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Preface

This thesis encompasses research done during a PhD project carried out at the Department of Environmental Engineering, Technical University of Denmark (DTU), from June 1, 2009 to August 31, 2014. Professor Thomas H. Christensen was the main supervisor; Associate Professor Charlotte Scheutz was co-supervisor. The thesis was funded by the DTU, the 3R research school, and the Ministry of Science, Innovation and Higher Education.

The thesis is organised in two parts: the first part puts into context the findings of the PhD in an introductive review; the second part consists of the papers listed below. These will be referred to in the text by their paper number, written with Roman numerals **I-III**.

I Bigum M, Brogaard L, Christensen TH. 2012. Metal recovery from high-grade WEEE: A life cycle assessment. Journal of Hazardous Materials 207-208 (2012) 8-14.

II Bigum M, Petersen C, Christensen TH, Scheutz C. 2013. WEEE and portable batteries in residual household waste: Quantification and characterization of misplaced waste. Waste Management 33 (2013) 2372-2380.

III Bigum M, Christensen TH, Scheutz C. 2014. Environmental impacts and resource losses when misplaced special waste (WEEE, batteries, ink cartridges and cables) from households is incinerated with municipal solid waste. Resources, Conservation and Recycling (submitted).

In this online version of the thesis, the papers are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from DTU Environment, Technical University of Denmark, Miljøvej, Building 113, 2800 Kgs. Lyngby, Denmark, reception@env.dtu.dk.

In addition, the following book chapter has been produced during the PhD study:

Bigum, M, Christensen, TH. 2011. Waste Electrical and Electronic Equipment. In Christensen, TH. (Eds.). Solid Waste Technology & Management, Chapter 11.2. John Wiley & Sons, Chichester (ISBN: 978-1-405-17517-3).

Bigum, M, Brogaard, L. 2010. LCA modelling of metal recovery from 1 tonne high grade WEEE. Proceedings. Crete 2010. 2nd International Conference on Hazardous and Industrial Waste Management. 5-8 October 2010, Chania, Crete.

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And to my supervisors, Professor Thomas H. Christensen and Associate Professor Charlotte Scheutz, thank you for your contributions to this work, which made it better and ensured the broader foundation.

Finally, I would like to thank the people in my life who were also there in the difficult moments and made them easier. Thank you for your never-ending support.

Marianne

"If a tree falls in the forest, and there's no one around to hear, does it make a sound?"

Summary

There has been an increased focus on special waste types (WEEE, batteries, ink cartridges and cables) in Denmark and abroad, as many of these fractions constitute a special threat to the environment, due to their content of hazardous compounds and valuable resources. Waste Electrical and Electronic Equipment (WEEE) and batteries are some of the special waste types receiving significant focus as hazardous and valuable substances in WEEE and batteries are plentiful. WEEE and batteries, which are not sorted out for recycling and recovery, do not only imply a loss of materials and metals but could also lead to pollution of other waste streams. In addition to this, there are significant environmental benefits to be obtained when recycling special wastes.

Many of the raw materials found in special waste are in an immediate supply risk for the development of emerging green technologies. The inherent resources in waste have become an obvious focus as a source of these critical raw materials, and the municipal solid waste is considered to be one of the largest potential sources for the recovery and recycling of scarce elements.

Special waste streams should, therefore, be collected and recycled. In particular, precious and scarce metals should be recovered due to environmental as well as sustainability issues.

In Denmark, there are still waste flows that are unaccounted for. One of these flows is the special waste that is being misplaced with residual household waste. Bigum et al. (II) investigated this by conducting a sorting analysis of the Danish residual household waste. The analysis showed that especially small household appliances, lamps, toys, leisure and sports equipment, and portable batteries were frequently misplaced with residual household waste. Misplaced special waste will, in Denmark, be incinerated. This leads to pollution of the surrounding environment with heavy and toxic metals, as well as being a significant source for abiotic resource depletion (Bigum et al., III).

Improvements with respect to the treatment of special waste are necessary. Traditional pre-treatment facilities seem to focus primarily on the traditional metals such as iron (Fe), aluminium (Al), and copper (Cu), which can be recovered in bulk amounts. Recovery of the precious and scarce metals is to a lesser degree carried out, as these appear in much smaller amounts. Future recovery facilities should, however, aim at recovering these metals, even though they appear in smaller concentrations, as the recovery of these can

have larger environmental relevance exceeding that of the traditionally recovered metals (Bigum et al., I).

Life cycle assessments (LCAs) are used as decision-making tools for supporting waste management decisions. LCAs must therefore also be able to incorporate issues related to special waste streams and management. The ability for LCAs to incorporate these issues is crucial for the tool to be able support decisions and to further justify the use of waste-LCAs when decisions are made.

One of these issues is related to special waste being a very heterogeneous waste type. The variation in composition is significant and data availability is scarce, which can make it difficult to include special waste in waste-LCAs. This also means that the environmental aspects connected with the special waste types can be difficult to fully assess, and that the consequences of these may risk being overlooked or underestimated.

The field of environmental assessment of special waste is relatively new, and many issues need to be resolved. One of these issues is the evaluation of resource depletion and scarcity. This area is in need of a much broader consensus and further scientific development in order to ensure that LCA is applicable and accepted as a decision-making tool.

This thesis shows the importance of including a detailed composition of the special waste types, as well as the importance of incorporating the resource depletion of unrecovered elements in waste-LCAs (Bigum et al., III). The thesis also shows that the recycling of metals is of significant environmental importance (Bigum et al., I) and quantifies the amount of special waste types being misplaced with residual household waste (Bigum et al., II). The thesis also concludes that there are still many issues that need to be resolved and suggested which areas need further research in order to improve the field of environmental assessments of special waste types.

Sammenfatning

Der har været et øget fokus på de specielle affaldstyper (WEEE, batterier, printerpatroner og kabler) i Danmark og udlandet, da mange af disse fraktioner udgør en særlig risiko for miljøet grundet deres indhold af farlige stoffer og værdifulde ressourcer. WEEE (også kendt som elektronikaffald) og batterier er nogen af de specielle affaldstyper, som der er særlig fokus på, da der er mange farlige og værdifulde elementer i WEEE og batterier. WEEE og batterier, der ikke udsorteres til genanvendelse, betyder ikke blot et tab af materialer og metaller, men kan også medføre forureningen af andre affaldsstrømme. Derudover kan der også opnås store miljømæssige fordele ved at genanvende de specielle affaldstyper.

Mange af de råmaterialer, som findes i de specielle affaldstyper, er i umiddelbart forsyningsrisiko hvilket kan hindre udvikling af såkaldt grønne teknologier. Affald som en ressource er derfor blevet en oplagt kilde for at sikre disse kritiske råmaterialer, og husholdningsaffald betragtes som en af de største potentielle kilder for genindvinding og genanvendelse af knappe ressourcer.

De specielle affaldstyper bør derfor blive indsamlet og genanvendt. I særdeleshed bør ædel- og kritiske metaller genanvendes af miljømæssige såvel som bæredygtige grunde.

I Danmark er der stadig affaldsstrømme, som der ikke er redegjort for. En af disse strømme er specielle affaldstyper, der smides ud med restaffaldet fra husholdninger. Bigum et al. (II) undersøgte dette ved en sorteringsanalyse af dansk restaffald fra husholdninger. Undersøgelsen viste at særligt småt husholdningsudstyr, sparepærer, legetøj, fritids- og sportsudstyr samt bærbare batterier ofte fejlplaceres og smides ud med husholdningsaffaldet. Specielle affaldstyper, der fejlplaceres med husholdningsaffaldet, vil i Danmark blive afbrændt. Dette fører til forurening af det omgivende miljø med giftige tungmetaller, ligesom det medfører et stort ressourcetab (Bigum et al., III).

Forbedringer i forhold til behandlingen af de specielle affaldstyper er nødvendigt. Traditionelle forbehandlingsanlæg synes primært at fokusere på de traditionelle metaller som jern, aluminium og kobber, som kan genanvendes i større mængder. Genanvendelse af ædel- og kritiske metaller sker i minde grad, da disse forefindes i meget mindre mængder. Fremtidige behandlingsanlæg bør dog fokusere på at genindvende disse metaller på trods af den mindre koncentration i affaldsstrømmen, da genanvendelsen af disse kan have større miljømæssige besparelser end de metaller, som traditionelt genanvendes (Bigum et al., I).

Livscyklusvurderinger (LCA'er) bruges som beslutningsværktøj i forbindelse med beslutninger på affaldsområdet. LCA'er skal derfor også kunne inkludere de problemstillinger, der relaterer sig til de specielle affaldstyper og håndteringen deraf. LCA'ers evne til at inkludere disse problemstillinger er højst nødvendige for at metoden fortsat kan bruges som beslutningsværktøj og også fremover berettige brugen af affalds-LCA'er når der træffes beslutninger.

En af disse problemstillinger relaterer sig til at de specielle affaldstyper som værende meget forskelligartet. Variationen i sammensætningen er stor og tilgængeligt data på sammensætningen få. Dette kan gøre det besværligt at inkludere de særlige affaldstyper i affalds-LCA'er. Dette betyder også at de miljømæssige aspekter af de specielle affaldstyper kan være vanskelige at vurdere til fulde og at konsekvenserne af disse risikerer at blive bliver overset eller undervurderet.

Miljøvurderinger af de særlige affaldstyper er et relativt nyt felt og mange problemstillinger mangler stadig at blive løst. En af disse problemstillinger er evalueringen af ressourcetab og knaphed. Dette område kræver, at der opnås meget bredere konsensus og generelt videnskabelig videreudvikling for at sikre at LCA er brugbart og accepteret som beslutningsværktøj.

Denne afhandling viser vigtigheden af at inkludere en detaljeret sammensætning af de specielle affaldstyper, samt vigtigheden af at inkludere ressourcetab af de ikke-genindvundne elementer i affalds-LCA'er (Bigum et al., III). Afhandlingen viser også at genanvendelse af metaller er af stor miljømæssig betydning (Bigum et al. I) og kvantificerer mængden af specielle affaldstyper, som fejlplaceres med husholdningsaffaldet (Bigum et al., II). Afhandlingen konkluderer også, at der stadig er mange problemstillinger, som kræver at blive løst og foreslår hvilke områder, der kræver videre forskning for at forbedre feltet for miljøvurderinger af de specielle affaldstyper.

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Abbreviations

AC	Acidification
CC	Climate change
CRT	Cathode ray tubes
DTU	Technical University of Denmark
EASETECH	Environmental Assessment System for Environmental TECHnologies
EASEWASTE	Environmental assessment of solid waste systems and technologies
EP	Eutrophication potential
ET	Ecotoxicity
EU	European Union
GHG	Greenhouse gas
GW	Global warming
HT	Human toxicity
ILCD	International Reference Life Cycle Data System
ISO	International Standardisation Organisation
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
OD	Ozone depletion
PE	Person equivalents
POF	Photochemical ozone formation
RD	Resource depletion
sWEEE	small Waste Electrical and Electronic Equipment
TS	Total solids
VS	Volatile solids
UNEP	United Nations Environment Programme
WEEE	Waste Electrical and Electronic Equipment

1 Introduction

1.1 Background

Resource efficiency, sustainability, zero waste societies, circular economy, raw material independency, and resource depletion are issues of current global concern governing the political scene related to the environment and waste management (UNEP, 2009; CEC, 2005a; Auken, 2012). In 2010, the European Union (EU) issued a working paper titled "Sustainable materials management for a resource efficient Europe" (EU, 2010). This initiative was a political culmination driven by a global growing demand for raw materials, and a focus on green growth and job creation, based on sustainable technologies as one of the ways out of the economic crises and to ensure future growth. The working paper was supplemented by the report "Critical raw materials for the EU" (ECEI, 2010), which concluded that there are raw materials for which there is an immediate supply risk that could hinder future development of emerging green technologies. The inherent resources in waste have obviously become a focus as a source of critical raw materials. The idea of waste as something that needs to be disposed of seems to have been effectively abandoned, and considering waste as a resource seems to be a paradigm shift that is here to stay.

Waste management in Denmark and many other countries is increasingly focusing on special wastes, as these constitute a special threat to the environment due to their content of hazardous materials and due to the valuable resources they contain.

Waste Electrical and Electronic Equipment (WEEE) and batteries are some of the special waste types receiving significant focus as hazardous and valuable waste types. The importance of safe treatment and recycling of WEEE and batteries was recognised when the EU implemented the European WEEE Directive (CEC, 2003; revised in 2012) as well as the European Battery Directive (CEC, 2006). It is considered crucial that these waste types are efficiently collected and recycled.

There are significant environmental benefits to be obtained when recycling special wastes. Dodson et al. (2012) consider municipal solid waste as being one of the largest potential resources for the recovery and recycling of scarce elements. WEEE and batteries, which are not sorted out for recycling and recovery, do not only represent a loss of materials and metals but could also lead to pollution of other waste streams (Bigum et al., III; Hischier et al., 2005; Wäger et al., 2011). The recovery of precious and rare metals to replace the primary production is considered to have especially significant environmental benefits (Bigum et al., I; ECEI, 2010; Schüler et al., 2011).

The concerns about the criticality of raw materials, as raised by the European Commission, disregard the notion of the geological scarcity of materials due to the difficulties of assessing shortages, within the considered time horizon, and the problems of using global reserve figures as reliable indicators (ECEI, 2010). Resource depletion is, however, a fundamental sustainability issue and should be included when conducting assessments (Klinglmair et al., 2013).

Life cycle assessments (LCAs) are increasingly being used as a decision-making tool for waste management, and are recognized as capable of supporting waste management decisions to the extent that they are recognized in the waste framework directive as one of the few reasons for diverting from the waste hierarchy (CEC, 2008). The ability for LCA as a tool to incorporate the issues related to the special waste types is, therefore, crucial in assuring that sound decisions are made. Currently, very few LCA studies on special waste management have been conducted (Hischier et al., 2005; Huisman et al., 2007; Wäger et al., 2011; Bigum et al., (I and III)), and the incorporation of special waste-related issues in the LCA methodology is a work in progress, which needs further development. A better understanding of the waste management routes that special waste undertake and specific knowledge of the treatment facilities (e.g., material losses during processing and overall recovery efficiencies) is highly necessary. Also, the establishment of a solid data background for evaluating resources, and knowledge of the content and behaviour of toxic and hazardous compounds in special waste, is necessary to ensure that these issues are adequately included in environmental assessments.

1.2 Objectives

The thesis mainly focuses on special waste types, WEEE, and portable batteries from households. The work was based on Danish and European conditions, as the management of these waste types is typically transnational. Special attention was devoted to WEEE, as this is a highly interesting and complex waste type, where many of the conclusions, and resulting LCA framework, are considered to be applicable for other special waste types, as well.

The main objectives of this PhD thesis are to identify important issues related to environmental assessments and waste-LCAs of special waste types and to contribute to a broader knowledge on the challenges and perspectives for waste-LCAs on special waste. The objectives of the thesis are meant to further improve the incorporation of the identified issues of special waste (WEEE, batteries, ink cartridges and cables) in LCAs. The thesis does not aim to provide the full answer or complete methodology relating to conducting waste-LCAs on special waste, as this would be very comprehensive.

The objectives can be summarised as follows:

- To quantify and characterize special waste that has been misplaced with residual household waste in Denmark;
- To make recommendations that can improve collection of misplaced special wastes based on the quantification and characterisation;
- To contribute to the further improvement of special waste in waste-LCAs by establishing Life Cycle Inventories (LCIs) for the recycling and recovery of certain metals and by providing necessary composition data of special waste based on available literature;
- To identify and discuss issues related to waste-LCAs of special waste that are particularly important;
- To contribute toward the establishment of a framework for waste-LCA modelling, in particular by including unrecovered elements in the resource depletion impact category.

