methodologies for determination of data quality and investigations of the range and impact of uncertainties (Björklund, 2000). Vigor and Jensen (1992) cited in USEPA

(1995) present 11 different types of uncertainties that can be further categorized into three different categories, after Björklund (2000):

Knowledge-based uncertainties – caused by lack of knowledge or large variations in presented data; for example the N_2O-release and nitrogen substitution-ratio from different types of organic fertilizers compared to chemical fertilizers.
Methodological uncertainties – caused by choices made by the user of the LCA-tool; for example substitution of average or marginal electricity and heat by waste generated energy.

Dynamic uncertainties – temporary and spatial variability; for example factors that can be dependent on political decisions or economic variations, such as the feasibility of using digistate on farmland or technological development with impact on the environmental profile of substituted energy, fertilizers or other materials.

Providing the readers with information needed to reflect upon these uncertainties and performance of sensitivity analyses in order to investigate the importance of presented uncertainties and made assumptions will reduce the risk of a false sense of security and accuracy in the study.

3.4.1 Type and quality of input data

An LCA is in many ways only as good as the data used in the modeling render possible. A problem when combining use of different types of input data in an LCA – as is done in this work – is the risk of applying different levels of accuracy and completeness for different parts of the waste management system, which could give misleading results. The user of the LCA-tool might be tempted to use as detailed and extensive data as possible for some processes, while she will have to be satisfied with using much less complete data for description of other processes.

One way of handling this difficulty is to use the least detailed and complete dataset as reference and scale down all other data to the same level of accuracy. However, in many cases, the importance of data varies between different processes in the waste management chain and the accuracy of data can therefore be of different importance in different parts of the assessment. Another way of addressing the uncertainties in quality differences is performance of sensitivity analyses. If the result of the assessment does not vary largely with use of different datasets for a specific process, the quality of used data for this process is not to be seen as vital for the overall result from the assessment. Sensitivity analyses can also be used to identify key elements in a process, with extensive contribution to the overall environmental impact. The user can then focus on gaining better data on these key elements to improve the overall quality of the assessment.

Data on the environmental impacts related to material and energy recycling used in this work has to a large extent been based on data from environmental reports from recycling facilities and producers of virgin materials. In a previous investigation of the possibility to use environmental reports to make a life cycle assessment it was concluded that the report alone not was sufficient (Erixon and Ågren, 1997). The evaluation was based on reports produced in mid-1990 and the primary problems were related to lack of data in the reports. It is possible that the general quality of environmental reports have increased

since the performance of the previous study (Magnusson, 2010). Also, due to the limited number of environmental impact categories assessed in the present study, there was no need for high-quality information regarding for example emissions of toxic substances. The largest quality deficit in used information is probably related to the use of auxiliary materials and the environmental impacts related to these.

A comparison between used site-specific data on recycling processes and the environmental profile from material recycling found in LCA-databases show that site-specific data can represent both more and less environmentally burdening recycling processes compared to data found in LCA-databases. The largest differences are seen when comparing recycling of steel and newspaper. Using LCA-data would decrease total environmental gains with 50% from the current recycling at the study-site (Table 5).

Table 5. Comparison between used data on recycling processes (site-specific) and examples from previous assessments of the same processes.

Recycling process	PE ton recycled material^{-1}*	Reference	Difference (%)
Reprocessing of aluminum, SE 2007	0.07	Stena Aluminium, Environmental report (2007).	
Aluminum scrap to reprocessed aluminum (remelting) Europé, 1993	0.14	Schmidt and Strömberg (2006).	94
Reprocessing of steel, SE 2007.	0.10	OVAKO BAR, Environmental report (2007).	
Steel scrap to steel sheets, DK 1992	0.29	EDIP, IPU-MSH-M0008	185
Recycling of paper packaging, SE 2006.	0.06	Stora Enso Fiskebybruk, Environmental report (2006).	
Recycled Paper to cardboard, SE 2005	0.03	Graphic Packaging International Sweden AB (2006); Schmidt and Strömberg (2006).	- 46
Plastic recycling, SE 2007.	0.01	SWEREC, Environmental report (2006).	
PE (Polyethylene) "Pure" – Regranulated	0.02	EDIP, IPU-NF-B2440 (based on Replast A/S environmental report, 2000).	68
Newspaper recycling, SE 2007.	0.03	Stora Enso Hyltebybruk, Environmental report (2007).	
Paper (Newpaper and magazines) to Newspaper, Generic, 2001	0.06	Frees et al., (2005); Schmidt and Strömberg (2006).	108
Glass recycling, SE 2008.	0.14	Ardagh, Environmental report (2008).	- 8
Glass cullet to new products (remelting), Europe, 1990	0.13	EDIP, M32362T98	

* Expressed as the sum of A, NE, POF, ODP and GWP per ton recycled material.

3.4.2 Substitution of goods

When a material is recycled or waste is used for energy production, assumptions have to be made regarding the capacity of these outputs to replace virgin materials and energy produced by other means than through waste treatment. These assumptions can be made in more or less advanced manners in regard to for example price elasticity in replacement

of virgin materials and marginal production of electricity and heat. Made assumptions regarding the substitution grade and qualitative factors describing the replacementness of re-used materials (in the case of material recycling) and the type of material/energy-production substituted by the applied waste management system can be crucial for the overall environmental profile of a waste management process. Sensitivity analyses can be of use in order to test the assumptions made in the study.

There is an ongoing discussion regarding the choice of using a marginal or average energy-perspective in performance of life-cycle assessments, and different choices can be more beneficial in different situations (Elforsk, 2006). Average production is often used in attributional analyses, whereas marginal production could be more reasonable to use in consequential analyses (Furegaard et al., 2009). A marginal perspective, often assuming a substitution of coal condense power, has previously been used in several LCA-studies similar to the present (Eriksson et al., 2005; Assefa et al., 2005).

Even after distinguishing and choosing between using an average or marginal energy substitution approach, the frames for either of these systems must be defined. Swedish average electricity is a result of production from mainly hydro and nuclear power together with combined heat and power, oil condensing power, gas turbines and wind power (Rydh et al., 2002). However, electricity used in Sweden can have its origin in any of the Nordic countries, as the grid systems are connected through Nord Pool (nordpool.com, 2009). Therefore, it can be discussed whether production of energy from waste incineration, landfill gas or biogas should be assumed to substitute Swedish average electricity production or a Nordic average. It has been shown that country specific average data on electricity production in European countries vary internally up to 160 times, where Norway represented the lowest GWP with 0.007 kg CO_2-eq/kWh and Poland the highest with 1.13 kg CO_2-eq/kWh (Mathiesen et al., 2009). Thus, the geographical framing of the average can be of large importance.

If using a marginal approach, the production of the marginal electricity has to be identified. Marginal electricity can be dominated by extended use of old coal-power plants, a postponed closing of Swedish and German nuclear reactors, the construction of new CHP plants for natural gas, etc. These have earlier been denoted as long-term marginal effects (Weidema et al., 1999). In the shorter perspective, an increased production of coal power in existing Danish, German or Finnish power plants is a viable marginal power production in a Swedish context (Elforsk, 2003). In reality, the origin of marginal electricity and the environmental profile connected to this will be more complex and consist of a mixture of coal, wind, bio and hydro power – where the level of contribution will depend on available supply and price, factors in their turn affected by political decisions. Use of historic data for determination of the environmental profile of marginal electricity is therefore not always advisable (Elforsk, 2003). The identification and choice of marginal technologies for electricity production can be of large importance as previous studies have shown that the GWP from country specific marginal technologies on electricity production in Europe can vary up to 400 times (Mathiesen et al., 2009). According to previous studies, the GWP from marginal electricity can vary largely from year to year also within a country (Elforsk, 2003). The average-approach used in this work was evaluated through sensitivity analyses, where average energy production was compared to marginal technologies with low and high environmental impacts respectively, a strategy suggested by Mathiesen et al., (2009).

The discussion above is to a large extent valid also for the origin of electricity used in the waste treatment chain as well as for the substitution of thermal energy. However, in less data is available regarding the environmental impacts of national average district heating. The most recent example of such data collection located is Halldorssonar (2001), base on data from 1999. As many changes in the Swedish production system have been made since then, it was preferred to base default thermal energy on current production methods in the region where the case-study was performed. The input energy to district heating systems differs in different regions, and the impact from waste incineration can therefore differ from one city to another. With the objective of determine site-specific environmental impacts, it could be suggested that the thermal energy substituted by waste incineration is equal to the current energy mix. Data was therefore based on regional sources (County of Scania, 2008). The used data was not available in the form of primary energy carriers and therefore the environmental impact was assessed through a calculation tool (Effektiv, 2009). This strategy can be said to decrease the reliability but was the only method available in order to reflect actual regional conditions.

3.4.3 Non-assessed environmental impacts and other limitations

Several factors were not included in the performed life-cycle assessments. The use of water for rinsing of packaging before source-separation, construction of waste bins, collection vehicles or recycling buildings and treatment facilities were all excluded. Also, only five different types of environmental impacts were addressed in this work: global warming, acidification, nutrient enrichment, photochemical ozone formation and stratospheric ozone depilation. Thus, results of the assessment are far from holistic enough to be used as basis in policy-making or municipal waste planning. The choice of using only these five categories can be seen as the result of a compromise. Including more categories such as eco toxicity and human toxicity would make the study more holistic, but as the level of uncertainty in data related to these categories and evaluation of the same tends to be high, this would also increase the level of uncertainties in the work as a whole.

The discussion above is related to a general difficulty in use of LCA, namely the problem of addressing environmental impacts that are difficult to translate into numeric values. An example is the difficulty of addressing resource depilation – such as use of non-renewable energy or minerals or long-term degradation of farmland, and contribution to toxicity – both eco- and human – in the environment. Also noise, odor, working conditions and soil quality are more difficult to address. Attempts to address these issues have been made through monetized environmental burdens in cost-benefit analysed. However, also such evaluation-systems have been criticized due to a limited scientific basis of such monetization (see for example Stirling, 1997). In addition to the lack of completeness in addressing different environmental conditions, other non-environmental aspects such as time-consumption, working conditions and creation of jobs are commonly left outside of the scope of the LCA of the waste management system, but can be very vital to address in the real situation. Users of LCA must therefore keep in mind that the model is a decision-support tool and not an objective decision-making tool and that other concerns such as economic resources, ethical issues and social willingness must be considered when making decisions related to solid waste management (Kirkeby et al., 2006).

Another general limitation of an LCA-approaches is that it seldom takes geographical aspects into consideration and therefore do not reflect site-dependent environmental constraints (Hauschild and Potting, 2004). This implies that the results of the LCA do not consider the particular qualities of the area in which the calculated emissions from the system in focus are released. Due to various reasons, some sites can be more sensitive to certain emissions than others. When using the results in the LCA, the decision-maker can have to choose between options with different types of environmental impacts, such as higher contribution to either global warming or nutrient enrichment; eg. negative environmental impacts on a global versus a regional/local level. In these cases, it can be of great importance to have information about local conditions and the possible effects of different emissions in the particular area in question. Site-dependent approaches have been integrated in some recent LCA applications, such as EDIP 2003 (Hauschild and Potting, 2004). However, the results of such integrations can nevertheless leave the decision-maker with a difficult choice; is it more beneficial to allocate an increased environmental burden to areas where the environment already has been highly impacted by human activities, or should previously untouched areas rather be exploited?

The accuracy of performed studies was also decreased due to limitations in the used LCA modeling tool. As an example, although the used tool has the ability to assess very detailed information regarding the composition of waste entering an incineration or landfill technology, the link between this composition and composition of different outputs from the same technologies is currently difficult to assess for the user.

4 RESULTS AND DISCUSSION

The work explores the environmental impacts from the Swedish system for management of solid household waste from several different perspectives; starting with a detailed description of the waste flows from a specific residential area, the effects of introduction of new source-separation possibilities of some waste fractions and information campaigns regarding waste recycling amongst households, continuing with system analyses using an LCA-approach to evaluate the impact of changes made in the waste management chain and behavioural changes. Results from studies realized within this work are presented and discussed below.

4.1 General knowledge of waste flows and recycling behavior

A waste flow analysis presenting the current situation within the case-study area could be determined through repeated waste composition analyses where the ratio of source-separation and miss-sorting was determined. The size of rejected material is determined by the ratio of miss-sorted material and does not take material losses in the recycling process into consideration (Figure 9).

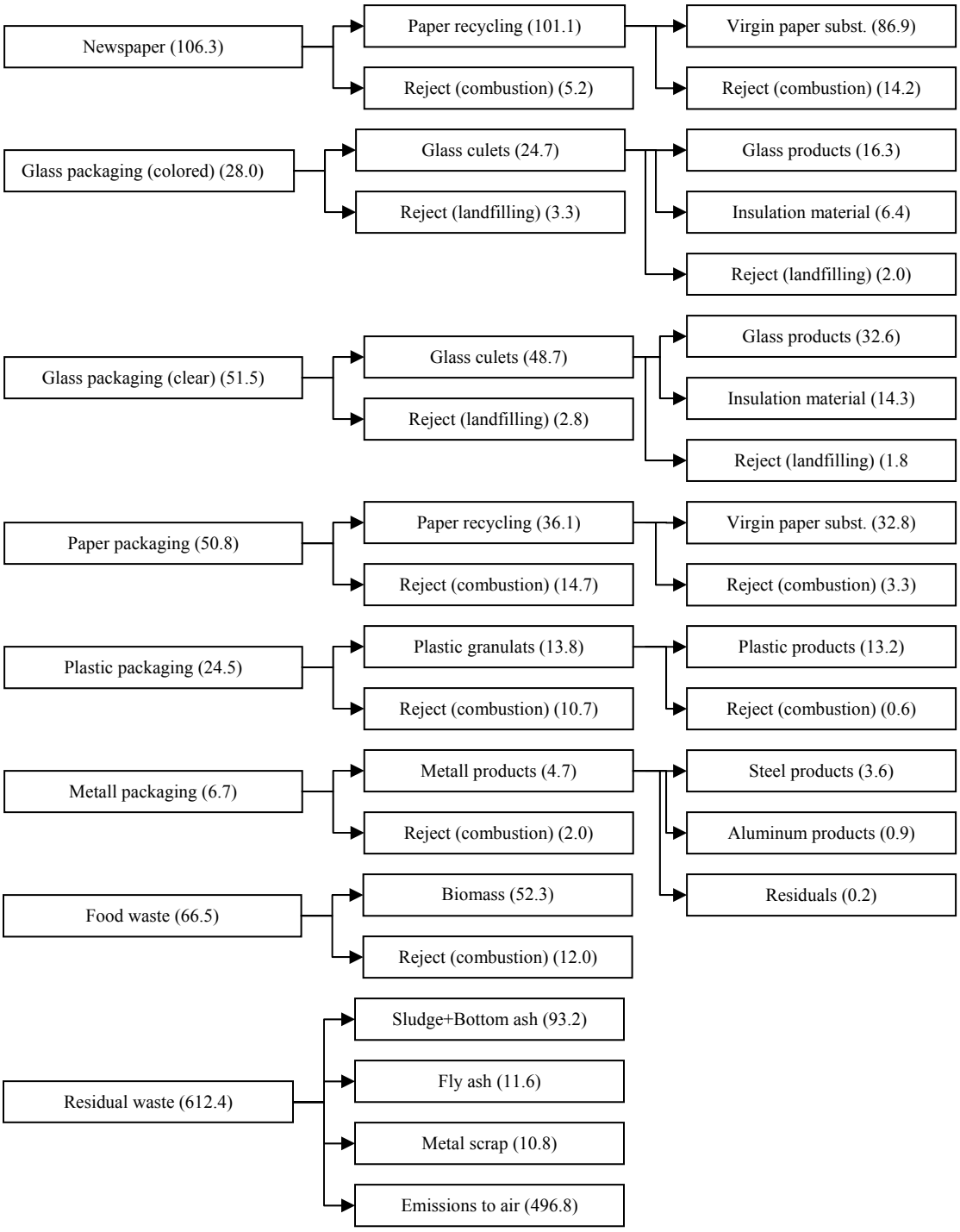


Figure 9. Waste flow at the study site (ton year^{-1}). Describes the current situation.

As pointed out by Dahlén (2008), different methods for solid waste composition analysis are used in different countries and many times also within different countries. If characterization and quantification of waste flows were performed and interpreted more consistently, comparisons of different collection systems and cause/effect discussions would be facilitated (Dahlén, 2008). A possible reason for these differences is that some

methods might be more applicable than others depending on the wanted information. In this work, the waste composition analyses where used to render information to be used in a waste flow analysis, for evaluation of recycling behavior and as input-data in later life-cycle assessments. Thus, detailed information was needed.

The weight based method for waste composition analyses used in this study – with division of waste into 8 main and 17 sub-categories – provides a comprehensive structure for waste composition analyses, and good understanding of waste flows in the area, source-separation ratio and ratio of miss-sorted materials in relation to dry recyclables and food waste. However, it might not be optimal for determination of source-separation behavior and environmental or economical evaluations of source-separation of smaller and heterogeneous waste fractions such as hazardous waste and WEEE, where the abundance is very irregular and the quality rather than the quantity is of large interest. As previously stated, the waste composition analyze method needs to be adapted to the information wanted in the particular situation. As an example: if the waste composition analyze is performed as a part of an LCA for food waste management, the ratio of proteins, carbon hydrates and fat as well as the calorific value and the content of N, P and K in disposed food waste is of interest. If the analyze is performed as input to a waste minimization campaign, the amount of disposed eatable food is of larger interest. Thus, though comparableness would be increased with an increased standardization of waste composition analyses, the methods would need to be adjustable to suit the needs in the specific case.

In the analyses performed in this work, 5 weight-% of the disposed waste was described as “others”. It was assumed that 50% of these were combustable and 50% inert, but an addition of more categories, such as “other combustable” and “other non-combustable” would make the evaluations of different treatment options more robust. However, in some cases, input data in LCA-studies is based on as little as twelve different waste categories (IVL, 2002). An examination of weather this has any larger impact on the overall results was beyond the scope of this work.

The overall source-separation ratio of dry recyclables varied between 57-62% (SD = 6.2%) and the ratio of missorted material between 18-25% (SD = 6.0%) during the experiment. However, both the source-separation ratio and the ratio of missorted material varied largely between different waste fractions and correlation was seen between source-separation rates and the ratio of miss-sorted materials amongst source-separated dry recyclables; the higher the source-separation ratio the lower the ratio of missorted materials. The same trend was not seen in the case of food waste, where both the source-separation ratio and ratio of missorted materials is low. However, as much as between 26 and 74 weight-% of all miss-sorted material amongst source-separated dry recyclables and food waste was of the same material as producer-responsibility materials in respective fraction; i.e. non-packaging plastics, metal or paper or organic matter other than food waste. This material amounts 20.5 kg household^{-1} year^{-1} based on four performed waste composition analyses (SD = 0.40 kg household^{-1} year^{-1}). Thus, a large part of the detected miss-sorting can be a problem on an economical and juridical level, rather than on an environmental level, as non-packaging are in most cases recyclable to the same extent as packaging.

The source-separation ratio of food waste in the study area is lower compared to the levels used in several prior studies (Eriksson et al., 2005; Kärrman et al., 2005, both

assuming a source-separation ratio of 70%). Experiences from an earlier pilot project in Aarhus, Denmark, imply that the actual amount of source separated organic waste can be smaller than expected. In the pilot project, the source-separation ratio only reached 42 weight-% of what had been expected (Madsen et al., 2003). Achieving high source-separation rates of food waste thereby seems more problematic than what can be expected. It could be expected that source-separation ratio would increase with time as households becomes more use to this behavior. However, this is not supported by results from this work (paper II).

Weekly weighing of disposed waste (as monthly averages) during a period of 24 months show a slight decrease of waste generation during the summer months during the first year but in general; little seasonal variation was seen in the generation of both residual waste and source-separated fractions (Figure 10). There were no clear trends in the total amount of generated waste. In the case of food waste the results suggest a trend towards decreasing amounts of source-separated waste. However, the correlation is weak ($R^2 = 0.27$).

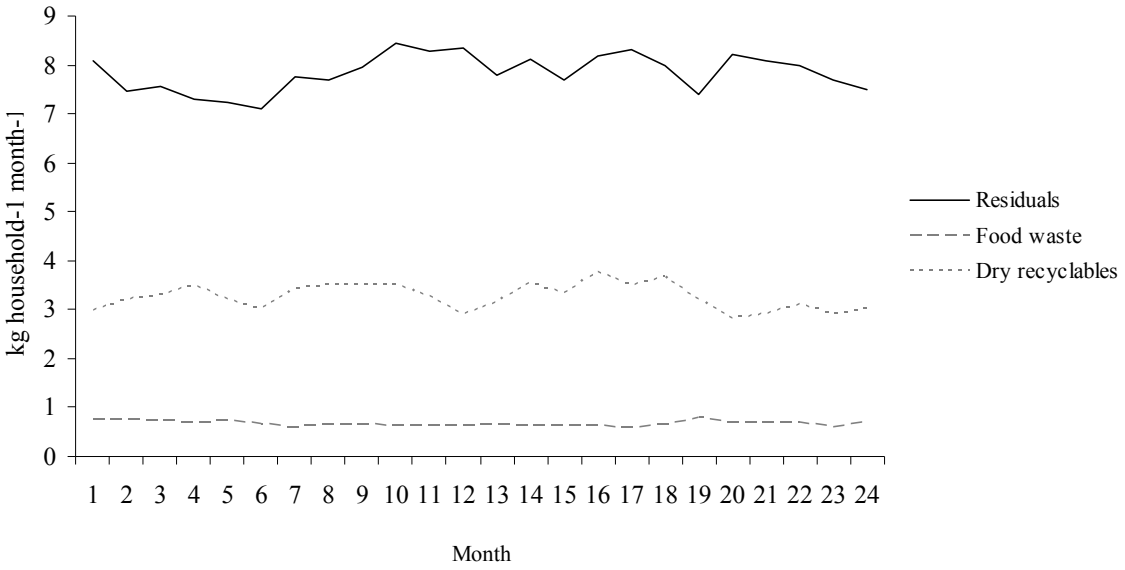


Figure 10. Results from 24 months of weighing of waste presented as monthly averages for dry recyclables, residual waste and food waste (kg household^{-1}, month^{-1}). Weightings were started January 2008.