1.3 Content of the thesis

The structure of the thesis is as follows:

Chapter 2: Provides the setting and background for the thesis. Describes the collection and management system in Denmark, and the amounts of collected and treated special waste. Presents the current status and goals for collection and recycling.

Chapter 3: Describes the methodological approach used to assess the environmental impacts of special waste management. The methods used in the project (waste sampling, characterisation and LCA) are described.

Chapter 4: Quantifies the flows of special waste in Denmark with a particular focus on the misplaced special waste. The misplaced special waste is characterised, and the significance of misplacement as a special waste flow is assessed. Measures to improve the collection of special waste are suggested.

Chapter 5: Describes the environmental aspects related to special waste with special focus on WEEE and batteries, hazardousness waste and potential resources.

Chapter 6: Presents and discusses the main findings and conclusions of the waste-LCAs on special waste types conducted during this PhD study. This chapter deals with the issues of varying waste routing, flows, composition, treatment, and the LCA methodology itself.

Chapter 7: Highlights and discusses the most important findings of the research, thesis and the papers in relation to the challenges and perspectives of conducting waste-LCAs on special waste.

Chapter 8: Concludes and summarizes the outcome of the thesis and the recommendations based on the work in relation to special waste and lifecycle assessment of special waste types.

Chapter 9: Presents the issues and topics on special waste and waste-LCAs that could benefit from further research.

2 Special waste in Denmark

2.1 Collection and management

Denmark has a population of around 5.5 million and consists of 98 municipalities (Danmarks statistik, 2014). All households have access to recycling centres, where waste can be disposed of without charge.

WEEE and batteries are subject to producer responsibility and marketed and collected amounts have since 2006 (WEEE) and 2008 (batteries) been recorded in annual reports by the Danish Producer Association (DPA). In 2006, the collected amounts of WEEE and batteries were only reported on a national level. In 2009, the amounts of collected batteries were also reported on a municipal level, and in 2010 the same was done for WEEE.

In Europe, marketed and collected WEEE is reported according to the ten WEEE directive categories (Table 1) and as household and business waste, respectively. Batteries are recorded as portable batteries, car batteries or industrial batteries, which again are subcategorized into button cell, lead acid, nickel cadmium (NiCd) and others.

The WEEE directive categories are politically determined and do not reflect the reality of how these wastes are collected and managed (DPA, 2013). Categorising WEEE according to the ten WEEE directive categories has the benefit of allowing for comparison between EU member states.

Table 1: The ten WEEE directive categories (CEC, 2012)

#	Category name
1	Large household appliances
2	Small household appliances
3	IT and telecommunications equipment
4	Consumer equipment and photovoltaic panels
5	Lighting equipment
6	Electrical and electronic tools
7	Toys, leisure and sports equipment
8	Medical devices
9	Monitoring and control instruments
10	Automatic dispensers

WEEE is commonly considered to be treated in six different ways (Huisman et al., 2007). This was discussed in Bigum et al. (I), who coined these “treatment categories.” When conducting environmental assessments on the management and recycling of WEEE, this should be carried out using the treatment categories, as they reflect how WEEE is, in reality, collected and subsequently managed and recycled. These treatment categories and the individual WEEE directive categories are shown in Table 2.

Huisman et al. (2007) subdivide WEEE directive category 5 into luminaries (5a) and lamps (5b). By “luminaries,” Huisman et al. (2007) mean the apparatus for the fluorescent light tubes but exclude household luminaries. Instead Huisman et al. (2007) include household luminaries in WEEE directive category 2 as a part of “small household appliances.” In Denmark, luminaries for households were previously exempt from producer responsibility (DPA, 2012). This was rectified in 2010 by the Danish EPA, and household luminaries are now included under WEEE directive category 5 as “5a” (DPA, 2012). The WEEE directive (CEC, 2012) exempts household luminaries from WEEE directive category 5 “lighting equipment,” but does not specify where household luminaries should then be included. In this thesis, luminaries from households are considered as category 5a “lighting equipment.” The term “lamps” (treatment category 5b) is used for the actual “light source” or bulb (excluding filament bulbs, which are not considered WEEE, according to the WEEE directive (CEC, 2012).

Table 2: The six most common treatment categories for WEEE in Europe (Huisman et al., 2007) and their associated WEEE directive categories. Small WEEE (sWEEE) is typically collected together and can then be further separated into a low-grade and a high-grade fraction (Bigum et al., I).

Treatment categories	WEEE directive categories
Large household appliances	1, 10
Cooling white goods	1, 10
sWEEE: Low-grade fraction	2, 5a, 6, 7, 8, 9
sWEEE: High-grade fraction	3, 4
TVs and monitors	4
Lamps	5b

Because WEEE has to be reported, according to the directive categories, but it is collected and weighed according to the treatment categories. It is, therefore, necessary to use a distribution key to calculate from one to the other (Table 3). The distribution key is determined by a number of sorting tests conducted in 2008 by the collectors and recyclers of WEEE and is only applicable for household WEEE (DPA, 2013). Because WEEE directive category 10 is not considered household WEEE, it does not appear in Table 3.

The distribution key needs to be regularly updated to ensure that the reported data is up to date with the technological development for EEE. However, it has not been possible for the DPA-system to verify that the distribution key is still valid (DPA, 2013).

The distribution key, in some ways, differs from the definition of the treatment categories as presented in Table 2. This is, for instance, the case for small WEEE (sWEEE), which in Table 3 is not divided into high-grade and low-grade WEEE. This is because sWEEE is not collected as high-grade and low-grade, respectively. Separating according to high-grade and low-grade sWEEE is something that can be done later on by the recycling industry, if this is deemed economically feasible. The treatment that high-grade sWEEE and low-grade sWEEE undergo is, however, often the same (Bigum et al., I). Another difference is that IT and telecommunications equipment (WEEE directive category 3) is allocated to the treatment category "TV and monitors." In 2011, the DPA-system issued a guideline on how to sort Danish WEEE (DPA, 2011b). In the guidelines, it was specified that the content of the treatment category "TV and monitors" was to include "equipment containing monitors." A primary reason for this change was due to technological development. Personal computers (PCs) used to consist of a screen and a computer. The computer would be considered "sWEEE," and the screen, which was easily detachable, belonged to the treatment category "TV and monitors." With the introduction of small laptops and tablets (where the screen is not apparently detachable), it was necessary to clarify which category these equipment types should belong to. With the 2011 guideline, it was decided that these items should be considered as "TV and monitors" (DPA, 2011b). Another difference is that sWEEE contains equipment types belonging to WEEE directive category 10 (large household appliances). This is because some smaller equipment types for the sake of convenience are collected with sWEEE (DPA, 2011b).

Table 3: Distribution key for household WEEE used to convert from treatment categories (Table 2) to WEEE directive categories (Table 1) (DPA, 2013).

Treatment categories	*Allocated [%]*	*WEEE directive categories*
Large household appliances	*100*	*1*
Cooling white goods	*100*	*1*
sWEEE	*3.7*	*1*
	17.3	*2*
	41.1	*3*
	30.9	*4*
	0	*5a*
	4.1	*6*
	0.8	*7*
	0.2	*8*
	1.9	*9*
TV and monitors	*16*	*3*
	84	*4*
Lamps	*100*	*5b*

Before the enactment of the WEEE directive in 2006, the responsibility of recording the amounts of collected electronic waste and batteries rested with the municipalities that reported back to the Danish Environmental Protection Agency (EPA). The change in 2009 with regard to the reporting scheme meant that the producer organisations (in practise, carried out by their partnership organisations) are now responsible for reporting the collected amounts, even though the municipalities still manage the collection. In the transition period, this resulted in some uncertainties in the reported data. Irregularities in reporting still occur. The reporting protocols are, however, continuously being optimized, and the data background is improving (DPA, 2013). Among other things, WEEE and batteries collected by one municipality are sometimes registered by a producer organisation in another municipality. This means that there can be uncertainties related to the collected amounts registered for the individual municipalities, which can make it difficult to evaluate collection on a municipality level (e.g., with regard to their dedicated waste management schemes).

2.1.1 Collection of WEEE and batteries

With the enactment of the producer responsibility, the management and treatment of WEEE and batteries is now the responsibility of the producers. In Denmark, this is put into practise by the municipalities continuing to manage the collection, but the collection costs are now to be covered by the producer organisations.

The choice of collection schemes for WEEE and batteries is decided by the individual municipalities. Municipalities can, in addition to the recycling centres, chose to offer additional dedicated collection schemes for the special waste types. These can be grouped into three types: full service, public collection points, and curbside collection.

A "full service" system is a system where the waste is collected frequently and directly at the citizen's home or curbside and is considered to be the most efficient system, because it is the most convenient for the citizens. Single-family households have individual waste bins, and it is therefore possible to introduce a bag/box system for special waste to be collected simultaneously with the residual household waste. Multi-family households do not have individual waste bins, and a full service system for multi-family households consists of a "joint full service," where a separate container would be placed in the common courtyard (Bigum et al., II).

"Public collection points" consists of placing boxes or containers in public areas where people regularly pass by, e.g., near supermarkets, libraries and schools.

"Curbside collection" refers to an arrangement where a municipality collects certain waste types, including special waste, at scheduled dates a certain number of times per year. The citizen would have to leave the waste by the curb on these specific dates, and, in most cases, alert the municipality in advance to ensure that there is waste to be collected (Bigum et al., II).

2.1.2 Management of the producer responsibility

In Denmark, the producer responsibility is managed by the producers entering partnership organisations. The partnership organisations will then collectively enforce the responsibility of ensuring that the collected WEEE and batteries are retrieved from the municipalities and subsequently managed. There are five partnership organisations in Denmark: Elretur, ERP Denmark aps, LWF, RENE and ReturBat (DPA, 2014). These include slightly more than 1,000 producers and importers of EEE. In addition, about 660 producers have maintained individual responsibilities (DPA, 2013). The partnership organisations are commercial organisations, which are organised differently with different focuses on e.g., customers and treatment fractions, but where there is still a level of competition maintained with each other. The individual producers will, based on their marketed amounts, be responsible for the management and treatment of a certain amount of WEEE or batteries. In practice, this responsibility will only be financially feasible if the partnership organisations actually implement the responsibility and contract businesses capable of conducting the treatment and management. The financial share of the treatment costs that the producers are responsible for is determined by the distribution key (Table 3) (DPA, 2013).

2.1.3 Collected and managed amounts of WEEE and batteries

Besides reporting the marketed and collected amounts of WEEE and batteries, the producer organisations are also required to report how the collected special wastes are managed.

In 2012, 75,134 tons of household WEEE and 1,511 tons of portable batteries were collected for recycling (DPA, 2013). Eighty-five percent (85%) of the collected WEEE (including around 1,072 tons of business WEEE) was considered recycled, meaning that it was sent to a recycling facility for further processing. Nine percent (9%) was incinerated, and 6% was deposited or emitted (e.g., as volatile compounds) during treatment. Sixty-one percent (61%) of the WEEE was pre-treated in Denmark, and the remaining 39% in the EU (DPA, 2013). The recycling rate of the collected batteries is given according to type: button cells, lead acid batteries, NiCd batteries and "others." Lead acid batteries are almost all recycled (99.6%), 82% of the NiCd was recycled, 73% of the button cell batteries containing mercury (Hg) were recycled, and 60% of the category "others" was recycled (DPA, 2013). The recycling or other treatment of portable batteries is grouped together with car batteries and industrial batteries and are not reported individually. It is, however, estimated that the recycling rate of these is in the area of 56-64% (DPA, 2012). The geographical location of the battery recycling was not disclosed in DPA (2013).

The producer organisations report on the treatment form, with respect to the facilities, which initially receives the waste, and not according to the final destination or treatment of the waste. "Recycled" thus means that the waste was initially received at a pre-treatment facility and does not necessarily mean that all of the waste was eventually recycled. Pre-treatment in Denmark primarily consists of some manual depollution followed by shredding. The shredded fractions are then traded as secondary raw materials on the international market (DPA, 2013). The further processing of the secondary raw materials might very well lead to increasing waste flows that are not recycled but otherwise utilized or deposited. DPA (2013) reports on the efficiency percentages for the initial recycling (pre-treatment), according to the ten WEEE directive categories. The efficiency percentages are given as "recycled" and "utilized," respectively (DPA, 2013). "Recycled" means that the waste has been processed "with the purpose of further recycling." "Utilized" covers both the amounts which are "recycled" and the amounts which are utilized. DPA (2013) states that, depending on the WEEE directive categories, the recycling efficiency is 79-96% and the utilization efficiency from 91 to 98%. This seems high, but since only the initial pre-treatment is accounted for, and because there are no definitions as to the quality or recyclability of the outputs from these, it is difficult to know how much WEEE in the end is actually recycled and utilized.

Collected special waste is considered a commodity and is typically traded to the highest bidder (for the valuable fractions) and the treatment facility requiring the lowest cost (for the fractions that have little value but still require treatment). Due to the producer responsibility and the competition between the partnership organisations, many of the details of special waste management are considered to be trade secrets. That economic factor plays a role in the waste routing of the special wastes, making it difficult to track the flows, which also increases the risks of illegal shipments of waste (EEA, 2009).

Uncollected WEEE and batteries

In 2012, there was a gap of 41,626 tons (corresponding to 36% of what was marketed) between the marketed EEE and the collected household WEEE. In 2011, the gap was 31,946 tons (DPA, 2013). The gap between marketed EEE and collected WEEE seems to be consistent, as can be seen from the preceding years (DPA, 2008; DPA, 2009; DPA, 2010; DPA, 2011a; DPA, 2012). EEE has varying lifetimes, and it cannot be expected that marketed EEE will become WEEE the following year. Predicting generated amounts of WEEE depends, among other things, on the saturation level (EEA, 2003). A fully saturated market for WEEE is a market where the purchase of a new product leads to the disposal of the same quantity of waste (UNEP, 2007). However, in reality, and especially considering technological development, the saturation level is more closely related to the number of items rather than the quantity. Saturation levels also depend on the product type. For some products, the European market shows signs of saturation of items such as larger household appliances, like refrigerators, washing machines, and television sets. For other products, the market seems to be unsaturated (e.g., IT and telecommunication equipment, and electronic toys) (EEA, 2003). The market for EEE can be hard to predict. A developed country like Denmark can be considered to be a saturated market for EEE, overall, which means that this gap ought to be closing as the amount of input (marketed EEE) would equal the amount of output (produced WEEE) (UNEP, 2007; DAKOFA, 2012). If this assumption is valid, it suggests that there are unaccounted flows of WEEE in Denmark.

There is also a consistent gap between marketed and collected portable batteries. In 2012, this gap was 2,193 tonnes, corresponding to 59% of the marketed amounts (DPA, 2013). The gap for portable batteries also seems to be consistent as can be seen from the preceding years (DPA, 2008; DPA, 2009; DPA, 2010; DPA, 2011a; DPA, 2012). The consumption of portable batteries can largely be related to technological development, as batteries are used as a power supply in EEE categories, which do not show signs of saturation (for IT and telecommunication equipment, and electronic toys).

Misplacement of special wastes with the residual household waste might account for the difference between marketed and collected WEEE and batteries. This was investigated and is discussed in Chapter 4.3.