The average source-separation ratio and ratio of miss-sorted materials varies largely between different waste fractions. The source-separation ratio was largest amongst glass packaging and newspapers while the miss-sorting was largest in the fractions plastic, metal and cardboard package (Table 6).

Table 6. Source-separation ratio and ratio of miss-sorted materials amongst dry recyclables and food waste, as averages with standard deviation (SD) and coefficient of variance (CV) based on performed waste composition analyses.

Fraction	Source-separation*	SD (%)	CV (%)	Miss-sorted materials*	SD (%)	CV (%)	Number of analyses
Food waste	23.8	4.2	17.6	4.1	3.2	79.3	4
Newspapers	71.2	7.0	9.8	4.9	1.2	23.8	3
Paper packaging	54.5	3.3	6.1	29.0	5.9	20.4	4
Plastic packaging	39.5	12.0	30.3	31.6	7.5	23.7	4
Glass colored	82.5	1.4	1.7	11.3	3.3	28.7	3
Glass clear	72.0	2.7	3.7	5.4	2.1	38.4	3
Metal packaging	38.0	8.1	21.2	30.2	11.8	39.2	4

* Average, weight-%.

The source-separation ratio of newspaper and glass packaging was significantly higher than the source-separation ratio of plastic, paper and metal packaging ($p<0.01$, t-test, 2-tailed). Results from the present work are thereby consistent with previous results presented by Dahlén (2008) and this result was therefore expected. However, the present study also show that the ratio of miss-sorting was higher amongst the later fractions compared glass packaging and newspapers ($p<0.01$, t-test, 2-tailed).

A correlation was seen between fractions where source-separation ratio is low and where the ratio of miss-sorted material is high (Figure 11). The same tendency is not seen in the case of food waste, were both the average level of source-separation and miss-sorting is low.

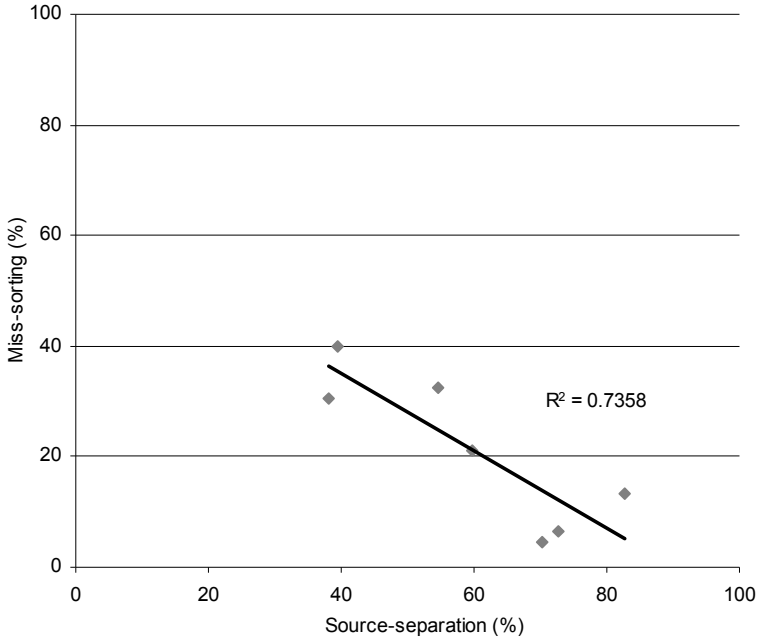


Figure 11. Source-separation ratio and miss-sorted material in fractions for dry recyclables. Averages based on four waste composition analyses.

As an effect of the long-term study design, both source-separation ratio and ratio of miss-sorting could be studied during a longer time period. Results show that the variations in

source-separation of both dry recyclables and food waste were minor. A slight increase in the source-separation ratio both amongst dry recyclables and food waste was seen in the third waste composition analyze (spring 2009) but levels had decreased again in December the same year. The only clearer trend was a steady decrease of the ratio of miss-sorted materials in source-separated food waste after the second analyze throughout the study period (Figure 9).

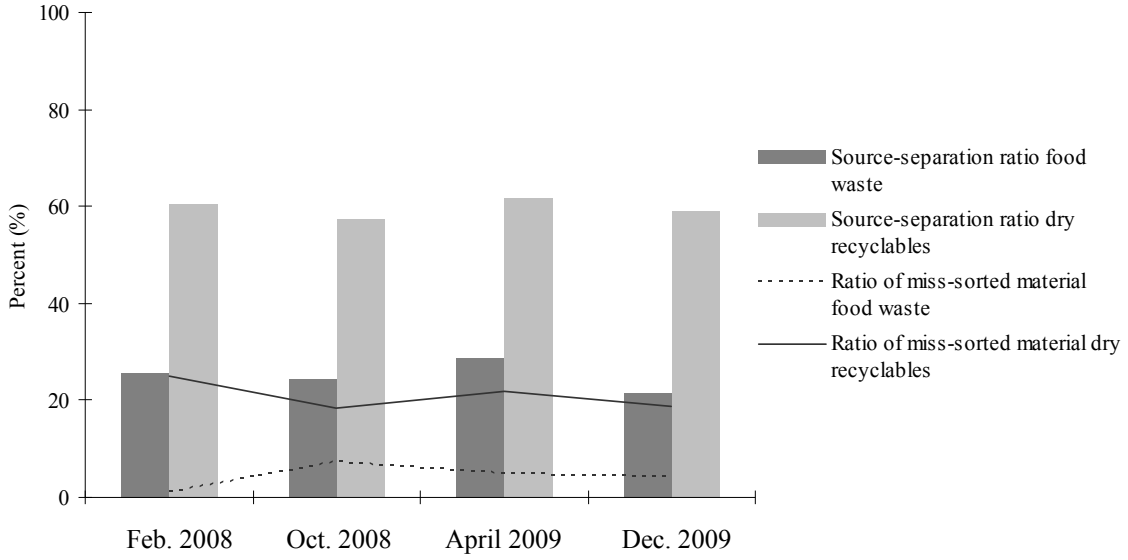


Figure 9. Source-separation ratio and ratio of miss-sorted material (weight-%) based on performed waste composition analyses.

The introduced system for property close source-separation of household waste in the study area has diverted 33% of disposed waste to material recycling or biological treatment (equaling 169 kg of dry recyclables and 35 kg of food waste per household and year). Based on available data, the diversion of recyclables from residual waste could be more than doubled, resulting in an overall recycling rate of more than 80% of generated waste, with the largest potential found in food waste, where the source-separation ratio could be three-folded (Figure 10).

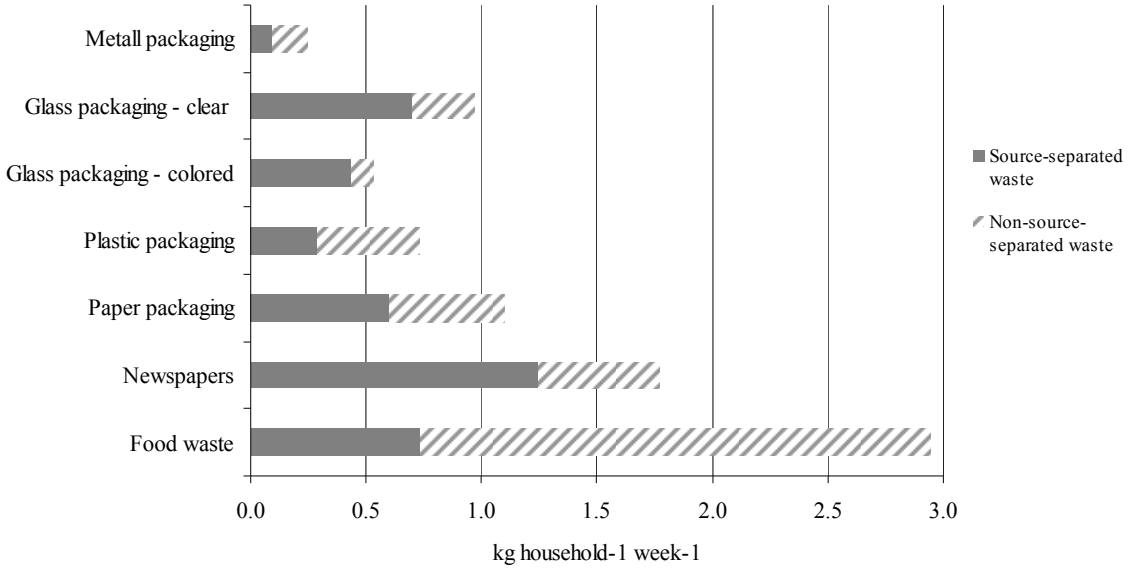


Figure 10. Amount of source-separated and non source-separated household waste as averages from four waste composition analyses.

Detected differences in source-separation ratios and ratios of miss-sorting between different dry recyclables could be explained by a longer tradition of source-separation of newspapers and glass packaging as the Producer Responsibility Ordinance on these fractions was introduced ten years before other dry recyclables in Sweden. It can also be an effect of confusion amongst households regarding the difference between packaging and newspaper recycling on the one hand and material recycling on the other as a correlation was seen also between ratios of miss-sorting and ratios of non-packaging sorted in bins for packaging (paper I). Previous studies have indicated that material-sorting can be widespread amongst Swedish households (SEPA, 2007) and that the division of responsibilities between producers and the municipality for different types of waste might be unknown or illogic to many residents (Ewert et al., 2009).

Findings from performed waste composition analyses provided information for construction of a waste-flow analysis in case of a perfect recycling behaviour amongst households at the study-site (Figure 11).