2.1.4 Collection and recycling: Status and goals

Collection

The WEEE collection goal was, until 2012, defined by the European WEEE directive to be 4 kg WEEE per inhabitant per year from private households (CEC, 2003). From 2007 to 2011, Denmark collected an average of 15 kg WEEE per inhabitant, and in 2012, 13 kg of WEEE per inhabitant was collected (DPA, 2013). This was well above the European collection targets, but the drop of 2 kg per inhabitant from 2011 to 2012 was unsatisfactory and not in line with Danish intensions (DPA, 2013). Collection targets given as a fixed amount does not take into account that richer countries that have a high consumption of EEE are also likely to have a high production and collection of WEEE. To address this, the newly revised WEEE directive (CEC, 2012) decided that the collection targets should be related to the marketed amounts of EEE in the individual countries. From 2016 to 2019, the collection targets in Denmark will therefore be 45% of the average marketed amounts (based on the three proceeding years) of EEE to both business and households (CEC, 2012).

Based on the marketed amounts in 2010-2012, this will mean a collection rate of 12 kg of WEEE per inhabitant per year. This is less than what is presently being collected and is inexpedient for Denmark, but the targets also apply to other member states and, for these countries, the proposed targets might be an improvement.

In 2019, the collection target for WEEE will increase to 65% of the average marketed amounts. Using the marketed amounts in Denmark from 2010-2012 as reference, this would mean that Denmark would have to collect 16 kg WEEE per person per year. This mandates that by 2019, Denmark will have to increase the collection of WEEE in order to fulfil the EU targets.

The targeted collection rates for batteries are a minimum of 25% of the marketed amounts for the three proceeding years, by 2012, and 45% by 2016 (CEC, 2006). Denmark has decided to expedite the 2016 goal for batteries to 2012. A 47% collection rate of batteries was obtained in 2011, but in 2012, the collection rate had dropped to 44.6%, meaning that Denmark is not meeting its own national goals (DPA, 2013).

Recycling

Overall, 85% of the collected WEEE in 2012 was sent for recycling (DPA, 2013). The European recycling targets are given per WEEE directive category, and the Danish results are well above these on all categories. With the new WEEE directive, the recycling targets are to increase by 5% per year from 2015 to 2018. This will have an influence on the recycling of large household appliances, lamps, and automatic dispensers, where Denmark will have to increase the amounts being sent for recycling (DPA, 2013).

The European targets for recycling of batteries are based on the type of batteries (button cells, lead acid, NiCd or others). The European targets for recycling are 50% for button cells, 65% for lead acid batteries, 75% for NiCd batteries, and 50% for others. In 2012, Denmark met and exceeded the European targets for recycling of batteries.

3 Methods

3.1 Waste sampling and characterisation

Bigum et al. (II) addressed the lack of information and knowledge regarding misplaced special waste in residual household waste by conducting a thorough waste sorting study. The study was part of six larger residual household waste sorting analyses performed in Denmark in 2010 and 2011 which involved 12 Danish municipalities (Petersen, 2011a; Petersen, 2011b; Petersen, 2012a; Petersen, 2012b; Petersen et al., 2012a; Petersen et al., 2012b). In this study, one week of residual household waste was collected from 3129 households (2272 single and 857 multi-family households) and manually sorted. In total, 26.1 tonne of residual household waste (on average, approximately 8.5 kg of waste per household) was sampled and sorted during 2010 and 2011.

The sampling was conducted by the municipalities that were asked to ensure that issues such as income, socioeconomic background and citizens' age profile were representative. However, the overall share of single-family households in the compiled six studies used in Bigum et al. (II) was higher (72%) in comparison to the overall Danish society (60%) (Danmarks statistik, 2014). Nonetheless, the study was considered to be representative of Denmark due to the large sample size.

Bigum et al. (II) took into account that the municipalities included provided different dedicated collection schemes for special waste. It was tested to determine if the different collection schemes influenced the results or if it could be justified in stating that the results could be considered as being representative of average Danish conditions. This was done by conducting a two-sided unequal variance *t*-test on the two systems: ''full service'' and ''public collection points,'' provided for WEEE and batteries, respectively. The *t*-test showed that the observed variations between municipalities could not be attributed to different collection schemes, meaning that the results could be seen as being overall representative for Danish conditions.

The waste sorting was carried out following a two-step procedure. Initially, the collected residual household waste was manually sorted, according to waste type (organic waste, paper and cardboard, hazardous waste, etc.). Following this, the special waste types underwent further detailed sorting.

Each battery was weighed and categorised according to type: primary batteries [alkaline, zinc carbon and button cells (including type of button cell)] and secondary (rechargeable) batteries (including type of secondary battery), and whether or not it was discarded as part of sWEEE equipment (built-in). Each WEEE item was weighed and categorised according to item type, WEEE directive category, and treatment category (see Table 1 and Table 2 for definitions). Cables that could be identified as belonging to a WEEE item or category were considered as WEEE (Bigum et al., II). Unidentifiable cables and chargers were in accordance with the WEEE directive (CEC, 2012), not considered as WEEE, and were, therefore, simply categorized as "cables." Toners were categorized as "ink cartridges" and considered to be a special waste type.

3.2 Determining the composition of special waste

The composition of special waste is very heterogeneous. The composition of WEEE especially varies significantly (Figure 4). The composition of the different WEEE directive categories, WEEE treatment categories, and even between the same items will vary from each other. The composition can vary with time, due to technological development (e.g., the layers of gold in printed circuit boards is decreasing with time due to technological advancements and increasing raw material prices). The heterogeneity makes chemical analysis of special waste very difficult and expensive, and data on mixed special waste types are therefore limited. Most of the conducted chemical composition analyses are done on a product level, where the composition reflects individual products rather than a mix [e.g., Johansson and Björklund (2009)]. Some analyses have been done on mixed special waste fractions but have focused on certain elements of interest or concern (Chancerel et al., 2009; Oguchi et al., 2013). Common to these studies are that the scope of the studies has been on optimising or assessing the treatment of special waste. They were, therefore, conducted on special waste that had been separately collected through the dedicated waste collection schemes and, to some extent, pretreated.

Dimitrakakis et al. (2009) conducted a study on WEEE that had been misplaced with residual household waste in Dresden, Germany. These researchers conducted a material composition analysis of the misplaced WEEE fractions (ferrous, non-ferrous, cables, electronic component fractions, etc.) They also included an analysis of the plastic fraction, which determined the different plastic types. The misplaced special waste in Dimitrakakis et al. (2009) differed when compared to the findings of Bigum et al. (II), as the Bigum et al. (II) sorting analysis also found a cathode ray tube (CRT). Differences could be due to locality and the different times that the studies were conducted or simply day-to-day variations, which could explain the presence of the CRT in one study and not the other.

Bigum et al. (III) conducted an environmental assessment of the incineration of special waste that had been misplaced with residual household waste. This required knowledge of the composition of residual household waste as well as the special waste types. A study by Riber and Christensen (2006) is considered to provide the most comprehensive chemical composition study of Danish residual household waste to date. However, Riber and Christensen (2006) did not include special waste as a separate waste type but aggregated this with others in the "other non-combustible fraction." Also, Riber and Christensen did not analyse for many of the scarce and precious metals. The chemical composition of the residual household waste, in terms of organic waste, paper and cardboard, glass, plastic, metal waste, other combustibles, and other non-combustible fractions (excluding hazardous and special waste) was detailed in Bigum et al. (III) based on Riber and Christensen (2006). Some errors appeared in Riber and Christensen (2006), which have subsequently been corrected (e.g., the content of Pb in the "other metal" fraction and the content of As, Cu and Zn in the ash fraction). The corrected values were never published but were incorporated in EASEWASTE (Kirkeby et al., 2006) and used in Bigum et al. (III).

The special waste types found in residual household waste might be too heterogeneous (infrequent, few and very diverse items) to obtain a chemical composition analysis within an acceptable uncertainty level (Morf et al., 2007). Due to this, and because the sample size (amount of misplaced special waste) from Bigum et al. (II) was quite small (merely 119 kg per 26.1 tons), it was decided to estimate the composition of misplaced special waste by combining available literature data instead of conducting a chemical analysis. This was done based on the sources presented in Table 4. The uncertainty of the robustness of this and the possible impacts on the results was then addressed by a sensitivity analysis in Bigum et al. (III).

Table 4: Data sources used to estimate the composition of residual household waste (RHW) in Bigum et al. (III)

Waste types	Amount in RHW	Material composition	Chemical composition
Organic	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Paper and cardboard	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Glass	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Plastic	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Metal waste (iron and aluminium)	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Other combustible waste	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Other non-combustible waste	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Riber and Christensen (2006) (updated values)
Hazardous waste	Petersen (2011), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a), Petersen et al. (2012b)		Estimated as 25% aluminium, 25% hard plastic packaging (from Riber and Christensen (2006) (updated values), and 50% unknown

Table 4 (continued): Data sources used to estimate the composition of residual household waste (RHW) in Bigum et al. (III)

Special waste	WEEE	Bigum et al. (II)		
	• *sWEEE*	Bigum et al. (II)		
	Ferrous metals		Dimitrakakis et al. (2009)	Assumed similar to the manually removed ferrous fraction in Chancerel et al. (2008)
	Non-ferrous metals		Dimitrakakis et al. (2009)	Not included due to lack of data
	Plastics		Dimitrakakis et al. (2009)	Dimitrakakis et al. (2009) (AAS results preferred over HXRF when these were available)
	Rubber		Dimitrakakis et al. (2009)	Not included due to lack of data
	Cables		Dimitrakakis et al. (2009)	Hischier et al. (2007a)
	PCBs		Dimitrakakis et al. (2009)	Oguchi et al. (2013), Salhofer and Tesar (2011), Friege (2012), Chancerel et al. (2008)
	LCD's		Dimitrakakis et al. (2009)	Wang (2009), Göetze and Rotter (2012) (indium and antimony); IUTA & FEM (2011) (indium)
	Capacitors		Dimitrakakis et al. (2009)	Not included due to lack of data
	Electr(on)ic components		Dimitrakakis et al. (2009)	Not included due to lack of data and uncertainty as to what this fraction entails
	"bonded" materials		Dimitrakakis et al. (2009)	Not included due to lack of data and uncertainty as to what this fraction entails
	"various" fractions		Dimitrakakis et al. (2009)	Not included due to lack of data and uncertainty as to what this fraction entails
	• *Lamps*	Bigum et al. (II)	Elijosiute et al. (2012); Poulsen et al. (2010) (Hg content)	
	• *CRT*	Bigum et al. (II)	Andreola (2007a)	Andreola (2005), Andreola (2007a), Andreola (2007b), Tasaki et al. (2007), Mueller et al. (2012), Oguchi et al. (2013)
	Batteries	Bigum et al. (II)	Fisher et al. (2006), EPBA (2007), Fjelsted (2007) (compiled)	
	Ink cartridges	Bigum et al. (II)	Salhofer and Tesar (2011), Ruan et al. (2011), Hischier et al. (2007b)	
	Cables	Bigum et al. (II)	Hischier et al. (2007a)	

3.3 Life cycle assessment

LCA is an international standardized method, which allows for environmental assessments of complex systems. LCA provides quantitative information putting potential environmental benefits and burdens into perspective (ISO, 2006a; ISO, 2006b). According to the ISO standard 14040 (ISO, 2006a) LCA consists of four phases:

- Definition of goal and scope
- Inventory analysis and construction of life cycle inventories (LCIs)
- Life cycle impact assessment (LCIA)
- Interpretation of results

The definition of goal and scope includes describing the purpose and aim of the study. It also includes stating the functional unit, the system boundaries, and the assessment criteria, which will be used. The goal and scope definition should enable the reader to understand in which context the results of the study apply and to which it does not.

The inventory analysis is a critical point as it entails collecting and processing all the necessary input data (LCIs) required for the LCIA. This phase is typically the most time consuming and demanding, as it is important to collect relevant data of good quality, to reflect the study.

LCIA is the phase where the environmental impacts are expressed per impact category or assessment criteria. LCIA results are obtained by multiplying all emissions that contribute to an environmental problem (the LCIs) with a common characterisation unit (e.g., kg CO_2-eq.) for that given environmental impact. The LCIA results are given per impact category in different units and can therefore not be compared directly or summed up to one total environmental score.

Commonly used impact categories in LCA are:

- Climate change (CC) [alternatively global warming (GW)]
- Ozone depletion (ODP)
- Human toxicity (HT)
- Photochemical ozone formation (POF)
- Acidification (AC)

- Eutrophication potential (EP)
- Ecotoxicity (ET)
- Resource depletion (RD) [can be further subdivided into $RD_{abiotic}$ and RD_{fossil} (Bigum et al., II)]

Interpretation of the results can include normalisation or weighting, but it is not mandatory. Normalising an impact category is done by dividing the LCIA results with a normalisation factor that is representative of the overall inventory. A commonly used normalisation factor is person equivalents (PE), where one PE refers to the impact caused by all the activities of one person in a given year (Wenzel et al., 1997). Normalised results only reflect the individual environmental impact categories and can—similar to the LCIA results—not be compared or summed up to one environmental score (ISO, 2006a). Weighting can subsequently be done by multiplying the normalised LCIA data with weighting factors. Weighting factors reflects the relevance of the different environmental impacts, which is inevitably politically decided. The weighting step thus ascribes relevance to the impact categories and ranks them according to concern. Weighting allows for comparing the impact categories directly or summing the results into one environmental score, which can make it easier to make a decision based on the LCIA results. Weighting is, however, not scientifically founded and should be used with care, and should always be used with transparency, so that the results can be properly discussed (ISO, 2006b).

3.4 Waste-LCAs

3.4.1 Necessary requirements

When conducting waste-LCAs, one would typically need knowledge and data on the following:

- Waste flows and waste routing
- Waste composition
- Relevant treatment technologies
- A LCIA methodology that is able to reflect the environmental aspects associated with the issues with the modelled waste and scope of the study.

The necessary requirements in relation to the special waste are briefly described in the following.

The necessary input and knowledge depends on the goal and scope of the LCA study. If the goal is to assess or compare technologies knowledge on the actual size of the waste flows, this might not be relevant, as the functional unit could be "per tonne" [e.g., Bigum et al. (I) and (III)]. If, on the other hand, the goal is to assess the environmental impacts arising from the waste management of an area (e.g., municipality or country), knowledge on the size of the waste flows would be required (Hischier et al., 2005). In both cases, waste routing is an important first step in identifying the relevant technologies and for obtaining the necessary LCI data. In the case of special wastes, which are subject to producer responsibility and traded as a commodity, this is not always possible.

Data on the fractional composition and chemical content of the special waste is necessary for any modelling of management and treatment processes, as there is often a direct correlation between content and impacts from the treatment (Bigum et al., III). Information on the composition is necessary both when assessing recycling scenarios, where materials are recovered, and other treatment scenarios (or lack thereof) where environmental burdens might arise.

LCI data on the treatment processes include the use of auxiliary materials, process specific emissions, elemental partitioning or transfer coefficients, recovery efficiencies of materials (including any losses during the processes), and solid residues that require further management. Treatment technologies can vary on these parameters, and it is important to obtain data of good quality that reflects the study. The use of generic data should, therefore, be used with this in mind and should be evaluated thoroughly in order to ensure that they reflect the study.

Special wastes are typically very complex in terms of material composition. An environmental assessment should be able to incorporate this in order to provide a complete and sound assessment. The heterogeneity of special waste also sets requirements for the LCA tool, which needs to be able to assess the environmental impact of many different compounds as well as being able to evaluate the resource aspects associated to these.

3.4.2 Modelling principles

The JRC (2011) describes three archetypal goal situations: A, B and C. Only goal situation C and C1 were used in this PhD thesis and will be described briefly here.

Goal situation C is referred to as “accounting” and can be further subdivided depending on whether it includes interactions with other systems (C1) or excludes interactions with other systems (C2). Goal situation C deals with environmental accounting or monitoring of a system without considering the large-scale consequences of the analysed system. Goal situation C is only preparatory, meaning that decisions are not to be based solely on these. C1 describes a system that accounts for interactions with other systems by, for instance, crediting avoided burdens from recycling (JRC, 2011).