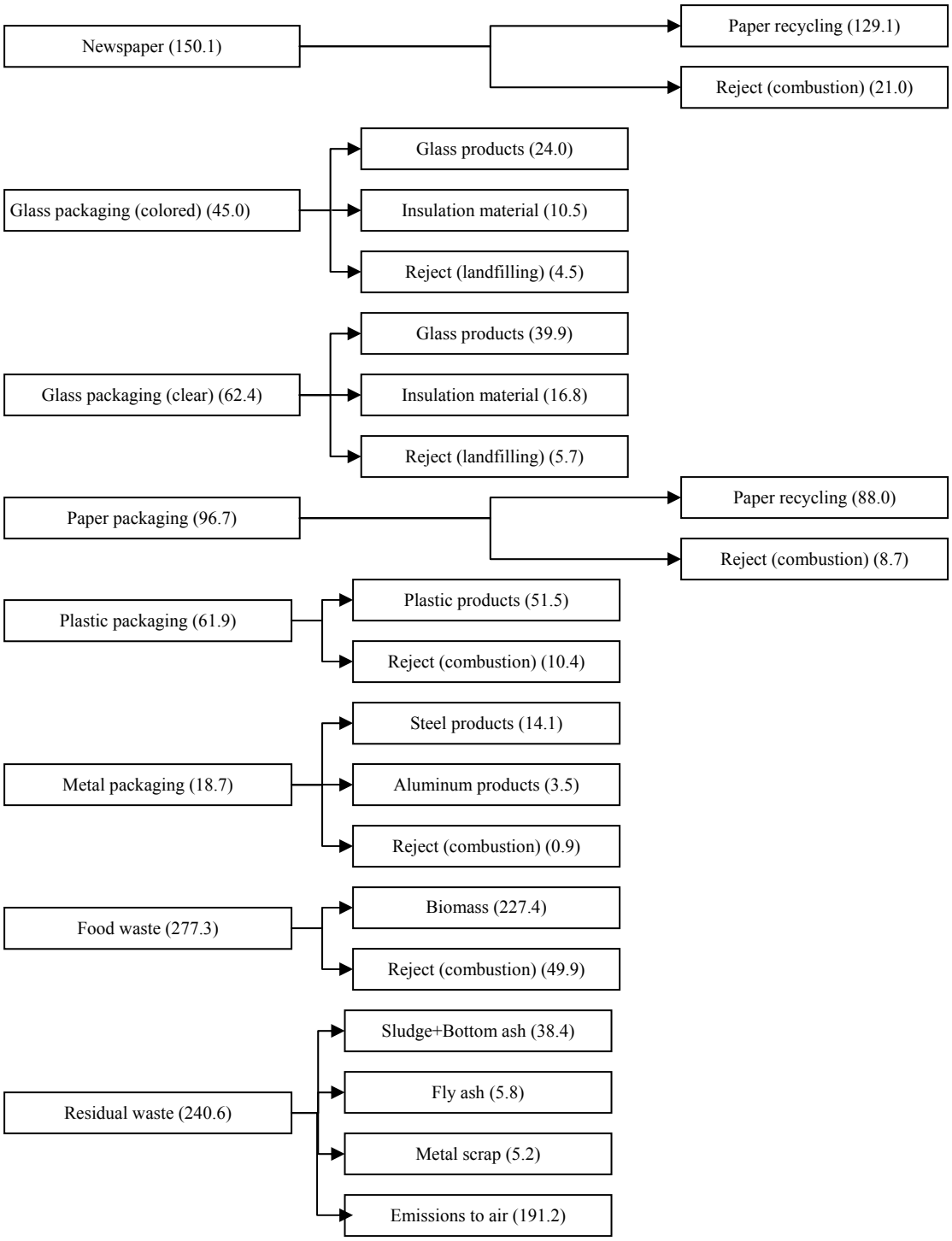


Figure 11. Waste flow in case of optimal recycling behaviour in the study area (ton year^{-1}).

Figure 11 shows that if a 100% source-separation as well as a 0% miss-sorting of dry recyclables and food waste is achieved, the total amount of residual waste could be decreased with 50% compared to the current situation, equivalent to a generation of non-sorted residual waste of 173 kg household^{-1} year^{-1}.

The method for waste composition analyzes used in this work did not consider the impact from moister and contaminations. In order to assess this, a separate test was performed on the influence of moister losses from food waste during the source-separation process. The test shows an average weight reduction of 19% (SD = 1.2%) in food waste collected in paper bags and 5% (SD = 1.3%) in food waste collected in plastic bags. Using this value and values earlier presented moister and dirt related correction values for metal, plastic and paper packaging and newspaper would affect earlier presented levels of source-separation (Figure 12).

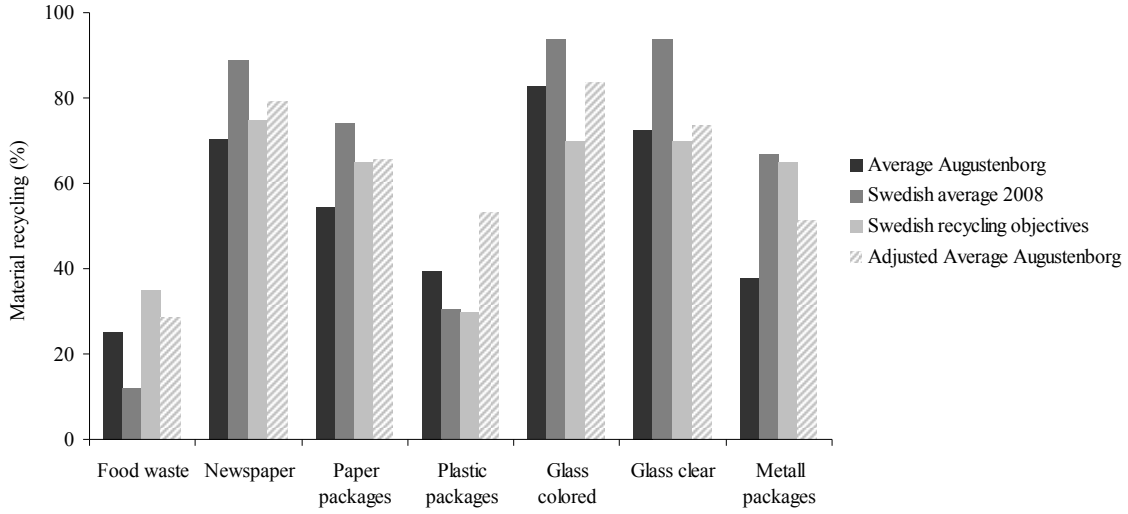


Figure 12. Source-separation ratios for different waste fractions assessed through waste composition analyses and adjusted with correction factors.

Using correction factors would increase the average recycling rate from 60 to 68% for dry recyclables and from 24 to 28% for food waste. As no assessment was made on the level of humidity and dirt-content in neither source-separated nor non-source-separated waste in the analyses performed in this study, it is not possible to indicate weather adjusted or non-adjusted levels are more correct in this particular case. Results from this work suggest that use of correction-factors can have a large impact on the results but further research would be needed to confirm the factors used in this comparison.

4. 2 Intervention techniques to increase recycling behavior

A vast number of previous research projects have discussed what and how various factors influence participation-rates of households in recycling of solid household waste. Such studies in many cases focus on finding explanations of differences in recycling behavior between groups of households with demographical differences such as age, gender, socio-economic background, dwelling types etc (Berglund, 2006; Dietz et al., 1998; Sterner and Bartelings, 1999) and/or personal and social norms and general environmental awareness (Hopper and Nielsen, 1991; Widegren, 1998; Sterner and Bartelings, 1999). Others have focused on investigating the effect of different types of interventions approaches on the recycling behavior of households, such as information and rising of general knowledge of environmental issues (Spaccarelli et al., 1990; Vicente

and Reis, 2008) or economical incentives, such as weight based taxes (Dahlén et al., 2007).

Demographic differences have been explored by various sociological researchers. Gardner and Stern (1996) state that the majority of these studies conclude that it has been difficult to reach any conclusions regarding the parameters deciding whether a person participates in waste recycling or not. However, amongst the most commonly tested parameters, age and sex are the ones that have been believed to explain variation between individuals to the highest degree. Scott (1999), Dietz et al. (1998) and Lindén (1994) all suggest that age or generation belongings can be an important factor and that older people tend to source-separate household waste to a larger extent than younger (Scott, 1999 and Dietz et al., 1998). According to for example Merchant (1992) women source-separate more than men. Previous studies have also shown that the type of housing can be an important factor (Merchant, 1992; Förpackningsinsamlingen, SIFO, 1999). However, housing can also be seen as a secondary factor, as housing often is connected to factors such as level of education, income and effort needed to participate in source separation schemes.

The level to which *norms and values* can influence the level of source-separation of household waste has gained large interest throughout the years. The general hypothesis in this field suggest that a person who has knowledge of the consequences to the environment, her self and others connected to non source-separating and feels a responsibility for such consequences, will be more propensity to participate in source-separation schemes. This hypothesis has been supported by researchers like Hopper and Nielsen (1991) and Widegren (1998). The hypothesis has also gained criticism in studies where only a weak correlations between "environmental friendly" norms and values (measured by the NEP – New Environmental Paradigm-scale) on the one hand, and level of source-separation on the other, has been detected (Guerin et al., 2001).

Previous studies have also shown that *economical incentives* can have a positive effect of source-separation, but only to a certain extent. Experiences from areas where a payment per kilo have been implemented show that after a while, habitants tend to get used to the extra payment for non-separated waste. Raising the fee even more might not be an option as this can lead to waste dumping when fees are seen as unacceptably high. The impact of economical incentives on recycling rates of solid household waste has previously been studied by Dahlén et al. (2008). Results from such schemes in Swedish municipalities show a significant decrease of total waste generation and non-sorted residual waste. However, an increase in generation of recyclables could not be seen and an increase of illegal disposal of household waste could not be ruled out (Dahlén et al., 2008). Results from Swedish municipalities also showed a larger ratio of impurities (i.e. miss-sorted materials) in source-sorted recyclables in municipalities with weight based taxes, compared to municipalities with volume based taxing systems. Thus, although monetary incentives can be efficient, they can also give non-wanted side effects.

Also *accessibility* has been lifted as an important factor influencing participation rates in recycling schemes. The impact of access and the level of effort needed to participate in source separation of household waste have also been addressed by Derksen and Gartrell (1993). The authors suggest that these factors are the overall most important factor for explaining different levels of source-separation in a Canadian example (Derksen and Gartrell, 1993). Guagno et al., (1995) suggest that values and norms must be seen in a

context where also the design of the recycling system and resulting access and convenience must be addressed. According to McDonald and Ball (1998), an uncomplicated source-separation system can be of greater importance than the general knowledge of the benefits of recycling. Studies have also shown that the best outcomes of systems with increased fees for non-source-separated recyclables have been seen in areas where fees have been combined with high accessibility of source-separation (Linderhof, 2001; Miranda and Aldy, 1998). It should be noted that the use of the term accessibility in cited literature refers to factors such as distance to the closest recycling point, i.e. the physical structure of the recycling system *outside* the household.

Use of *information* as an intervention technique suggests that once people understand why and how to change their behavior, they will do so (de Young, 1993). In relation to recycling of household waste, information can be either factual, with focus on the possibilities of recycling in the closeness of the informed household, or of a more persuasive nature with a focus on the environmental benefits from recycling. Information can also be delivered in different forms and through different types of communication strategies; through mass-media, in writing, orally/face-to-face etc. Vicente and Reis (2008) conclude that information and communication related to improved recycling behaviour can be of large importance. According to the writers, citizens who are better informed about recycling have a greater propensity to participate in recycling than those who are not so well informed. Information that clarifies which materials are recyclable and which container is appropriate for each material is important to assure households that they are doing the 'correct thing' (Vicente and Reis, 2008).

Separation of factors that might influence recycling behavior from each other can be complicated. The set-up of experiments where these issues are investigated are often complicated by the fact that compared groups of households tend to vary in a number of ways other than the factors tested for in the particular study. Also, many factors believed to be of importance to recycling rates (such as education, socio-economic level and dwelling type) can be interdependent and difficult to separate from each other. Evaluation of the impact of different factors is thereby difficult. As an example, Berger (1997) have shown that the distance to the closest public source-separation point had a close correlation to the socio-economic level of the household, and that this distance increased in areas with lower socio-economic levels (Berger, 1997).

4.2.1 Impact of oral information at the study-site

As previously mentioned, households at the study site were divided into three groups. Two of the groups received oral information regarding source-separation of food waste. One of these groups also received oral information regarding recycling of dry recyclables. Results from weighing of waste disposed were divided in these three groups (Figure 13).

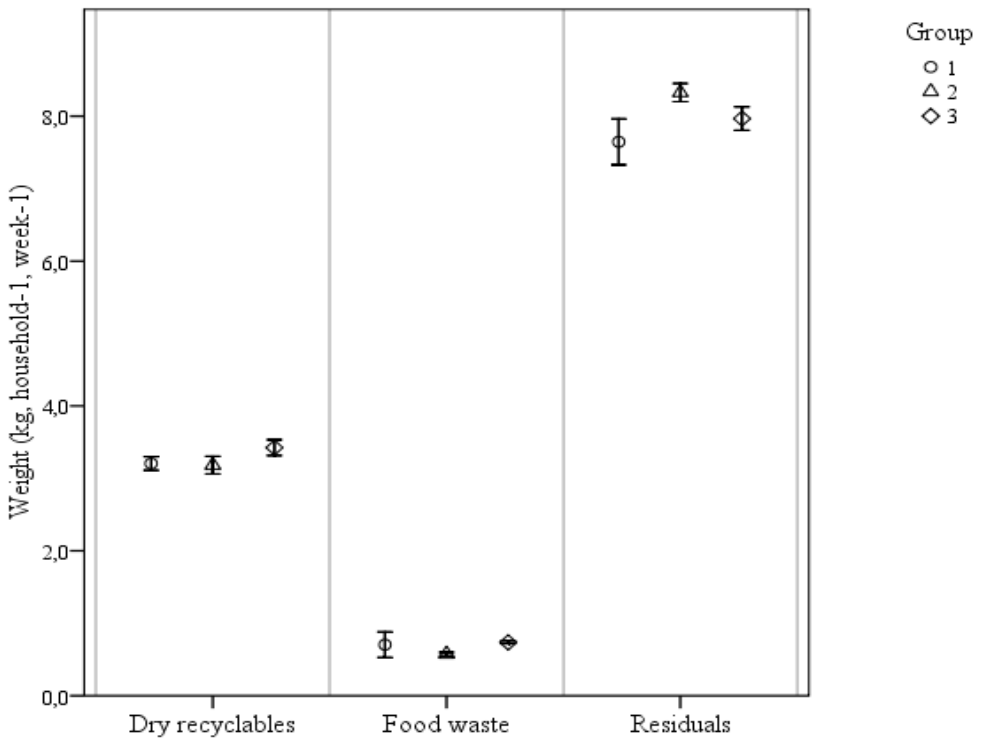


Figure 13. Specific waste generation, averages, of dry recyclables, residual waste and food waste from group 1, 2 and 3. Error-bars represent +/- 1 standard deviation.

The source-separation ratio and the ratio of miss-sorted material in analyzed household waste was determined based on four waste composition analyzes (Figure 14 and 15). Source-separation of food waste was introduced only in households belonging to group1 at the time for the first waste composition analysis at the study-site.

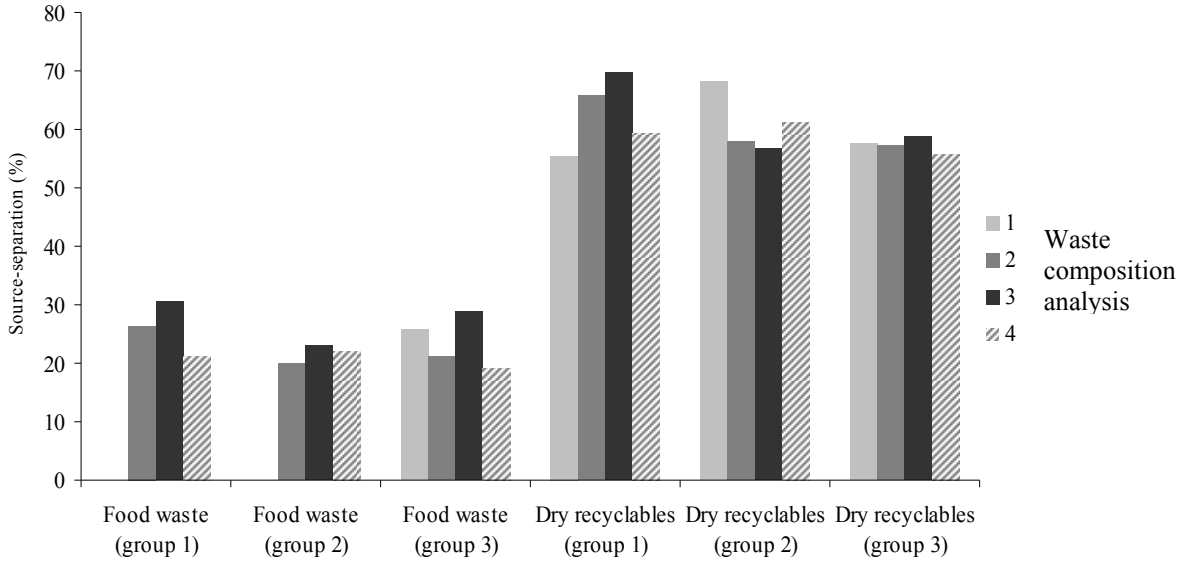


Figure 14. Source-separation ratio (weight-%) based on performed waste composition analyses (1-4).

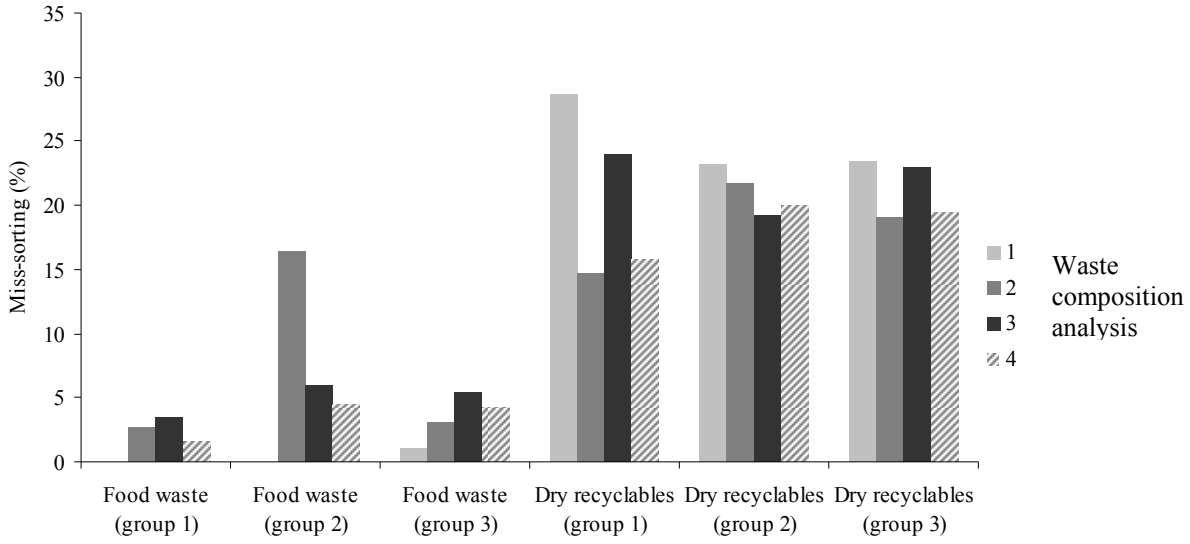


Figure 15. Ratio of miss-sorted material (weight-%) based on performed waste composition analyses (1-4).

Differences between generated residual waste, food waste and dry recyclables between different groups were tested using one-way ANOVA (post-hoc HSD Tukey). Results show that disposed residual waste in area 2 was significantly higher compared to group 1 and 3 ($p<0.05$ in both cases). The amount of both food waste and dry recyclables was significantly higher in group 3 compared to group 1 and 2 ($p<0.05$ in both cases).

A questionnaire was also used in the evaluation of the impact of the oral information campaign used in the area. Although the answering frequency from the questionnaire was low (12%), some interesting trends could be seen in the material. Answers from the questionnaire show that a large part of the households where source-separation of food waste was self-reported had received oral information regarding this fraction (59%). However, only 38% of these reported that the oral information hade provided them with any more profound information regarding the recycling scheme than what they had already gotten through the written information. 54 % of all responding households reported that they did not have sufficient room in the kitchen for source-separation of food waste. Almost 70% of households who reported that they did not participate in source-separation of food waste claimed that lack of space in kitchen was the overall negative motivation factor (paper II).

Presented results can be said to support previous findings by Vicente and Reais (2008), but as both source-separation and purity in food waste increased over time amongst households where no oral information had been delivered, the oral information could be regarded more as a catalyst, initially increasing recycling behaviour amongst households when introducing novelties in the waste management structure, rather than as a determining factor to weather households will participate in waste recycling or not. The decreased levels in the last analysis could be a result of the turn-over rate in the area and a need to repeat the oral information campaign on a yearly basis (paper II).

Also, in accordance with McDonald and Ball (1998), the design of the recycling system – starting already inside the actual household – is of great importance and this work indicates that information regarding environmental benefits of recycling cannot compensate for a

recycling system that is experienced as non-user friendly. More attention should be drawn to encourage waste separation and recycling behaviour already inside the household (paper II).

4. 3 Introduction of new source-separation systems

As a part of this work, new systems for source-separation of food waste as well as hazardous waste and WEEE were evaluated. The effects of the prior have been presented above as this was monitored also from the perspective of prompting activities. In the case of the latter, the effects were evaluated merely based on demonstrated source-separation behaviour.