Within waste-LCAs, two modelling principles (or frameworks) are often used: “attributional” and “consequential” (Christensen, 2011).

Attributional modelling can be described as “descriptive” and accounts for the flows and processes in a system at a given time (JRC, 2011). The background processes of the study are modelled as average data (e.g., uses an average energy mix) as opposed to marginal data. As an LCI method approach, attributional modelling will use allocation where a system provides more than one function (Christensen, 2011). Allocation should be done according to a common technical aspect of the functions of the studies system.

Consequential modelling is a change-orientated approach with the aim of identifying what consequences changes to a system might have (Christensen, 2011). The background processes of the study will use marginal data (e.g., electricity production from coal rather than an average energy mix) rather than average data. As an LCI method approach, consequential modelling will seek to use system expansion or substitution instead of allocation (Christensen, 2011).

Another modelling principle, applied specifically for waste-LCAs, is the “zero-burden approach.” Using the zero-burden approach means that waste is considered not to carry an environmental rucksack, when it enters the waste management system (Cleary, 2010). In reality, the zero-burden approach seem to be largely related to the issues of resource consumption as waste or input-specific emissions are not excluded but will be accounted for in the other environmental impact categories (Riber, 2007). Using a zero-burden approach in a consequential framework, means that any environmental burdens arising from the waste, but where neither of the investigated waste management scenarios addresses this (or where they perform equally bad) will not be included in the LCIA. This means that there are issues that could be overlooked in the decision-making process.

As a way to address this, and by using the goal situation C1, Bigum et al. (III) modelled the incineration of residual household waste considering that all elements in the waste are a burden on the resource depletion impact categories unless they are recovered. Bigum et al. (III) coined this “the resource burden approach.”

3.4.3 Modelling tools – EASEWASTE and EASETECH

EASEWASTE and EASETECH are waste LCA-model tools developed at the Technical University of Denmark (DTU). EASEWASTE is the original model, which has been used widely since 2006, whereas EASETECH is a recent and updated model, which builds on the principles of EASEWASTE. EASETECH is, therefore, intended to replace EASEWASTE. Both models were used in the course of this PhD research [EASEWASTE in Bigum et al. (I) and EASETECH in Bigum et al. (III)], and both will, therefore, be briefly described.

EASEWASTE is an LCA tool with particular focus on assessing waste management aspects in a life cycle perspective (Kirkeby et al., 2006). EASEWASTE uses the EDIP method for quantifying “environmental impacts” and “resource consumption” (Wenzel et al., 1997). The impacts and resources are normalized into PE by the normalizations references presented in Stranddorf et al. (2005).

EASETECH is based on the same principles as EASEWASTE and the knowledge gained with waste-LCAs over the last ten years. Like EASEWASTE, it uses mass balances to account for the elementary exchanges and is able to evaluate the environmental impacts of the final destinations of the elements as it includes comprehensive material flow modelling. This means that it is possible to characterise each flow according to properties and composition and, as a result, also is able to trace and identify the cause of the environmental impacts. This tool has been designed so that it is easier and more user-friendly to set up scenarios than EASEWASTE. This means that a variety of modelling options are now possible (Clavreul et al., 2014).

EASETECH recommends using the International Reference Life Cycle Data System (ILCD) methodology (JRC, 2010; JRC, 2011; Hauschild et al., 2012) for the assessments, but other methodologies are available and possible to import to the programme. Likewise, it is now possible to import data from other data sources such as Ecoinvent (Ecoinvent, 2006), which is another new addition.

4 Special waste flows in Denmark

4.1 Estimating special waste flows

Estimating the flows of special waste is necessary when wanting to establish a basic understanding of the current situation, and it challenges these waste types. Quantifying and determining the flows of special waste in Denmark relies heavily on available statistical data being recorded. Marketed and collected amounts of WEEE and batteries are reported on a yearly basis by the producer organisations (DPA, 2013). However, other output flows, such as misplacement (with residual waste or other collected waste fractions), illegal shipments, theft from recycling stations, and uncollected waste (e.g., stock piling) are not similarly accounted for.

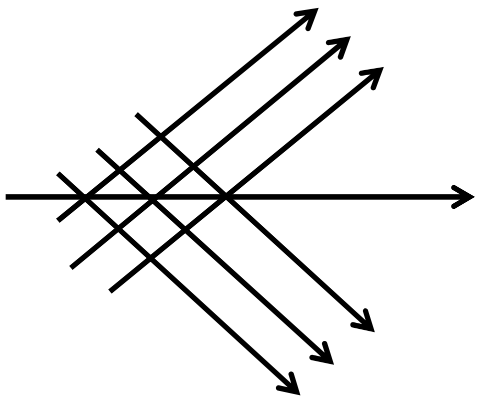
WEEE is often stated as being one of the fastest growing waste streams (EEA, 2003; Morf et al., 2007; Widmer et al., 2005). However, how do we predict the size of these arising flows when wanting to plan for future waste management?

A simple mass balance model where

Marketed amounts = amounts collected for recycling + stockpiling

is not possible in the case of special waste as there are many other outputs which have not been determined or quantified. Estimating the flows of special waste is challenging. Several international papers and reports on WEEE have covered this aspect, intensively, trying to determine models, which are able to incorporate the many variables (sales data, lifetimes, storage, and saturation level), and to predict the arising flows (EEA, 2003; UNEP, 2007; Araújo et al., 2009).

Figure 4 sum up some of the variables and difficulties with estimating and predicting the flows of WEEE. One of the aspects that make it difficult to predict and estimate WEEE flows is the heterogeneity of the waste. Heterogeneity appears with respect to size and weight between different categories, within each category, and even for individual product types, which can vary in size over time. Many products have become multi-functional. Thus, to some extent, they are replacing certain items completely. As an example, mobile phones now contain multiple functions, such as calendars, cameras and computer functionalities. Other variables affecting the flows of WEEE are: increase or decrease in population size; decreasing prices on EEE, due to automated processes; increasing prices on EEE, due to increases in metal prices; purchasing power or state of the market (boom or slump); the introduction of new product types or applications (e.g., e-books) on the market; technical developments, making products obsolete at an accelerated rate, and shorter life spans. The latter could be due to planned obsolescence and due to consumers discarding their products before they have reached their actual technical life times (e.g., due to difficulties to repair or simply to update the product, or that it has become “out of style.”)



- New product types
- Shorter lifetime of products
- Higher purchasing power and lower prices due to automated processes

Net flow?

- Smaller products
- More applications per product
- Lower purchasing power and higher prices due to increase in metal prices

Figure 1: Examples of influencing variables on the net flow (amount) of WEEE.

4.2 Misplaced special waste with residual household waste

Special waste ought to be disposed of via the dedicated waste collection schemes, and should not be misplaced with the residual household waste. Nonetheless, misplacement of these waste types happens and is rarely accounted for.

4.2.1 Content and type

Bigum et al. (II) conducted a sorting analysis of residual household waste in order to determine to what extent misplacement of special waste occurs and what special waste types and fractions are misplaced.

Bigum et al. (II) found that, on average, a Danish household misplaces 29 g of WEEE, 4 g of batteries, 1 g of ink cartridges and 7 g of cables per week with the residual household waste. This constitutes 0.34% (w/w), 0.04% (w/w), 0.01% (w/w) and 0.09% (w/w) of the residual household waste respectively. The 29 g of WEEE corresponds to a yearly average of about 7 items per household and the 4 g of batteries corresponds to a yearly average of 9 batteries per household.

WEEE

Bigum et al. (II) analysed and characterised the misplaced WEEE, both in terms of weight (w/w) and number of items (n/n), and according to the WEEE directive categories and treatment categories. The characterisations, according to WEEE directive categories, are shown in Table 5. The characterisations, according to the treatment categories, can be found in Bigum et al. (II).

Table 5: Misplaced WEEE with residual household waste, both in terms of weight (w/w) and number of items (n/n), characterised according to WEEE directive category (Bigum et al., II)

WEEE directive categories,	Amount [g $hh^{-1}week^{-1}$]	Distribution [%,w/w]	Distribution [%, n/n]
1. Large household appliances	0	0	0
2. Small household appliances	10	36	24
3. IT and telecommunication equipment	5	18	20
4. Consumer equipment and photovoltaic panels	8	29	24
5. Lighting equipment	1	3	11
6. Electrical and electronic tools	0.3	1	1
7. Toys, leisure and sports equipment	3	11	18
8. Medical devices	0	0	0
9. Monitoring and control instruments	0.3	1	2
10. Automatic dispensers	0	0	0
Total	29	100	100

hh: household

Bigum et al. (II) found that misplaced WEEE could primarily be characterised as: Small household appliances (WEEE directive category 2); IT and telecommunication equipment (WEEE directive category 3); consumer equipment and photovoltaic panels (WEEE directive category 4); and toys, leisure and sports equipment (WEEE directive category 7). Lighting equipment (WEEE directive category 5) consisted solely of lamps, as no luminaries were found, was on a weight base not found to be frequently misplaced, but when looking at the number of items, the opposite was true.

Electrical and electronic tools and monitoring and control instruments were misplaced in smaller amounts, and none of the categories 1 (large household appliances), 8 (medical devises) or 10 (automatic dispensers) were found. That larger items, such as categories 1 and 10, were not found can be explained by the residual household waste bins being generally smaller than these particular items. The low misplacement of medical devices could possibly be due to small amounts of marketed medical devices to households (DPA, 2013).

Bigum et al. (II) analysed and characterised the individual misplaced WEEE items. Figure 2 shows the weight of the individual WEEE items found in the residual household waste as a function of the WEEE directive category number and the average weight and standard deviation of the items. In general, the average weight of each item varied between 0.1 and 0.3 kg, with four much larger outliers (one CRT, one speaker set, one radio and one printer). The largest outlier was the CRT weighing 11 kg, which constituted 12% (w/w) of the overall amount of misplaced WEEE found in the residual household waste. The standard deviations are significant, and in most cases exceed the average, which supports the statement that WEEE is a heterogeneous waste type and shows that the notion of an "average weight of misplaced WEEE items" would not be representative.

Within the WEEE directive categories, some items seem to be discarded more frequently than others. Category 2 appears very mixed but electric tooth brushes (9% n/n) and wrist watches and clocks (10% n/n) seem to be misplaced more frequently. In category 3, the cable fraction was the most dominant item type (32% n/n), and, in category 4, headphones (46% n/n) and cables (25% n/n) dominated the fraction. Within category 7, toys (45% n/n) and flashlights and bicycle lights (25% n/n) were commonly found to be misplaced.

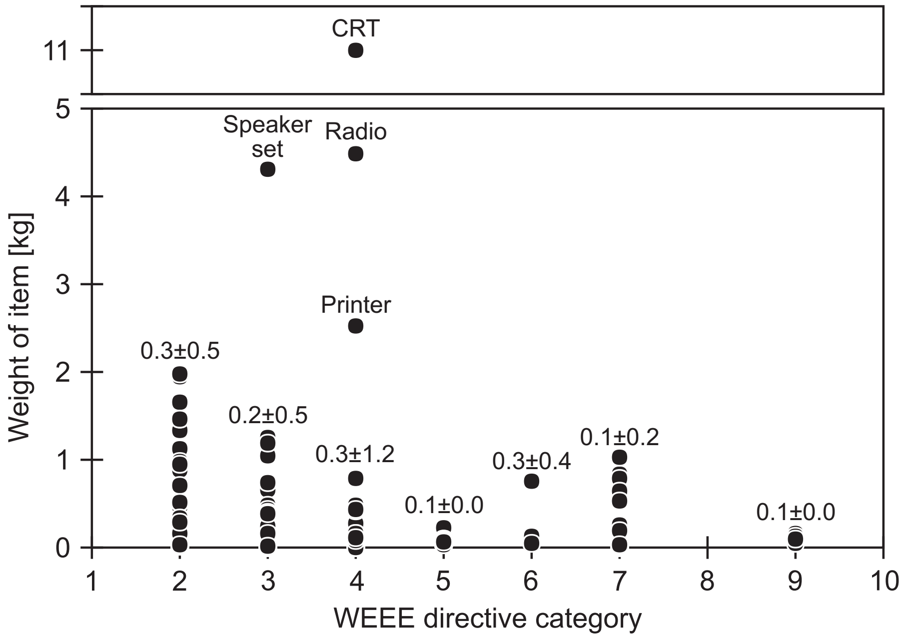


Figure 2: The weight of the individual misplaced WEEE items found in the residual household waste as a function of the WEEE directive category number. Each point in the figure corresponds to one single item. The figure also shows the average weight and standard deviation of the items within each WEEE directive category (Bigum et al., II)

Batteries

Bigum et al. (II) also analysed and characterised the misplaced batteries in terms of weight (w/w) and actual number of misplaced batteries (n/n), as well as recorded, if the battery was found to be a built-in WEEE.

The sorting analysis showed that 80% (w/w) and 79% (n/n) of the batteries in the residual waste were found as single portable batteries, and that the remaining 20% (w/w) and 21% (n/n) were built-in WEEE.

Figure 3 shows the weight of the individual batteries as a function of the battery type. The average weight of the batteries within each battery type appears together with the standard deviations. In general, the average weight of batteries discarded with the residual waste varied between 0.5 and 38.5 g. The standard deviation of the average weight of the different battery types is smaller than in the case of WEEE and showed that batteries misplaced with the residual waste seem to be relatively homogeneous in weight. It can also be seen that most of the alkaline batteries are within the range of 10–12 g and 22–24 g, and the zinc carbon batteries in the range of 6–8 g and 16–18 g corresponding to AAA batteries and AA batteries, respectively. The weight of the lithium and alkaline button cells was found primarily to be in the order of 0.5–2 g and 0.3–2 g, respectively.

The sorting analysis showed that alkaline batteries (69%, n/n) and zinc carbon batteries (11%, n/n) are frequently misplaced with residual household waste. Alkaline button cells do not seem to be frequently misplaced, as they only constitute 1% (w/w), but when looking at the actual number of alkaline button cells, they are actually frequently misplaced (12%, n/n). The sorting analysis also showed that silver oxide button cells, NiCd and lithium ion (Li-ion) secondary batteries are rarely discarded with the residual household waste, and that no zinc air button cell batteries, mercury oxide button cell batteries, and lead acid batteries were misplaced with the residual household waste. This could be due to the smaller marketed amounts of these battery types, but since statistical data on the marketed amount of portable batteries does not report on the different types of batteries, this information is not possible to determine.