4.3.1 Property close source-separation of hazardous waste and WEEE

A clear trend in Swedish waste management during later years is an increased interest for property-close source-separation of hazardous waste and WEEE (Swedish Waste Management Association, 2009). The main motivation for this is a reduction of hazardous waste in combusted residual waste and source-separated recyclables on the one hand and increased possibilities to recuperate valuable metals from WEEE on the other hand. Introduction of possibilities for property close source-separation of WEEE and hazardous waste in the study area in focus in this work was evaluated through repeated waste composition analyses and weightings of source-separated and illegally disposed hazardous waste and WEEE at the study site.

Results from performed analyses imply a decrease of both hazardous waste and WEEE in residual waste and source-separated recyclables. In both cases, an average reduction of more than 50% was reached. The difference was tested (univariate ANOVA, post-hoc Tukey, $p<0.05$) and was not statistically significant, as there was a large variance between samples. Results from the study showed an increase in the total amount of disposed hazardous waste and WEEE after the introduction of property close source-separation possibilities. This indicates that property-close disposal of these fractions was preferred over the previously more long-distance disposal possibility at municipal recycling centers. The source-separation ratio of hazardous waste and WEEE during the study period was determined to 70 and 76% respectively (Figure 16).

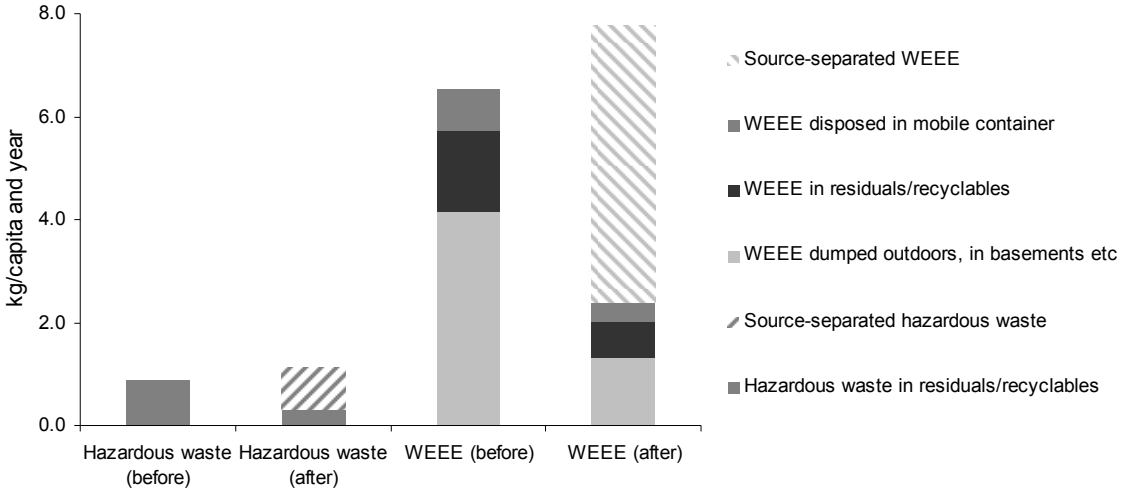


Figure 16. Total amount and distribution of disposed WEEE and hazardous waste at the study site before and after the introduction of possibilities for property close source-separation of WEEE and hazardous waste in recycling buildings. Based on data from 6 months before and 18 months after the introduction of the system and three waste composition analyses.

It was seen that more than 65% of all non-source-sorted WEEE was found in bins for dry recyclables, especially in containers designated for plastic and metal packaging. As a development of the used method for waste composition analyses, the weight of each non source-separated WEEE and hazardous waste item was collected. Use of such data in combination with databases with average composition of a variety of electronic devices can give relevant estimations of the economic and environmental impact connected to WEEE-recycling.

4.3.2 Introduction of new recycling systems in general

Based on results from this work, some suggestions on valuable aspects to consider in the introduction of new source-separation fractions could be stated.

Physical structures – Households must have the possibility to source-separate their waste already in the household in a rational and convenient manner. Also the placement of vessels for waste disposal in recycling buildings or similar is of importance to the quality of the source-separation.

Comprehensiveness – Waste recycling will probably always be an un-prioritized matter amongst modern households. Therefore, it is important that introduced systems are comprehensive and that the risk for confusion is low. If the information is unclear or non-logic, the risk of low quality and low ratio of source-separated materials could increase.

Information – Due to cultural differences related to for example food preparation and thereby production of household waste, it is relevant to consider the need for information in different languages and use of symbols and pictures for different waste fractions. Use of oral information can be helpful in order to improve recycling results and can also be useful as a means of distributing materials needed for source-separation activities. More professional terms, such as “hazardous waste” might not always bee well-understood

amongst households. A more easily phrasing or explanations might be needed. The use of domestic health care might increase the need for information regarding source-separation possibilities of different waste fractions also amongst health care staff.

Benefit – Closely interlinked to the concept of information is the concept of benefit. If households are required to put extra effort into something, their right to know why must be respected. Thus, it could be useful to present the motives for a specific source-separation scheme. Life cycle assessments could be used as a method to provide information regarding environmental benefits connected to the source-separation activities preformed by the households.

4. 4 Evaluation of recycling behavior and waste treatment alternatives using life-cycle assessment methodology

It was seen that the current source-separation behavior of dry recyclables and food waste at the study-site result in avoidance of 180% of the negative environmental impacts associated with a scenario without any material recycling, where all generated waste is incinerated. Thus, the current situation has changed a net contribution to net avoidance of negative environmental impacts. Optimizing source-separation behaviour would result in a further reduction of negative impacts with more than three times compared to the current situation (Figure 17).

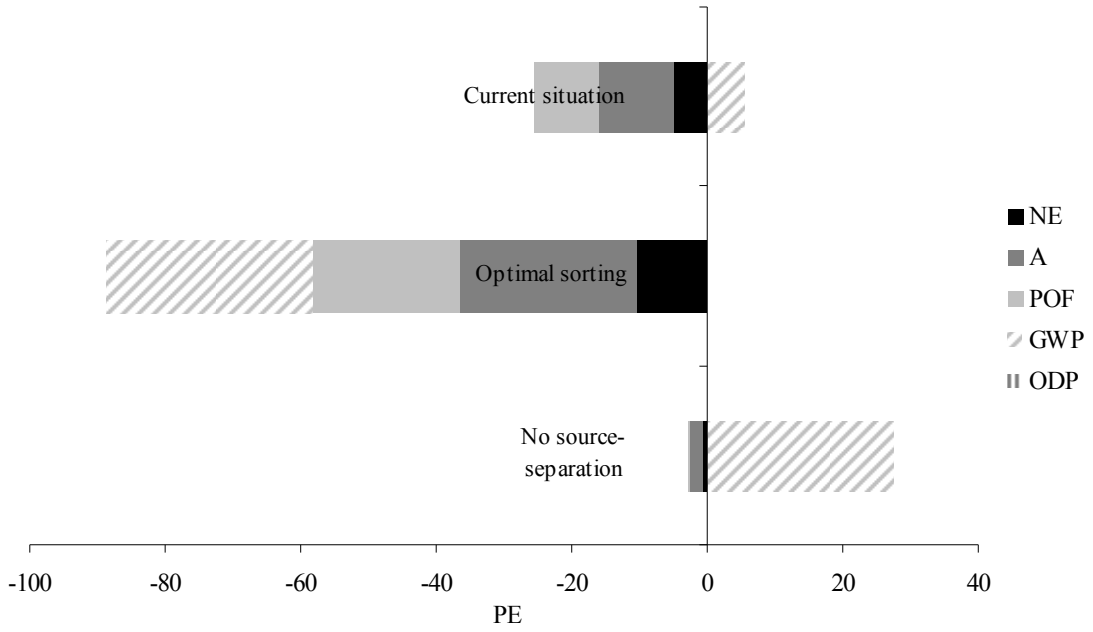


Figure 17. Environmental impacts related to current source-sorting, optimal sorting and non-sorting from all household waste generated at the study-site during one year, presented as yearly person equivalents (PE) for assessed environmental impact categories.

Previous studies have shown that assumptions regarding origin of electricity substituted by energy recovered from incineration of waste have a major impact on results from waste-related LCA studies (Sahlin et al., 2004; Tyskeng and Finnveden, 2007; Ekvall et

al., 2007; Mathiesen et al., 2009). The impact of changed assumptions regarding substituted energy (power and heat) from incineration of non-source-separated residual waste and rejects from recycling plants were assessed through several sensitivity analyses. Results from these show that the overall environmental impacts from compared scenarios largely depend on assumptions regarding the environmental profile of energy substituted from waste incineration. A substitution of coal based electricity (Dansih coal power, 2001) and fossil intensive district heating (Halldorssonar, 2001) from waste incineration will result in a large avoidance of waste treatment also in a non-separation alternative (paper IV). The large impact related to the environmental profile of used and substituted energy is a clear indication of the need of accurate and constantly updated data on these parameters in order to decrease the overall uncertainties in this and similar studies.

There are also large uncertainties regarding the actual amount of recovered metals from bottom ashes after incineration. In the base scenario it was assumed that recovered ferrous metal can substitute virgin material to 50%. Assuming that 25 or 75% of non source-separated metals can be used for substitution of virgin material decreases respectively increases the total avoidance of negative environmental impacts with 9% in both two cases. In the non-sorting scenario, the impact reaches almost 35%. Consequently, increased recovery of metal from bottom ash could substantially improve the environmental profile of both the current a non source-separation scenario substantially. The example also shows that an improved knowledge of actual recovery and substitution rates would decrease the uncertainties in the study.

The currently most frequently recycled waste fractions at the study-site (newspaper and glass) together with food waste give the lowest environmental benefits per ton recycled material. The largest potential environmental gains from an increased source-separation of household waste generated at the study-site are seen in the fractions plastic and metal packaging, as current recycling rates of these fractions are low and the potential per ton savings associated with material recycling of both materials compared to an incineration alternative are high (Figure 18).