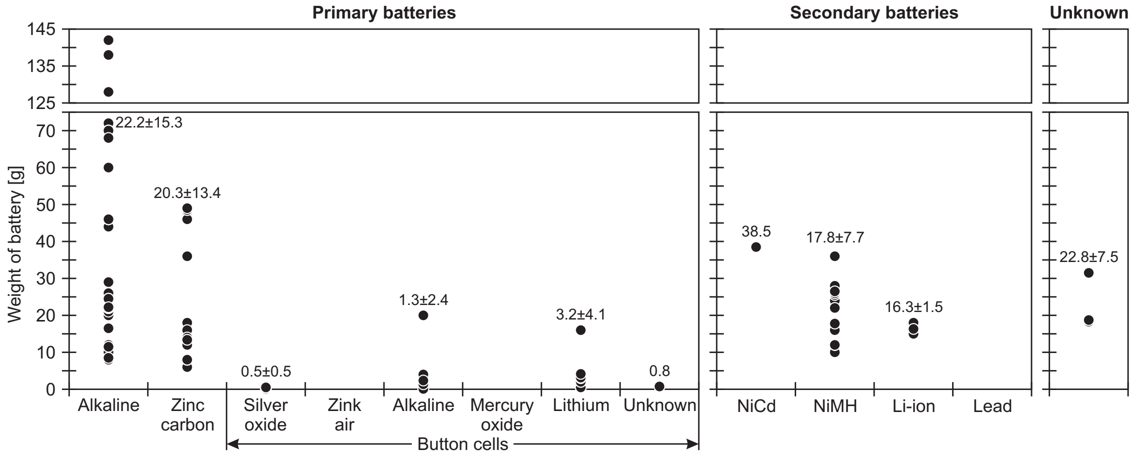


Figure 3: The weight of the individual batteries found in the residual household waste as a function of type. Each point in the figure corresponds to one single battery. The figure shows the average weight and standard deviation of the batteries according to type (Bigum et al., II)

4.2.2 Composition

Bigum et al. (III) estimated the composition of the residual household waste based on literature values (Table 4). The elemental compositions, reported for special waste, revealed a significant variation between studies and products. Figure 4 is an illustration of this and depicts the average, minimum, and maximum reported values of the different elements found in special waste. It can be seen that the average composition of the special waste is not necessarily reflective of misplaced special waste on a general level, as the composition is highly susceptible to the composition of the individual items being misplaced.

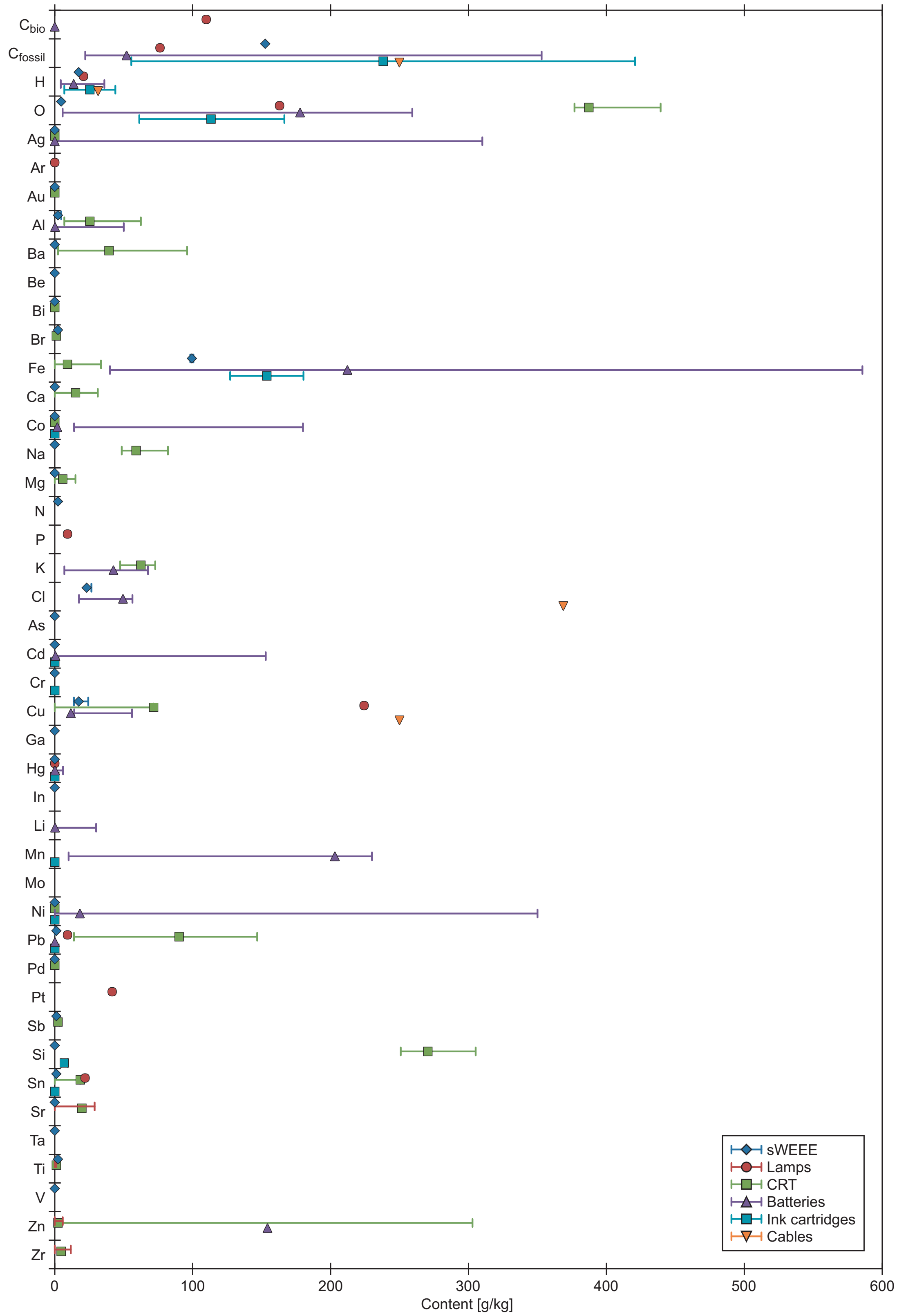


Figure 4: The chemical composition of the special waste types: sWEEE, lamps, CRT, batteries, ink cartridges and cables, along with the minimum and maximum reported values per element, misplaced with residual household waste (Bigum et al., III). The composition only applies for the part that could be accounted for (cf. bottom row in Table 6).

Average compositions for special waste were calculated, despite the significant variation used in Bigum et al. (III) and are presented in Table 6. The influence that the compositional variation of special waste might have on LCA results were in Bigum et al. (III), and were determined with a sensitivity analysis by using the minimum and maximum reported compositional values for the special waste.

Table 6 shows the average composition of 1 tonne of residual household waste (RHW) and the individual waste types that were used in Bigum et al. (III). The table shows the water, total solid (TS), volatile solid (VS), ash content, and lower heating value (LHV), in addition to the average chemical composition of the modelled waste types. Blank cells (-) means that no data were available, and cells with "0.0" are where measurements have been done, but where the given element was not found within the detection limit. If no data were available, they were not included in the LCA.

Overall, 89% of the chemical composition of residual household waste could be accounted for. Speculations regarding the remaining components are difficult. Glass was only 12% of that accounted for, which could be due, according to Riber and Christensen (2006), to not including silica (Si) in their study. If the remaining 88% of the glass fraction is indeed Si, it would reduce the unaccounted elements in residual household waste from 11% to 8%.

Of the individual waste types, more than 90% of the organic, paper and cardboard, plastic, metal and other combustibles waste can be accounted for. For the hazardous and other non-combustibles fractions, only 49% and 17% of the chemical composition could be accounted for. The content in the "other non-combustible" waste is ambiguous. It is, therefore, difficult to speculate what the remaining 83% of this waste fraction consists of.

Table 6 shows that the average chemical composition for CRT, batteries and light sources are largely accounted for with 109%, 94% and 68%, respectively. The chemical composition of CRTs is overestimated due to uncertainties in the literature values, which, in several sources, added up to more than 100%. The composition of cables seems to be accurately accounted for with 90% but is based on rough estimations (Bigum et al., III). This is due to few detailed sources, where none included data on the additives known to be present in cables. Ink cartridges and sWEEE were accounted for by 54% and 33%, respectively. One of the reasons for not being able to account for more of the sWEEE content is due to the material fractions "Electr(on)ic components," "Bonded material," and "Various" (see Table 4). These material fractions constitute 36% of sWEEE, and it was not possible to find studies that had analysed the content of these entities. Another example is printed circuit boards, where the available literature data was only able to account for 32% of the content. When using these chemical compositions, one has to take into account that these are likely to be underestimated. In Bigum et al. (III), misplaced Danish special waste could be described as consisting of sWEEE (61%), lamps (2%), CRT (11%), batteries (11%), ink cartridges (2%), and cables (13%). Overall, this meant that 56% of the chemical composition of the special waste fraction was accounted for.

Table 6: Chemical composition of the residual household waste types, including the overall composition of the residual household waste (RHW) used in Bigum et al. (III). The chemical composition is based on literature shown in Table 4.

	Unit	Organic Waste	Paper, cardboard	Glass	Plastic	Metal	Haz. waste	sWEEE	Lamps	CRT	Batte -ries	Ink cartr.	Cables	Other comb.	Non-comb.	RHW
Water	%	65.3	16.8	7	8	11.5	5.8	0.0	0.0	0.0	8.9	0.0	0.0	26.5	13.3	39
TS	%	34.7	83.2	93	92	88.5	94.2	100	100	100	91.1	100	100	73.5	86.7	61
VS	%TS	90.6	86.4	0.0	94.7	3.7	48.9	17.6	36.9	38.7	14.2	37.7	32.5	85.4	16.2	82
Ashes	%TS	9.4	13.6	100	5.3	96.3	51.1	82.4	63.1	61.3	85.8	62.3	67.5	14.6	83.8	18
LHV	MJ/kgTS	19.5	16.9	0.0	33.2	3.3	9.4	6.9	2.8	0.0	0.6	9.4	11.7	21	0.7	19
Ag	mg/kgTS	-	-	-	-	-	-	34.3	-	120	23.1	-	-	-	-	0.2
Al	g/kgTS	1.1	14.4	8	11.1	300	157	2.6	-	24.9	0.2	-	-	2.9	16.1	12
Ar	mg/kgTS	-	-	-	-	-	-	-	162	-	-	-	-	-	-	0.02
As	mg/kgTS	0.5	0.4	4.5	0.2	26.2	2.4	7.2	-	-	-	-	-	1.2	122	4
Au	mg/kgTS	-	-	-	-	-	-	4	-	5	-	-	-	-	-	0.01
Ba	mg/kgTS	-	-	-	-	-	-	164	-	39405	-	-	-	-	-	20
Be	mg/kgTS	-	-	-	-	-	-	0.1	-	-	-	-	-	-	-	$2.8 \cdot 10^{-4}$
Bi	mg/kgTS	-	-	-	-	-	-	7.7	-	280	-	-	-	-	-	0.2
Br	mg/kgTS	-	-	-	-	-	-	1809	-	673	-	-	-	-	-	5
C_{bio}	g/kgTS	491	415	0.0	3.66	62.4	-	-	110	-	3.1	-	-	374	28.9	380
C_{fossil}	g/kgTS	6.1	10.2	0.0	728	6.9	200	153	76.2		52	238	288	143	0.2	88
Ca	mg/kgTS	18914	26072	67844	8818	863	1049	45.6	-	14496	-	-	-	12237	24459	18620
Cd	mg/kgTS	0.1	0.1	0.1	0.1	1.1	-	2.6	-	-	528	0.02	-	0.4	1.1	0.5
Cl	g/kgTS	8.2	1.1	0.0	35.6	0.4	0.3	23.3	-	-	49.5	-	425	5.4	4.2	8
Co	mg/kgTS	-	-	-	-	-	-	3.2	-	58.5	1810	2.7	-	-	-	1
Cr	mg/kgTS	4.7	13.4	463	8.5	134	45.6	53.6	-	-	-	34.2	-	390	45.7	126
Cu	g/kgTS	0.01	0.1	0.01	0.1	1.3	0.3	17.3	224	72	11.7	-	250	0.5	12	1
F	mg/kgTS	100	227	0.0	100	15.6	25	-	-	-	-	-	-	100	45.1	113
Fe	g/kgTS	0.5	1	1.5	0.9	489	86.7	99	-	9.4	212	154	-	10.7	16.4	15

Ga	mg/kgTS	-	-	-	-	-	-	0.3	-	-	-	-	-	-	-	$8.4 \cdot 10^{-4}$
H	g/kgTS	67.9	57.7	0	103	11.2	26.3	17.4	20.5	-	13.7	25.1	36.3	69.7	4.9	64
Hg	mg/kgTS	0.1	0.0	0.1	0.0	0.3	-	1.8	28.8	-	52.5	0.3	-	0.1	0.1	0.1
In	mg/kgTS	-	-	-	-	-	-	0.2	-	-	-	-	-	-	-	$6.7 \cdot 10^{-4}$
K	g/kgTS	10.1	0.8	6	1	0.5	0.1	-	-	62.7	42.5	-	-	1.7	13	6
Li	mg/kgTS	-	-	-	-	-	-	-	-	-	202	-	-	-	-	0.1
Mg	mg/kgTS	1331	1278	8410	427	2087	3022	30.8	-	5787	-	-	-	1149	1890	1467
Mn	g/kgTS	0.1	0.03	0.1	0.03	3	1.3	-	-	-	203	0.1	-	0.1	0.4	0.3
Mo	mg/kgTS	0.7	0.9	1.5	1.4	15.1	0.9	-	-	-	-	-	-	1.3	6.1	1
N	mg/kgTS	33870	2829	0	6399	862.4	13750	2791	-	-	-	-	-	17212	2521	20370
Na	mg/kgTS	5058	1155	24356	1091	628.5	146.8	151.5	-	59491	-	-	-	12367	6718	6624
Ni	mg/kgTS	2.3	6.4	155.4	4.1	150.9	16.5	410	-	223	18212	5.1	-	27.8	75	29
O	g/kgTS	306	382	0.0	89.4	14.1	4.3	4.2	163	387	178	114	-	242	24.8	264
P	mg/kgTS	4848	268	92.2	4279	280	46.4	-	9032	-	-	-	-	1022	975	2722
Pb	mg/kgTS	4.5	4.7	77.1	17.6	2930	84.4	743	9698	90556	118	1.5	-	79.9	201	145
Pd	mg/kgTS	-	-	-	-	-	-	1	-	20	-	-	-	-	-	0.01
Pt	mg/kgTS	-	-	-	-	-	-	-	41453	-	-	-	-	-	-	4
S	mg/kgTS	2464	876	369	573	203	255	-	-	-	-	-	-	2564	1000	1963
Sb	mg/kgTS	-	-	-	-	-	-	707	-	2060	-	-	-	-	-	3
Si	g/kgTS	-	-	-	-	-	-	0.4	-	271	-	7	-	-	-	0.1
Sn	mg/kgTS	-	-	-	-	-	-	688	22343	18000	-	11.1	-	-	-	13
Sr	mg/kgTS	-	-	-	-	-	-	11	-	20111	-	-	-	-	-	10
Ta	mg/kgTS	-	-	-	-	-	-	2.7	-		-	-	-	-	-	0.01
Ti	mg/kgTS	-	-	-	-	-	-	2726	-	953	-	-	-	-	-	8
V	mg/kgTS	-	-	-	-	-	-	147	-	-	-	-	-	-	-	0.4
Zn	g/kgTS	0.1	0.1	0.04	0.1	0.8	0.1	0.5	-	2.3	154	-	-	0.6	8.2	0.5
Zr	mg/kgTS	-	-	-	-	-	-	-	-	4069	-	-	-	-	-	2
Accounted for [%]		96	92	12	99	90	50	33	68	109	94	54	90	90	17	89

4.2.3 Misplacement in western European countries

Bigum et al. (II), Bernstad et al. (2011), Dimitrakakis et al. (2009), and Huisman et al. (2012) are all western European compositional studies of residual household waste that have quantified misplaced special waste. Table 7 compares the findings of the four studies.