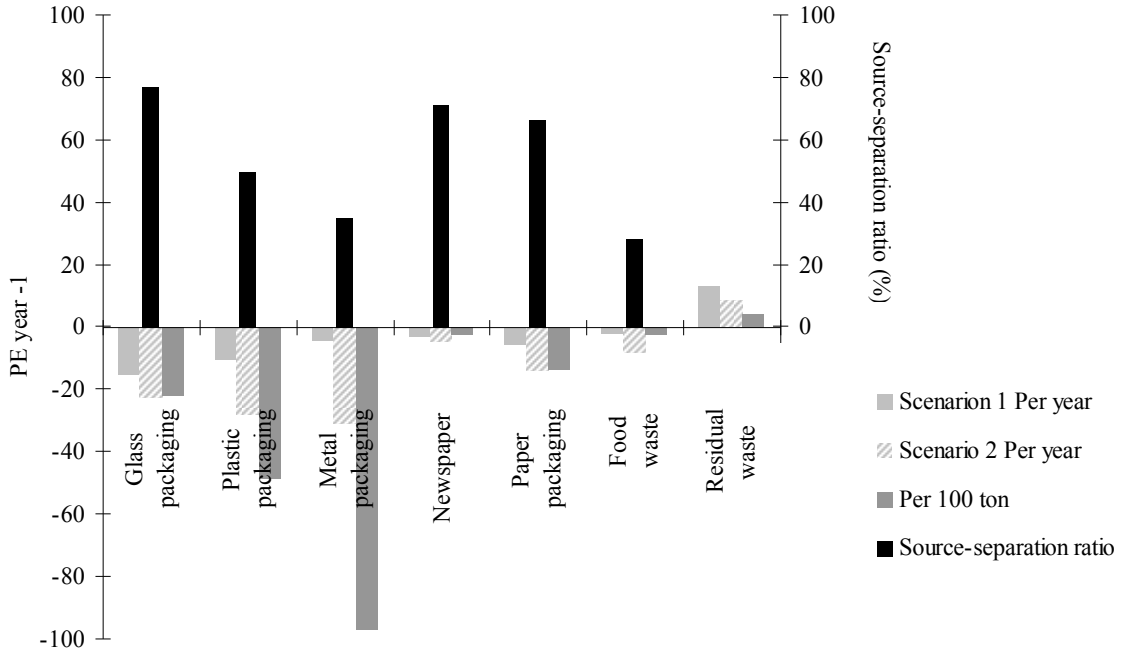


Figure 18. Environmental impact related to different recyclables presented as PE, scenario 1, scenario 2 and per 100 ton recycled material as well as current source-separation ratio of recyclables presented in percentages.

Results show large differences between the potential environmental benefits associated with optimal source-separation of different waste fractions in relation to the current sorting. Achieving a 100% source-separation of metal and plastic packaging would contribute to almost 60% of the total improvement potential, while reaching a 100% source-separation of newspaper only would contribute with 2% of the total improvement potential. It is also seen that the contribution to avoidance of negative impacts related to different environmental impact categories varies between assessed recycling fractions. The largest avoidance of acidification is related to metal package recycling while the largest avoidance of photochemical ozone formation is related to recycling of plastic packaging. Achieving an optimal source-separation of newspaper would, according to results from this work, result in a net contribution to photochemical ozone formation, which is mainly an effect of the needed transportation of the large amount of newspaper generated at the study site. In the case of food waste, an optimal source-separation can result in a net contribution to nutrient enrichment due to releases of nitrous compounds from digestate on farmland. However, the level of such emissions can vary largely according to previously performed studies and performed sensitivity analyses have shown that use of different values found in the literature regarding leached of nitrate and ammonia evaporation can have a large impact on the assessment (paper V) (Figure 19).

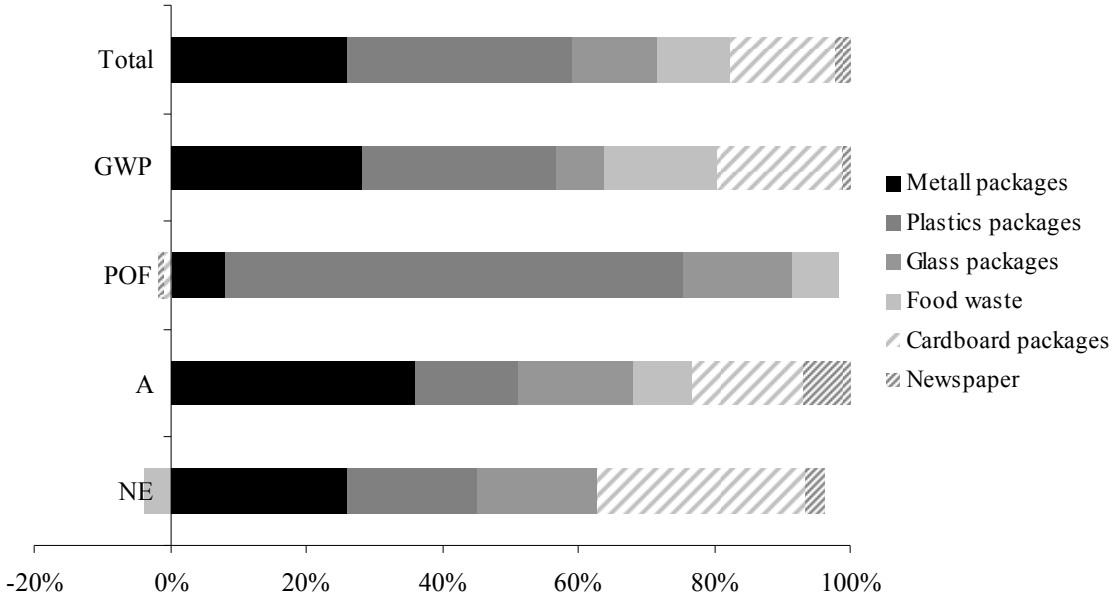


Figure 19. Relative contribution to environmental benefits if achieving an optimal source-separation of recyclables at the study site.

Performed sensitivity analyses show that results largely depend on made assumptions regarding the environmental profile of the energy used and substituted in different parts of the management chain. However, different waste fractions are differently sensitive to such changes, as demonstrated by the examples of plastic packaging and newspaper in this work. An increased recycling of plastic packaging would result in an increased avoidance of GWP even if fossil intensive energy is used in the recycling process and produced energy from combustion is substituting coal based energy. In the case of newspaper however, combustion is more beneficial if produced electricity substitutes coal power, as paper is regarded as a biofuel (paper IV). Earlier studies have concluded that recycling of paper and cardboard result in energy savings, but not always avoidance of negative environmental impacts (Finnveden and Tyskeng, 2007).

In the case of glass and metal packaging, material recycling is preferable before combustion as no energy is produce from an alternative combustion of these fractions. However, an increased efficiency of collection of metals from bottom ashes after incineration decreases the relative environmental benefits from source-separation as metals than can be recovered for later material recycling. For fractions were there is an alternative energy production potential if combusted, the environmental profile (in relation to GWP) of substituted energy can be of importance in deciding the hierarchy between different treatment alternatives. In the case of newspaper and paper packaging there is an alternative environmental benefit when these biogenetic fractions are incinerated with energy recovery. Incineration is more beneficial if produced energy substitutes fossil intensive energy production, if electricity used in recycling process is fossil intensive or if the substitution-ratio of virgin products decreases below 70%. Due to the fossil CO_2-emissions related to combustion of plastics, recycling of plastic packaging is more environmentally beneficial than incineration even if fossil intensive electricity is used in recycling processes or combustion of plastics would substitute fossil intensive energy production. Although the impact from energy substitution and production was

large and could have an impact on the hierarchy between different treatment alternatives with regard to GWP when looking at different waste fractions separately, combustion of waste without source-separation was seen to have a larger negative impact on all assessed impact categories compared to a scenario with optimal source-separation, even when energy from waste incineration substitutes coal based energy production in both scenarios (paper IV).

Life-cycle assessments were in this work also used to identify environmentally suitable treatment technologies for certain waste fractions. A study was made on treatment of food waste with incineration, compost and anaerobic digestion as possible treatment alternatives. Anaerobic digestion result in larger avoidance of global warming and formation of photochemical ozone than if food waste is composted or incinerated. Both anaerobic and aerobic treatments leads to a larger net contribution to nutrient enrichment and acidification compared to incineration, whereas incineration result in a slightly better energy balance compared to incineration (paper V). Use of biogas as car fuel is preferable over use of biogas for production of electric and thermal energy only, if produced electricity is used as substitution for Swedish average electricity. Substitution of chemical fertilizers with produced digistate replies to between 45 and 60% of the total environmental benefits from an anaerobic digestion alternative, depending on the use of produced biogas (e.g. car fuel or electricity production). Thus, maintaining high quality produced digistate it is vital for the overall environmental impact of anaerobic digestion (Figure 20).

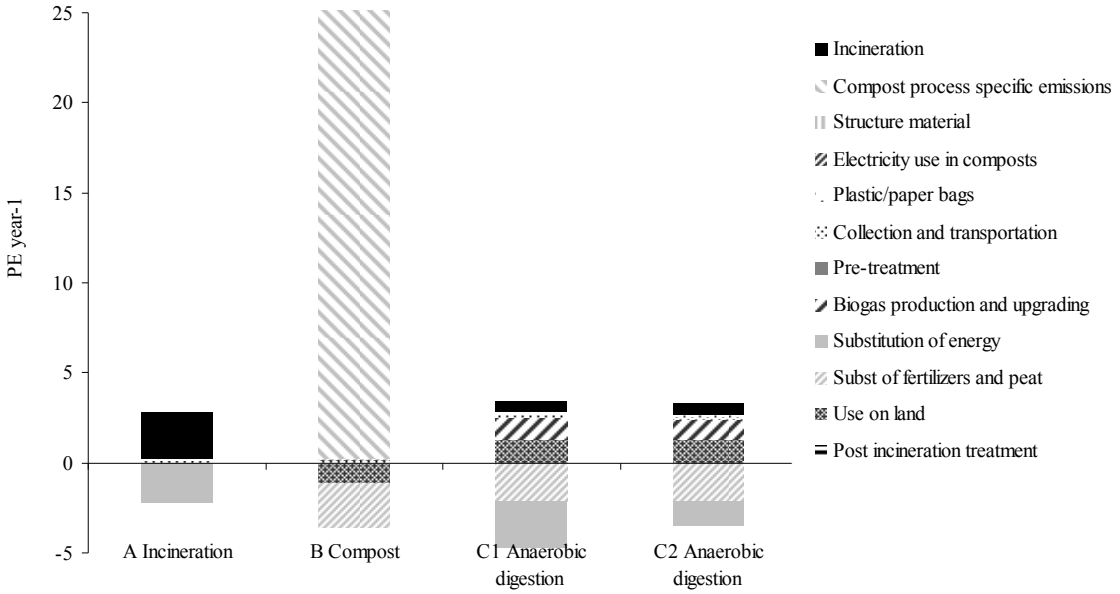
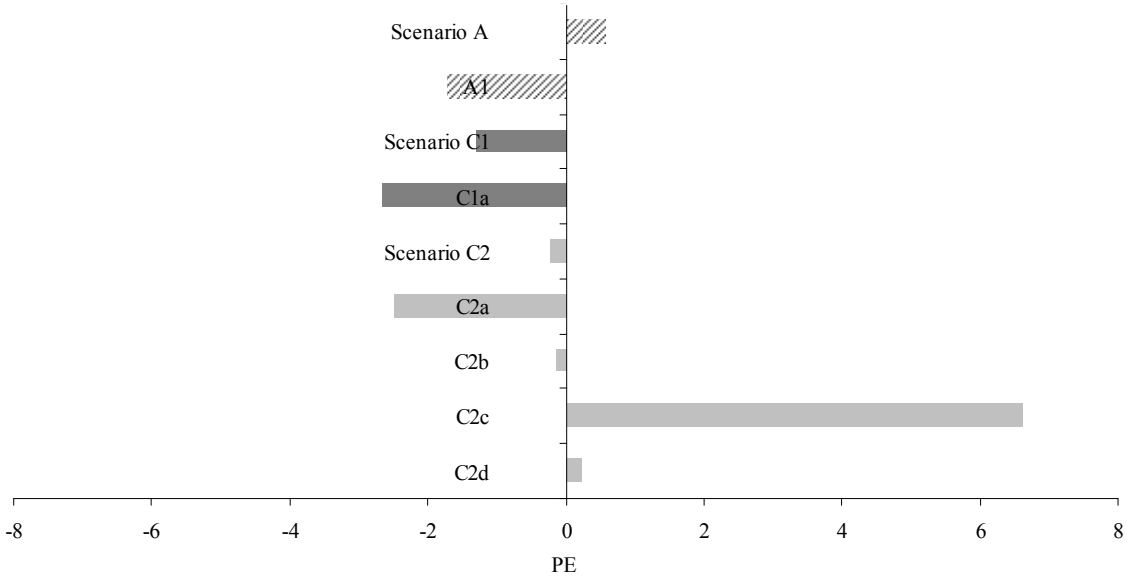


Figure 20. Environmental impacts from treatments of food waste through A Incineration, B Compost, C1 Anaerobic digestion with use of biogas for production of car fuel and heat production and C2 Anaerobic digestion with use of biogas for production of electricity and heat production, divided on different processes.