Bernstad et al. (2011) conducted a study of a Swedish residential area, before and after the introduction of source segregation. Excluding cables and filament bulbs [as this is not defined as WEEE according to the WEEE directive (CEC, 2012)], Bernstad et al. (2011) found that before the introduction of source segregation an average of 35 g of WEEE and 7 g of batteries per household per week was misplaced with the residual household waste. After the introduction of source segregation misplaced WEEE decreased to 18.4 g per household per week and misplaced batteries to 2.1 g. Dimitrakakis et al. (2009) (German study) and Huisman et al. (2012) (Dutch study) presented the misplaced amounts of WEEE in "kg per inhabitant per year". The two studies are therefore not directly comparable to Bernstad et al. (2011) and Bigum et al. (II). However if assuming that an average Danish household consist of approximately 2.2 inhabitants the 29 g WEEE per household per week (Bigum et al., II) calculates as 0.7 kg per inhabitant per year (Danmarks statistik, 2014).

Dimitrakakis et al. (2009) found that the average share of misplaced WEEE and batteries in the German residual household waste constituted 1.27% and 0.04%, respectively. In the Netherlands, the misplaced WEEE in residual household waste constituted 0.44–0.88% (Huisman et al., 2012). Bernstad et al. (2011) found that the misplaced WEEE was 0.7%, before the introduction of source segregation and 0.4% after. The share of misplaced batteries in the residual waste before the introduction of source segregation was not accounted for but constituted 0.04% afterwards.

There seems to be relatively good agreement between the studies on the amounts of misplaced WEEE and batteries in the residual household waste. The four studies could, therefore, likely be considered representative of western European conditions. The amount of WEEE in the German residual household waste is relatively larger than the Dutch, Swedish and Danish findings. This is most likely due to Dimitrakakis et al. (2009) including a ''category 11'' which contained fractions that are not considered to be WEEE, according to the WEEE directive (CEC, 2012).

Table 7: Comparison of four western European studies of residual household waste that included misplaced WEEE and batteries

	Misplaced WEEE		Misplaced batteries	
	Amount [g hh^{-1} $week^{-1}$]	**Share of residual household waste [%]**	**Amount [g hh^{-1} $week^{-1}$]**	**Share of residual household waste [%]**
Denmark - Bigum et al. (II)	29	0.34	4	0.04
Germany - Dimitrakakis et al. (2009)	*(2.52)**	1.27	-	0.04
the Netherlands - Huisman et al. (2012)	*(1-2.1)**	0.44-0.88	-	-
Sweden - Bernstad et al. (2011)	18.4-35	0.4-0.7	2.1-7.0	0.04

*kg $inhabitant^{-1}year^{-1}$

hh: household

4.2.4 Development over time

This chapter includes a comparison of two larger sorting analyses of Danish residual household waste. The objective is to show how misplacement of special waste has developed from 2001 to 2010/2011.

Bigum et al. (II), Petersen (2011a), Petersen (2011b), Petersen (2012a) Petersen (2012b) Petersen et al. (2012a) and Petersen et al. (2012b) show the most recent larger sorting analysis of residual household waste. In 2001, a similar sorting analysis on Danish residual household waste was conducted (Petersen and Domela, 2003; Riber and Christensen, 2006). This included 2210 households (both single and multi-family).

In order to gain knowledge on whether or not misplacement of special waste with the residual waste has been increasing or decreasing in this ten-year period, these two studies (both considered to be representative of Danish conditions) were compared. Riber and Christensen (2006) and Petersen and Domela (2003) did not consider WEEE as a separate waste type but aggregated WEEE with the "other non-combustible" fraction. Based on Petersen and Domela (2003), and unpublished background data from Riber and Christensen (2006), it was nonetheless possible to determine the amounts of misplaced special waste in the Danish residual waste in 2001.

The content of the residual household waste is shown in Table 8 in terms of 9 major fractions including special waste.

Table 8: The 9 major material fractions in residual household waste, examples of included waste items, along with the amounts in the Danish residual household waste in 2001 (Petersen and Domela, 2003; Riber and Christensen, 2006) and 2010/11 (Bigum et al., II; Petersen, 2011a; Petersen, 2011b; Petersen, 2012a; Petersen, 2012b; Petersen et al., 2012a; Petersen et al., 2012b)

Material fractions	Included items	Amount [$kghh^{-1}week^{-1}$] (2001)	Amount [$kghh^{-1}week^{-1}$] (2010/11)
Organic	Food waste, garden waste	4.67	3.73
Paper and cardboard	Recyclable paper and cardboard, other paper and cardboard packaging (pizza boxes etc.)	1.84	1.32
Glass	Glass packaging and drinking glass	0.27	0.25
Plastic	Plastic packaging	0.86	0.51
Metal	Metal packaging and other metal	0.31	0.19
Hazardous	Containers with chemicals and paint, razor blades, spray cans, medicine, syringes, lighters, polyvinyl chloride (PVC)	$3.00 \cdot 10^{-3}$	0.06
Special waste	WEEE, batteries, ink cartridges and cables	0.03	0.04
Other combustibles	Textiles, office articles, cigarette butts, diapers and other combustible waste	1.05	2.25
Other non-combustibles	Construction and demolition waste, ceramics, soil, bricks, stones, drinking glass and filament bulbs	0.35	0.17
Total		*9.39*	*8.52*

hh: household

In 2001, the amount of discarded residual household waste was 9.39 kg per household per week, and in 2010/11, this had decreased by 9% to 8.52 kg per household per week. Most of the waste fractions in the residual household waste have changed significantly in the period from 2001 to 2010/11. The only exception is glass waste, which remains more or less the same (decreased by 6%). The discarded amounts of organic waste, paper and cardboard, plastic, metal and other non-combustibles all decreased with a magnitude of 20% to 52%. The waste fraction "other combustibles" increased by

114% from 2001 to 2010/11. Some of this could be due to differing sorting criteria used in 2001 and 2010/11 studies, respectively, as the “other combustibles” fraction is an aggregated waste type consisting of many different items. This is, however, not considered to affect the conclusion that the amount of other combustibles has increased since 2001.

The amount of hazardous waste has increased significantly since 2001, and the presence of this waste type was found to be 19 times that of 2001 in 2010/11. The amounts are, however, still quite low (0.06 g per household per week as opposed to 0.003 g per household per week in 2001) and might be easily influenced by varying definitions of what is considered to be hazardous waste by the municipalities.

The presence of special waste has increased by 37% from 2001 to 2010/11. In the case of hazardous waste, the actual amount is still relatively low (0.04 g per household per week in 2010/11). Of the individual special waste types, the amount of misplaced batteries has actually decreased by 64%, which is surprising, as Bigum et al. (II) found that batteries (to a large degree) are still being misplaced. WEEE and cables were grouped for the purpose of being able to compare the two studies, as it was not possible to distinguish between WEEE and cables in the Petersen and Domela (2003) study. The combined misplacement of WEEE and cables has increased by 112%.

Ink cartridges were, according to Petersen and Domela (2003), grouped with “lead products,” and it was not possible to determine the sole amount of ink cartridges in 2001. Nevertheless, it can be seen that there is an increase of 267% in this ten-year period. This means that the amount of misplaced ink cartridges might even be higher.

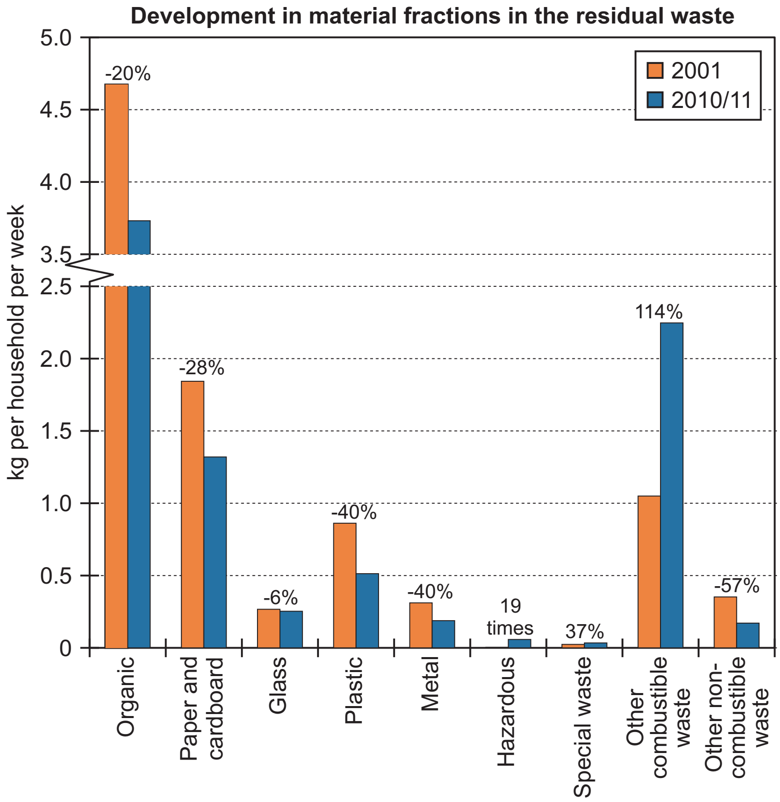


Figure 5: Development in the Danish residual household waste amount and content from 2001 to 2010/11. The changes are obtained by comparing Petersen and Domela (2003) and Riber and Christensen (2006) to Petersen (2011a), Petersen (2011b), Petersen (2012a), Petersen (2012b), Petersen et al. (2012a) and Petersen et al. (2012b).

4.3 Flow of misplaced special waste: Amount and significance

The European collection targets for WEEE and batteries are both related to the marketed amounts and thus aim at minimizing the differences between marketed and collected (CEC, 2006; CEC, 2012). Misplacement is an obvious alternative disposal route that could explain this difference between marketed and collected (hereafter referred to as “the gap”), and quantification of misplacement in relation to the gap is, therefore, relevant.

The gap between marketed and collected batteries was, in 2011, 2,193 tons (DPA, 2012). The amount of misplaced batteries was determined in Bigum et al. (2011) and corresponds to 538 tons in the same reference year. Misplacement would be able to account for 25% of the observed gap in 2011. Therefore, despite the findings of Bigum et al. (II) that the misplacement of batteries is significant, their research does not seem to be able to account for the entire gap. This suggests that alternative waste disposal routes for batteries exist.

The gap between marketed and collected WEEE was, in 2011, 31,946 tons (DPA, 2012). The amount of misplaced WEEE was determined in Bigum et al. (2011) and corresponds to 3,848 tons in the same reference year. Misplacement would be able to account for 12% of the observed gap in 2011 but depends strongly on equipment types. The gaps seen, according to the individual WEEE directive categories, will be analysed in the following discussion.

When analysing the flows and gaps of special waste, the heterogeneity and difficulties with estimating net flows should be kept in mind. The gap does not necessarily equal waste that is circumventing the collection systems. However, a consistent yearly gap between items marketed and collected strongly suggests unaccounted for flows. Also, other flows, such as misplacement with other fractions at the recycling stations, illegal shipments, dumping and uncollected waste (e.g., stock piling) still remain to be quantified and accounted for. The following should, therefore, be taken tentatively.

Table 9 shows that the gaps are most significant for WEEE directive category 1 (large household appliances) followed by WEEE directive category 2 (small household appliances), WEEE directive category 6 (electric and electronic tools), and WEEE directive category 7 (toys, leisure and sports equipment).

Table 9: Amount (in tons) of marketed, collected and misplaced household WEEE in Denmark, 2011. The marketed and collected amounts are based on DPA (2012) and the misplaced amounts on Bigum et al. (II)

Denmark 2011	Marketed [tons]	Collected [tons]	Gap [tons]	Misplaced [tons]
1. Large household appliances	59,298	36,420	22,878	0
2. Small household appliances	13,305	4,464	8,841	1,386
3. IT and telecommunications equipment	15,621	14,132	1,489	709
4. Consumer equipment and photovoltaic panels	15,465	25,937	-10,472	1134
5a. Luminaries	114	1	113	0
5b. Lamps	1,424	658	766	131
6. Electrical and electronic tools	6,290	1,033	5,257	39
7. Toys, leisure and sports equipment	3,606	383	3,223	408
8. Medical devices	148	50	98	0
9. Monitoring and control instruments	235	482	-247	41
10. Automatic dispensers	0	0	0	0
Total	115,506	83,560	31,946	3,848

Some of the individual gaps are negative, basically meaning that the collected amounts are higher than the marketed. This is the case for consumer equipment and photovoltaic panels, as well as monitoring and control instruments. A consistently negative gap (as is the case for monitoring and control instruments) could suggest inconsistencies in the registration of marketed amounts, and that, perhaps, these equipment types are registered under another category. It is also very likely that a negative gap is due to products weighing less than similar products marketed just a few years ago. This, in particular, could be suspected for IT and telecommunication equipment (which in 2007 and 2010 had a negative gap, and relative low gaps in 2008, 2009, 2011 and 2012), as well as consumer equipment and photovoltaic panels, which have had a consistent negative gap since 2009 (DPA, 2009; DPA, 2010; DPA, 2011a; DPA, 2012, DPA, 2013). Other reasons for negative values could be if large amounts of stored or accumulated items are entering the recycling system.

Positive values could suggest that WEEE is not being collected. Alternatively, this suggests that products being placed on the market are heavier than previously or that the market for some items is not saturated (e.g., by new

EEE products replacing non-EEE products). Finally, it could be explained by WEEE being registered as belonging to other WEEE categories.

Large household appliances are, by far, the largest contributor to the gap, with an average of approximately 22,878 tons in 2011. It is possible to assume that the market for large household appliances is relatively saturated in a developed country such as Denmark. Though some of the gap could be explained by marketed large household appliances being bigger and heavier than previously, the very large and consistent discrepancy suggests that a sizeable amount is circumventing the legal collection system. No large household appliances were found in the Danish residual household waste (Bigum et al., II), and misplacement is, therefore, unlikely to explain the gap of large household appliances.

The amount of misplaced small household WEEE only constitutes 16% of the gap, which suggests that misplacement is not the primary reason for the missing amounts of this waste type. Other explanations for the gap could be due to multi-functionality, an unsaturated market, accumulated WEEE in households or other disposal routes, like secondary green shipment as products to other countries, dumping, illegal shipments, etc.

The amount of misplaced toys, leisure and sports equipment constitutes 13% of the missing gap, suggesting that misplacement is an unaccounted flow, but that it is not the primary reason for the missing amounts.

The presence of misplaced electric and electronic tools with the residual household is insignificant, and the misplacement can only account for 1% of the unaccounted flow. So, although the gap between marketed and collected electric and electronic tools is significant, misplacement with the residual household waste cannot explain this. The missing amounts of electric and electronic tools can possibly be explained with an unsaturated market and increased consumption of these product types.

A gap is also seen for IT and telecommunications equipment. Misplacement would be able to account for 48% of this, suggesting misplacement of IT and telecommunications equipment within the residual household waste to be significant.

There is also a gap between the marketed and collected amount of lamps. Unlike the other WEEE directive categories, the weight of these items are rather uniform in size (Bigum et al., II). It would, therefore, be likely to assume that the gap is less influenced by varying item size, and that it is more reasonable that some of the gap is influenced by other factors. Some of the gap could be attributed to a change in the guidelines for reporting marketed amounts of lamps (DPA, 2012), but this is not considered to be the primary reason. The main factor is most likely an unsaturated market for lamps, which can be considered to be the case, since the phase out (2009) and final ban of the conventional filament light bulbs in 2012 (CEC, 2005b). An unsaturated market for lamps can be seen from the increasing marketed amounts, which almost tripled from 2009 to 2010, while the collected amounts remained almost the same (DPA, 2010; DPA, 2011a). Filament light bulbs are not categorised as EEE, which means that EEE products are replacing non-EEE products. The long expected lifetime of lamps, can be expected to influence the gap for this specific product category, as the majority will most likely not become obsolete for the next 10 years.

Misplaced WEEE, with the residual household waste, might be a significant unaccounted flow for small household appliances, as well as IT and telecommunications equipment, lamps, toys, leisure and sport equipment. Significant amounts of consumer equipment were also found in the residual waste (Bigum et al., II). However, since the gap is negative for this category, it was not possible to quantify this misplacement in relation to the gap.

4.4 Improving collection

The previous chapter analysed to what extent misplacement is able to account for the gap between marketed and collected material. This chapter aims at analysing citizen behaviour when disposing of special waste. The purpose of this section is to make recommendations with the purpose of minimizing misplacement.

The amount of misplaced WEEE and batteries was found to constitute 16% and 39% of what is collected through the dedicated special waste collection schemes (Bigum et al., II). This suggests that people are, overall, using the dedicated special waste collection schemes for WEEE, but that the same cannot be said for batteries.

WEEE

An assessment of citizen behaviour towards disposing WEEE can be accomplished by roughly assuming that people either use the dedicated waste collection schemes or misplace WEEE in their household waste bins. The assumption of using the dedicated disposal routes or misplacement with residual household waste allows for an insight into what people prefer to do with their WEEE. The assumption does not take into account that people might use other disposal routes, such as dumping and misplacement with other fractions at recycling stations.

Table 10 shows the relationship of collected amounts of WEEE to the misplaced amounts of WEEE given in percentages. Table 10 shows what categories of WEEE people are most prone to dispose of using the dedicated waste collection schemes, and which materials tend to find their way to the waste bin. The table is weight based and does not take into account the number of misplaced items, which might be a better estimation for "frequency" (Bigum et al. II). This is because no data on the number of discarded items via the dedicated waste collection schemes exist.

Table 10 shows that people are generally using the dedicated waste collection schemes for IT and telecommunications equipment. This seems to contradict the analysis in the previous chapter that showed that misplacement constituted almost half of the gap. However, the reason is simply that the gap for this category is small, and that these "missing" small amounts, to a large degree, are misplaced. To improve the collection of IT and telecommunications equipment, as well as consumer equipment (shown to have a negative gap, but nonetheless to be frequently misplaced), certain equipment types should be targeted. Bigum et al. (II) found that within these two categories, cables and headphones are frequently misplaced and, hence, should be addressed and used as examples in information material and sorting guidelines.

Table 10 also shows that when it comes to disposing of small household appliances, toys, leisure, sports equipment, and lamps, people have a tendency to use their private waste bins. This tendency seems distinct when it comes to toys, leisure and sports equipment, where the amount that was misplaced exceeded what was collected through the dedicated schemes (51% to 49%) in 2011. The significant misplacement of toys, leisure and sports equipment should be addressed by both increasing the recognition of this category as being WEEE and by targeting certain item types. Bigum et al. (II) showed that toys, as well as flashlights and bicycle lights, are frequently misplaced. In order to improve collection and reduce misplacement, these items should be addressed specifically.

Small household appliances, except for a few reoccurring items, appear mixed. This suggests that people might have difficulties identifying or recognizing these items as being WEEE, and thus might unintentionally be misplacing them. Increasing the collection of these waste types could possibly be obtained by proper labelling the EEE or with an overall focus on informing the public that this WEEE category requires separate collection.

Similar to issues with small household WEEE, the misplacement of lamps could be due to lack of knowledge that these require separate collection. This could be because it is allowed to dispose of the conventional filament light bulbs with the residual household waste, and people might, therefore, now be aware of lamps being WEEE, requiring specialised treatment. Special attention should, therefore, be on informing people that this waste type is, in fact, WEEE and should be collected separately.

Table 10: Collected household WEEE via the dedicated collections schemes as opposed to misplaced with the residual household waste, given in percentages [%]. The calculations are based on DPA (2012) and Bigum et al. (II).

Denmark (2011)	Collected from households [%]	Misplaced [%]
1. Large household appliances	100	0
2. Small household appliances	79	21
3. IT and telecommunications equipment	95	5
4. Consumer equipment and photovoltaic panels	98	2
5a. Luminaries	100	0
5b. Lamps	84	16
6. Electrical and electronic tools	96	4
7. Toys, leisure and sports equipment	49	51
8. Medical devices	100	0
9. Monitoring and control instruments	92	8
10. Automatic dispensers	0	0

Batteries

Bigum et al. (II) found that 20% of discarded batteries were misplaced as built-in WEEE. This is quite a high share of built-in batteries, and one way to reduce the amount of misplaced batteries would, therefore, be by increasing the collection of the WEEE that tends to be misplaced.

Portable batteries are a fairly recognizable waste type, and separate collection of these has been advocated since 1999 in Denmark (MST, 1999). It is, therefore, unlikely that misplacement of batteries is due to citizens lacking information that this waste type requires separate collection.

This statement is supported by a survey conducted by the Danish Environmental Protection Agency (EPA), among 1126 Danish citizens, where 90% of the responders claimed that they were already using the dedicated special waste collection schemes for their used batteries (Petersen et al., 2012). This is in contrast to the findings in Bigum et al. (II) that showed that the misplacement of batteries is significant. It is unlikely that the remaining 10% of the responders are solely responsible for the significant misplacement of batteries. The Danish EPA survey points to an existing high level of information on the proper disposal of batteries, but according to the findings in Bigum et al. (II), this seems not to have an effect.

Fees and other measurements

Increasing information about the dedicated waste collection systems for obsolete batteries seems not to be sufficient. Improving collection of batteries could, therefore, very well be supplemented with a fee.

In the case of WEEE, a fee alone might not necessarily be enough but would have to be supplemented by better sorting information as discussed previously. Simply adding a fee on WEEE would not necessarily raise awareness or provide the necessary recognition of certain items as being WEEE, requiring separate collection.

The possibility of introducing a fee on WEEE and batteries was recently suggested by the Danish Economic Council as one of the means to ensure a higher collection rate of especially small WEEE items and batteries (DEC, 2013). Introduction of a fee and a return and deposit system for batteries was seriously considered in 2007 but was never realized (DEC, 2013). The Danish Economic Council recommends that a return and deposit system for smaller WEEE items and batteries be reconsidered (DEC, 2013). An alternative to this could simply be to assess the efficiency of existing dedicated collection schemes and determine how special waste types are most effectively collected. Bigum et al. (II) could not prove that the observed variations between municipalities could be attributed to different collection schemes. However, Bernstad et al. (2011) clearly showed that the introduction of source segregation is effective. Implementation of efficient collection schemes in all municipalities, supplemented with increased information and sorting guides, could possibly be a sufficient alternative to a return and deposit system.

5 Environmental aspects of special waste

The environmental aspects associated with the special waste types are related to the hazardousness of the waste types and the environmental risks, benefits or savings of these during management. Material recovery from special waste is not only a resource issue but also an environmental issue and will be presented and discussed separately with a specific focus on the metal resources. The discussion of the environmental aspects of the management of special waste are, in this thesis, related to Denmark and Europe and will not address unmanaged special waste or management under unsafe conditions elsewhere (e.g., in developing countries).

5.1 Hazards and risk

Special waste is known to contain many hazardous and problematic compounds, such as heavy metals, flame retardants, and xenobiotic organic compounds. CRTs, printed circuit boards, gas discharge lamps and plastics are the components of WEEE that contain the majority of the hazardous compounds and substances (Tsydenova and Bengtsson, 2011). Some of the commonly found hazardous compounds and substances in WEEE are: lead (Pb), barium (Ba), beryllium (Be), cadmium (Cd), mercury (Hg), antimony (Sb) asbestos, chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs), hydroflourocarbons (HFCs), hydrocarbons (HCs), polychlorinated biphenyls (PCBs), brominated flame retardants (BFRs), and Polyvinyl Chloride (PVC) (DEFRA, 2006; Tsydenova and Bengtsson, 2011). The primary hazardous compounds in batteries are Cd, Pb and Hg, and, in ink cartridges, Cd is of concern (Tsydenova and Bengtsson, 2011). The protective jacket of cables is very likely to consist of PVC, including various phthalates and stabilizers (Cd, Pb and Sn) (Reisinger et al., 2011). The above mentioned elements of concern are just a few examples, as hazardous and toxic substances in special waste are numerous (AEA Technology, 2004; Gross et al., 2008; Onwughara et al., 2010; Robinson, 2009). The significance of CFC as being problematic and of environmental concern is seen by an environmental assessment of the WEEE directive, which found that the most important WEEE items to manage were the CFC-containing cooling appliances.

The Montreal protocol (UNEP, 2000) banned the use of CFCs, and the Restriction of Hazardous Substances (RoHS) directive (CEC, 2002) has banned other problematic substances compounds in EEE. However, this does not mean that WEEE is now free of hazards and risks. Banned substances might still appear in the waste stream as historic waste, and other compounds that have yet to be banned are of concern, as they remain and continue to be used in production. If not properly managed, there is a risk that these substances will pollute the environment and pose a risk to the workers working in the waste management and recycling sector. For instance, landfilled special waste could constitute a risk of leaching and evaporation of volatile substances to the neighbouring environment.

Oguchi et al. (2013) conducted an analysis of various metals contained in CRT glass and printed circuit boards, including toxic metals (Ba, Be, Cd, chromium (Cr), Pb and Sb). Oguchi et al. (2012) found that CRT contained the highest concentrations and amounts of Ba, Pb and Sb and suggested that priority to the waste management of CRTs should be given. However, Oguchi et al. (2013) also stated that not only the concentration but also the total amount of metals should be considered when categorising WEEE as a source of toxic metals.

The environmental benefits of recycling WEEE and batteries are related to the safe treatment of the hazardous compounds and the avoidance of them ending up in the environment. The primary hazards of mechanical pre-treatment seem to be associated with the separation, size reduction, and shredding, as this is the stage where the emission of hazardous compounds is more likely to occur (Tsydenova and Bengtsson, 2011). During size reduction, there is a risk of accidental release of hazardous compounds as well as dust, which can carry hazardous substances that may pose a risk to the workers as well as the environment. Tsydenova and Bengtsson (2011) compiled the few available studies that had quantified the risk towards humans and the environment during mechanical pre-treatment of special wastes. Tsydenova and Bengtsson (2011) found that the workers and neighbouring environment were not only severely affected by the management of special waste, but also that changing the recycling processes could reduce the occupational exposure.

Metal recovery by the pyrometallurgical processes was also found to constitute risks to the environment. Many of the elements present in the special waste could be detected in the environment surrounding the facilities (Tsydenova and Bengtsson, 2010). This means that these elements of concern are being emitted during processing. Pyrometallurgical processing also poses a risk of producing dioxin and furans (PCDD/Fs), due to the content of chlorine (Cl) and Br in the special waste stream. PCDD/Fs can be removed, if adequate flue gas cleaning equipment is installed (Tsydenova and Bengtsson, 2010). This is the case for most of the integrated metal smelters, but standard copper smelters, designed for the treatment of mining concentrates or simple copper scrap, are usually not equipped with this (Hagelücken et al., 2006).

Hischier et al. (2005) and, later, Wäger et al. (2011) (a follow-up of the 2005 study) conducted an assessment of the environmental impacts of the Swiss collection and recovery systems for WEEE. The study calculated the overall environmental impacts of the collection, pre-processing and end-processing of the Swiss WEEE using the ecoinvent life cycle inventory database (ecoinvent v2.01). The study compared the Swiss recovery system to the same amount of WEEE being incinerated or landfilled. The functional unit of the study was the total amount of recovered resources from 1 tonne of WEEE plus the energy produced by incineration of the WEEE. The study included all WEEE treatment categories but only included the WEEE material fractions available in the ecoinvent data base (batteries, CRT devises, cables and printed circuit boards). In contrast to Bigum et al. (I), Wäger et al. (2011) also included the management of the plastic fraction based on company data that the authors had obtained. The results showed that the environmental impacts from the recovery scenario were significantly lower than incineration and landfilling. The burdens of incineration and landfilling were dominated by the necessary primary production of the materials contained in WEEE, which, in the incineration and landfilling scenario, was not recovered. This is in accordance with the findings of Bigum et al. (I), and shows that the avoided burdens of primary production when recovering (especially metals) are an important factor. The greatest environmental savings obtained in Wäger et al. (2011) was found to be associated with the recovery of batteries, metals (aluminium (Al), copper (Cu), gold (Au), palladium (Pd), silver (Ag), and steel), cables, and printed circuit board treatment. The recycling of plastics was shown to have a clearly lower environmental impact. Looking only at the recovery system and disregarding the primary production, the environmental impacts from metals treatment and the treatment of CRT devices and plastics was the main contributor. The collection and pre-processing contributed only marginally.

5.1.1 Incineration

Astrup et al. (2011) found that the combustion of batteries could lead to an increase in the presence of Hg and Cd in the flue gas, and Cd, Hg, nickel (Ni) and sulphur (S) in the solid residues. The potential risk to the environment by incineration of special waste was supported by Molto et al. (2011), who performed similar tests on the combustion of mobile phones and found that this could lead to an increased level of dioxins and furans (PCDD/Fs). Tsydenova and Bengtsson (2011), to some extent, contradict this and state that the environmental impacts from the incineration of WEEE are less clear if the incineration is followed by proper flue gas treatment.

The special waste types, and in particular WEEE are quite complex products that are typically not easily separated and the metals not easily liberated. In addition to this, the content of hazardous additives, such as flame retardants, makes a waste stream quite complex and not easily managed. With this in focus, Vehlow et al. (2000) stated that the incineration of WEEE plastic might even be a viable treatment method, although still secondary to recycling.

5.2 Resources in special waste

Several sources have documented the presence of valuable metals in WEEE (Morf et al., 2007; Chancerel et al., 2008; Chancerel et al., 2009, Salhofer and Tesar, 2011; Oguchi et al., 2013), and it is said that up to 60 elements from the periodic table can be found in complex electronics (Goodship and Stevels, 2012). In addition, special waste types are often stated as a potential urban mine, because the concentrations of metals in special waste in many cases are higher than in the ores mined for primary production (Hagelüken, 2006; Betts, 2008).

Morf et al. (2007) conducted an analysis of sWEEE and compared the substance flow of a range of elements to the flow of the same elements contained in municipal solid waste. The results showed that the flow of Ni, Cr, Cu and Br in sWEEE was 5.6-1.2 times higher than in the municipal solid waste, even though the size of the flow of sWEEE was much smaller.

Oguchi et al. (2013) concluded that not only the metal concentration, but also the total amount of metals contained in WEEE, is important when characterising WEEE as a valuable secondary metal resource. This means that recovering metals contained in mid-sized communications equipment such as video tape recorders, personal computers (PC's), and printers should be given priority, as the combination of flow size as well as concentration makes them rich in valuable metals. Additionally, smaller items like mobile phones, telephones, videogames, portable audio players and digital cameras were characterised as important targets for precious and toxic metals.

Resources in misplaced special waste

Bigum et al. (III) included an estimation of the chemical composition of the individual residual household waste types based on available literature data.

From this data, it could be seen that even though special waste only constitutes 0.5% (w/w) of the residual household waste, it contributes to 15.2-52.3% of the total amount of Cd, Cu, Hg, manganese (Mn), Ni, Pb and zinc (Zn) present in the residual household waste (see Table 11). This makes misplaced special waste the most significant fraction with respect to metal content when excluding Fe and Al, and it highlights that misplaced special waste is a significant source for the loss of valuable resources, as well as a source of toxic metals. For the rare and precious metals (Au, Ag, Pt, etc.) special waste was, in Bigum et al. (III), considered the dominant fraction, but lack of data on the content of these metals in other waste types limits the quantification of this.

Table 11: The distribution of selected metals in special waste types was found to be present in residual household waste. Only the metals where special waste contributes disproportionately to its weight share (0.5 w/w %) are shown. Blank cells mean that no data on the content of the element in the special waste type was available.