The majority of the negative environmental impact connected to the compost scenario is related to emissions from the compost process. As emissions to air were not monitored at the study-site, these values were based on literature values. Variations between previous studies are large and therefore the uncertainties are large. In order to assess the impact of

these uncertainties on the overall result from the comparison and to locate key factors with high impact on the overall environmental profile, a number of sensitivity analyses were performed. Data from three previously performed studies on emissions of N_2, NH_3, N_2O and CH_4 during composting of food waste were compared (Chung, 2007; Boldrin et al., 2008 and Sonesson et al., 1996). The comparison showed that depending on the reference used as input data in the assessment, the total environmental impact could vary with a factor 10 from the compost scenario in total. However, also in a scenario where data input reflected lowest levels of emissions with negative environmental impacts, the hierarchy between compared treatment alternatives did not change (paper V).

It was seen that the most relevant parameters for determination of environmental impacts from treatment of food waste with processes where energy is produced are related to the environmental profile of substituted energy. Using produced biogas as fuel is more beneficial than production of electricity if Swedish average electricity is substituted and increasing the production of fuel in relation to heat would lead to further avoidance of negative environmental impacts from anaerobic digestion. However, if fossil intensive power or heat generation is substituted (in this case exemplified by Danish coal power and district heating), combustion of food waste is more beneficial than anaerobic digestion. Also, using biogas for electricity production is more beneficial compared to production of fuel, if assuming that produced electricity from combusted biogas substituted Danish coal power. Performed sensitivity analyses also show that incineration is preferred over anaerobic treatment if digestate produced in an anaerobic treatment alternative not can be used to substitute chemical fertilizers. However, the ratio of which organic fertilizers can be assumed to substitute chemical fertilizers and the environmental profile of these fertilizers are also of large importance. The cleaner the production of chemical fertilizers, the lower the environmental gains from biological treatment options. Assuming a production based on best available technique (BAT) (Jansen and Kongshaug, 2003) would weaken the environmental profile of both composting and anaerobic digestion substantially. A sensitivity analysis was also performed in order to assess the effect of use of plastic bags for separate food waste collection – compared to the current use of paper bags. Results show that this has a large negative effect on the overall environmental performance from the anaerobic treatment alternative (paper V) (Figure 21).



Label	Scenario and scenario modifications	PE year-1
Scenario A	Incineration substitution of power and heat	0.6
A1	Substitution of Danish coal power	-1.7
Scenario C1	Anaerobic treatment substitution of car fuel and heat	-1.3
C1a	Increased substitution of biogas	-2.7
Scenario C2	Anaerobic treatment substitution of power and heat	-0.3
C2a	Substitution of Danish coal power	-2.5
C2b	Use of plastic bags for food waste collection	-0.1
C2c	No use of digestate	6.6
C2d	Substitution of BAT chemical fertilizers	0.2

Figure 21. Results from sensitivity analyses related to energy substitution and use of digestate as substitution of chemical fertilizers.

Also assumptions regarding emissions of N_2O, NH_3 and NO_3- from organic fertilizers (compost and digistate) on farmland will affect the results, especially in the case of anaerobic digestion as well as unwanted emissions of CH_4 from biogas production plants. These will to a large extent depend on techniques used for spreading of digestate on farmland and design of compost and biogas production facilities.

Thus, the environmental impact related to different waste treatment options and levels of source-separation will be affected by a variety of factors:

- Technological development with impact on the environmental profile of materials (e.g. virgin materials and fertilizers) and energy substituted in different parts of the waste treatment chain, as well as the efficiency in recycling and treatment processes.
- Assumptions regarding the fate of non-used virgin products, substituted by material recycling.
- Political decisions/development of standard with impact on the usability of materials produced in the waste treatment chain, for example digestate or compost.

- Biological processes with impact on emissions of non-wanted substances and the level to which a non-waste fertilizer can be substituted by a waste based fertilizer – processes that can vary from case to case and where the level of true uncertainty still can be high.

Many of these factors are related to processes only indirectly linked to the waste management chain and outside the sphere of control of decision-makers of waste management strategies. An example is emissions of N_2O and NH_3 that are influenced by the used technique for spreading of fertilizers on farmland. This poses a difficulty in the use of LCA as a decision support tool, as agents making choices between different waste treatment alternatives in most cases probably not will have the possibility to control neither all up- or down flow processes identified as vital for the overall environmental impact for different recycling and treatment alternatives. However, the LCA can provide useful information regarding the relative importance of these factors, information that can be used in dialogue with the agents with control of respective process. This indicates the importance of a holistic approach and extended collaboration between the different agents involved in the waste management chain in order to increase potential environmental benefits, avoid negative side-effects and achieve a sustainable management of household waste.

5 CONCLUSIONS

Knowledge of waste streams and recycling behaviour is essential for assessments of the environmental impacts from current waste management and potential improvements of the same. The case-study approach used in this work provided detailed data on waste streams from the specific study site with property-close source-separation of several different waste fractions. It was seen that 80% (bulky waste not included) of the waste generated at the study site could be source-separated for material recycling, biological treatment or special treatment (in the case of hazardous waste) with the current system for source-separation. Despite the high accessibility for waste source-separation, only 33% of generated waste was diverted from residual waste. The management of the large amount of non source-separated packaging and newspapers is thereby paid for twice; once by producers and once by the municipality.

Material recycling, in opposite to recycling based on the Producer Responsibility Ordinance on packaging and newspapers, was proven common in the residential area used as study site in this work. This shows that the very base of the solid household waste recycling system in Sweden to a large extent is unknown or not respected by households. At the same time

It was seen that extensive distribution of information regarding both practical recycling possibilities as well as environmental gains related to waste recycling not can be assumed to automatically lead to high rates of food waste source-separation. As the highest recycling rates were seen in fractions with the longest history of source-separation in Sweden (newspaper and glass packaging) it could be assumed that increase of recycling rates is a matter of 'getting used to the system' and that recycling rates will increase automatically with time. However, in the extensive system for property-close source-separation of household waste used at the study site, the households are key actors. It should be considered that their participation starts already *inside* the household. Results from the work suggest that in order to ensure their participation, greater attention should be drawn to the physical structure for waste disposal already in the household, before disposal in recycling buildings or similar. Thus, there is a need to expand the current use of the concept of accessibility in relation to household waste recycling to address also the possibility of waste separation also inside the household. Using a multi stakeholder approach when introducing recycling systems, where facility owners and managers as well as collection entrepreneurs and treatment facilities all participate can be an important strategy in the creation of waste collection systems that are beneficial both from an environmental perspective as well as for the different stakeholders involved in the chain.

The model for solid household waste management currently put in practice in Sweden, based the Producer Responsibility Ordinance and subsequent source-separation of packaging and newspapers can result in large environmental benefits compared to incineration without prior sorting. The recycling behaviour of households will to a large extent determine the magnitude of the environmental gains provided from the system and the potential increase of environmental benefits from the system is large. The waste fractions connected to the largest environmental benefits when material recycled are also the fractions where the lowest ratio of source-separation currently is seen. A similar incoherence is seen in Swedish national recycling objectives, i.e. the toughest targets for household waste recycling are currently not related to waste fractions where the per ton

environmental benefits are highest. Thus, the basis for current objectives could be questioned.

It should be recognized that the determination of environmental impacts from material recycling, incineration and biological treatment of solid household waste to a high degree are connected to made assumptions regarding the environmental profile of energy and materials substituted by respective waste management system. This makes results from the assessments highly sensitive to choices related to the environmental profile of consumed and substituted energy. In general, many of the factors with the most vital influence on the overall environmental impact from the household waste treatment chain are found outside of the sphere of control of decision-makers of management strategies for solid household waste in Sweden today, i.e. local politicians and municipal waste management departments. The same is seen in the case of behavioural changes related to household waste separation, where results from this work indicate that facility owners play a key role in increasing of accessibility and construction of needed infrastructure for efficient waste separation. This indicates the importance of a holistic approach and extended collaboration between the many different agents involved in the waste management chain, in order to increase potential environmental benefits and achieve a more sustainable management of municipal solid household waste.

6 FURTER RESEARCH

The method for waste composition analyses used in this work proved to be useful in determination of source-separation ratio and miss-sorting. However, as moister-losses from source-separated food waste as well as surplus moister-content amongst dry recyclables not were addressed, the accuracy in determinations of source-separation ratio as well as ratio of miss-sorting was decreased. Use of thoroughly assessed correction factors to adjust for moister gains and losses would mean a refining of the current analyze-method and an important area for further research. Depending of the purpose of the analysis, different types of adjustments of the method used in this work would be relevant:

- The most relevant factors to include in order to increase the usefulness in relation to performance of life-cycle assessments would be
  - the ratio of vegetal and animal food waste
  - the ratio of aluminum and ferrous metals in source-separated and non source separated fractions
  - the ratio of non-recyclable laminate-plastics in source-separated and non source-separated fractions

- In order to give a better understanding of household behaviour in relation to source-separation of food waste, the following parameters would be relevant
  - ratio of eatable food in source-separated and non source separated fractions
  - ratio of packed food waste in source-separated and non source separated fractions

- In order to give better information regarding economic and environmental impacts related to property close source-separation of WEEE and hazardous waste, the type and weight of each source-separated as well as non source-separated item would need to be assessed.

As results from this work indicate that an infrastructure in which the whole chain of waste source-separation is made convenient and accessible for households could be needed in order to reach high acceptance and participation rate for solid waste recycling, studies on how this can be accomplished are an area for further research.

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