Elemental distribution (%)	Cd	Cu	Hg	Mn	Ni	Pb	Zn
sWEEE	1.5	6.7	4.5		4.0	1.8	0.3
Lighting equipment		14.5	11.8			4.0	
CRT		1.2			0.1	9.4	0.1
Batteries	50.8	0.8	21.4	40.7	29.6	$4.9 \cdot 10^{-2}$	15.8
Ink cartridges	$4.3 \cdot 10^{-4}$		$3.0 \cdot 10^{-2}$	$3.3 \cdot 10^{-3}$	$2.1 \cdot 10^{-3}$	$1.6 \cdot 10^{-4}$	
Cables		20.2					
Special waste type, total	*52.3*	*43.4*	*37.7*	*40.7*	*33.7*	*15.2*	*16.2*
Residual waste, remaining	*47.7*	*56.6*	*62.3*	*59.3*	*66.3*	*84.8*	*83.8*

6 LCA of special waste

The following two sub-chapters presents an overview and the main findings and conclusions from the LCAs of special waste conducted in this PhD project.

6.1 Incineration of misplaced special waste types

Using an LCA approach, Bigum et al. (III) assessed the environmental impacts of the incineration of residual household waste containing misplaced special waste. The objective of Bigum et al. (III) was to assess the environmental consequences of incineration of residual household waste and to evaluate the contribution of misplaced special waste to the overall environmental impact. A secondary objective was to show how assessing the resource depletion of the elements, not being recovered, could be included in an LCA.

Approach and method

The functional unit in the study was 1 tonne of wet residual household waste (recyclables had been collected separately). The composition of the waste is shown in Table 6. The system boundaries were from “gate to grave,” meaning that it included the processes after the residual waste was received at the incineration facility as well as the fate of the elements and solid residues after incineration. Recovery of electricity and heat from the combustion process, as well as the recovery of metals from the ashes, were included. The LCA was performed in accordance with the recommendations by the ILCD (JRC, 2011). The results were normalized and presented in PE, and according to the twelve impact categories:

Global warming (GW), stratospheric ozone depletion (OD), photochemical oxidant formation (POF), terrestrial acidification (AC_{soil}), eutrophication potential (EP_{soil}), freshwater eutrophication (EP_{fw}), marine water eutrophication (EP_{mw}), human toxicity, carcinogenic ($HT_{carcinogenic}$), human toxicity, non-carcinogenic ($HT_{non\text{-}carcinogenic}$), ecotoxicity (ET), depletion of abiotic resources (fossil) (RD_{fossil}), and depletion of abiotic resource (elements) ($RD_{abiotic}$).

The ILCD recommendations include the CML methodology (Oers et al., 2002), which aggregates the resources of fossil origin (e.g., coal and oil) and elemental or abiotic resources [e.g., iron (Fe), Cu, and platinum (Pt)] into one impact category. In Bigum et al. (III), it was decided to keep the depletion of fossil resources and abiotic resources separate, as a specific focus of the study was on the recovered/lost metal resources.

The study used a generic Danish incinerator, which was modelled as a combined heat and power plant, with a respective energy efficiency of 73% and 22%, based on the lower-heating value and accounting, separately, for the energy use of the plant. The modelling framework was consequential, and as the system was to represent Danish conditions, the substituted marginal energy was coal (Energistyrelsen, 2011). The incinerator was modelled as equipped with wet flue gas cleaning and with ferrous (Fe) metal and aluminium (Al) recovery from the bottom ashes. Fifty percent (50%) Al and 80% Fe was assumed to be recovered following the recommendations by Jacobsen et al. (2013). The bottom ash was used in road construction, and the fly ashes and APC residues were landfilled. Elements remaining in the bottom ash, fly ash and APC residues were considered to be lost and, hence, contributed to resource depletion.

The modelled scenario included four processes: The incinerator, Al recovery, Fe recovery, and the unrecovered resources contained in the depositing of residues from the incinerator (bottom ash deposit) (see Figure 6). The system included the recovery of electricity and heat from the combustion of the residual household waste, the recovery of Al and Fe, and the depletion of the unrecovered elements in the ashes.

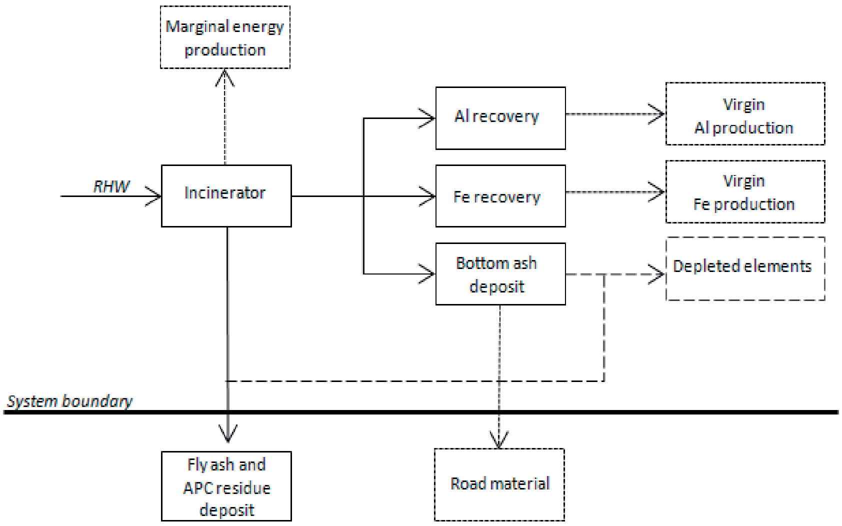


Figure 6: The modelled scenario. The dotted boxes and lines show the substituted processes and the dashed flow and box the resource depletion of the unrecovered elements (Bigum et al., III)

Results

Figure 6 shows the environmental impacts of the incineration of 1 tonne of residual household waste containing misplaced special waste. The incineration of 1 tonne of residual household waste resulted in net savings in all impact categories except for ET and $RD_{abiotic}$. Environmental savings are due to the energy, Fe and Al recovery and the substituted primary production of these. This supports the importance of material recovery.

Special waste misplaced with the residual household waste contributed very little to the savings in GW (<1% related to plastic in the special waste) but is the dominant contributor to the load of $RD_{abiotic}$ where it accounts for 96% of the burdens. This is mainly due to losses of Pt (85%), and, to a smaller degree, Cu (7%), Au, and Ag (data not shown). For the toxic categories, special waste contributes about 28% to ET, 4% to $HT_{carcinogenic}$ and about 25% to the $HT_{non\text{-}carcinogenic}$ impact categories, which is significant, as special waste constituted only 0.5 w/w % of the residual waste.

The burdens from special waste on the ET category can be attributed to the content of Cu, Ni and Sn. Lamps and cables were shown to have a high impact due to the Cu content. Batteries also contain Cu, but Ni is the main element responsible for their impact on ET. CRT and sWEEE are also seen to be a burden on ET, which is primarily due to the Cu content and partly to tin (Sn) (CRT) and Ni (sWEEE).

The burdens in the $HT_{carcinogenic}$ and $HT_{non\text{-}carcinogenic}$ impact categories are highly related to the presence of Hg and Cr in special waste. Lamps, sWEEE and batteries result in relatively high burdens on these categories (in the order of 10^{-4} to 10^{-6} PE/kg) due to their content of Hg. CRT does not contain Hg, according to the literature survey, and the burdens to these toxic categories were related to its content of Pb. The incineration of ink cartridges and cables resulted in small savings due to the low estimated content of Hg and Pb.

Generally speaking, all 6 special waste types (sWEEE, lamps, CRT, batteries, ink cartridges and cables) were found to exhibit significant potential impacts, although in different categories. The actual contribution, however, depends on how much special waste is actually misplaced with the household waste.

The results in Bigum et al. (III) are highly dependent on the literature based estimation of the composition of the special waste types and the elemental partitioning in the incineration process (transfer coefficients). For the latter, the lack of air emission data for the more rare metals in the incineration model was of particular concern.

These two aspects were assessed by changing the input data in 3 sensitivity scenarios - one using the minimum reported values for the content of the special waste, the second using the maximum reported values, and finally the third, where it was assumed that 0.001% of the metals, which had no transfer coefficient to air in the base scenario, was emitted to the air. The metals included in the sensitivity analysis were: Ag, Au, bismuth (Bi), gallium (Ga), indium (In), Pd, Pt, strontium (Sr), tantalum (Ta), titanium (Ti) and zirconium (Zr).

The results showed that the impact categories GW, OD, POF, AC_{soil}, EP_{soil}, EP_{fw}, EP_{mw} and RD_{fossil} are mainly unaffected by the compositional variation, and that the toxicity categories ($HT_{carcinogenic}$, $HT_{non\text{-}carcinogenic}$ and ET) and the $RD_{abiotic}$ are highly affected (Bigum et al., II). The effect on $RD_{abiotic}$ is primarily due to the large variation in reported values for Ag in sWEEE and Pt in lamps, but also the content of Cd in batteries is an issue. The effect on $HT_{carcinogenic}$ and $HT_{non\text{-}carcinogenic}$ is due to increased emissions of Hg, where the impact is 100 times higher than from any other metal, due to high concentrations in lamps and batteries. The impact on ET is primarily due to the release of Cu from lamps and cables.

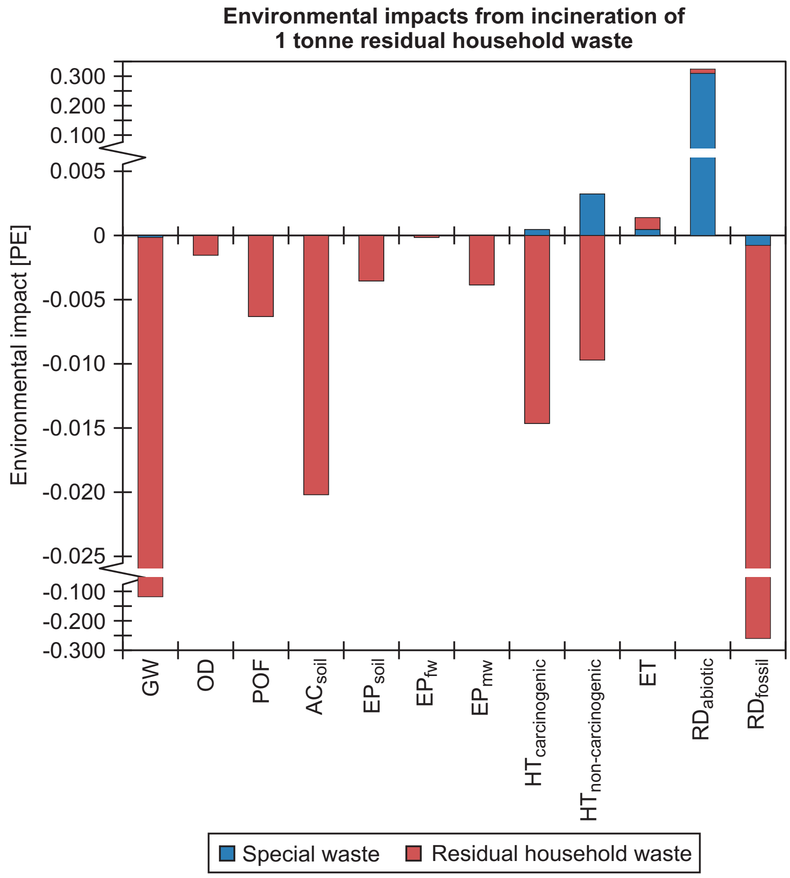


Figure 7: Environmental impacts from the incineration of 1 tonne of residual household waste (RHW) with energy, ferrous metal (Fe), and aluminium (Al) recovery, given in person equivalents (PE). The impacts from special waste are shown as a separate category.

Conclusion

The assessment of the importance of the misplaced residual waste was based on an extensive review of data available regarding the chemical composition of special waste types as well as the elemental partitioning in the incineration process. The data on the chemical composition of special waste found in residual household waste is still scarce and incomplete. Developing better and more complete datasets should be of priority in the future because the environmental impacts and the resource depletion observed is directly related to chemical composition of the special waste fractions. In particular, the content of rare, precious and hazardous metals should receive focus. However, despite the data being scarce and incomplete, the results in Bigum et al. (III) suggest that the presence of misplaced special waste in residual household waste is a significant burden on the environment, and that especially the toxicity and $RD_{abiotic}$ impact categories are affected.

There is a political focus on resource recovery from waste. Resource scarcity is an issue, and it has become necessary to look more critically on the current waste management choices and on which resources are being targeted for recovery. LCA is increasingly being used as a decision-making tool, and LCA should, therefore, be able to address resource recovery and to show the environmental burdens or consequences of not recovering resources. In Bigum et al. (III), this was addressed as a secondary objective. Resource depletion of the unrecovered elements was included by extending the system boundary beyond the traditional “zero burden approach.” This approach showed unrecovered abiotic resources from the incineration of residues to be an important impact category and one to which special waste contributed significantly. The recovery of Fe and Al contributed relatively little to this impact category and enhancing their further recovery from the ashes would not affect the loss of abiotic resources significantly. Only by recovering or minimizing the input of elements, such as Ag, Au, Cu and Pt, would it be possible to reduce the loss of abiotic resources from the system. These elements are primarily found in the misplaced special waste (sWEEE, lamps, CRT, batteries and cables) and further support the concept that continuing to address the misplacement of the special waste types with the residual waste and improving the overall collection of these wastes is necessary.

6.2 Metal recovery from high-grade sWEEE

Using an LCA approach, Bigum et al. (I) assessed the environmental impacts of recovering metals from high-grade sWEEE (WEEE directive categories 3 and 4). The objective of Bigum et al. (I) was to evaluate the environmental burdens of the recovery of Al, Ag, Au, Cu, Fe, nickel (Ni) and Pd, including the avoidance of the environmental impacts associated with the primary production of these same metals. A secondary aim was to establish LCIs for the recycling and recovery of these metals from sWEEE.

Approach and method

The functional unit of the study was: Recovery of Al, Ag, Au, Cu, Fe, Ni and Pd from 1 tonne of high-grade sWEEE. The system boundaries were from “gate to grave” and included the waste when it entered the recycling system and until the investigated metals had been recovered. For schematic presentation of the pre-treatment process, please see Bigum et al. (I). The hazardous compounds that require special treatment, as stated by the WEEE directive [Annex VII (CEC, 2012)], and the plastic waste fraction were not included in the assessment. The goal situation can be described as C1 (JRC, 2010) and the modelling framework as attributional. Allocation was used to attribute the burdens from the secondary and primary production to the individual metals.

Two common technical aspects for allocation were possible: Mass and economic value. The study allocated according to both issues. The study used the EDIP method (Wenzel et al., 1997) for quantifying the environmental impacts and resource consumption, as EASEWASTE was used as the modelling tool, and EDIP was the preferred methodology at the time. The results were normalized and presented in PE, and according to the following ten impact categories: Acidification, ecotoxicity in soil, ecotoxicity in water (chronic), global warming (100 years), human toxicity via air, human toxicity via soil, human toxicity via water, nutrient enrichment, photochemical ozone formation and stratospheric ozone depletion

Pre-treatment facilities in Europe are numerous and can vary in terms of technology. Obtainable recovery rates would depend on the facility, as well as the input to the facility. The recovery rates for the pre-treatment facility as shown in Table 12 can, therefore, not be considered to be a generic, as it is based on a specific process and a specific input. Obtaining other recovery rates might very well be possible (Chancerel et al., 2010; UNEP, 2011). In this study, the highest recovery rates are achieved for Fe (96 %), followed by Ni, Al, and Cu (57-90%). The modelled recovery rates for the precious metals, Ag, Au and Pd, are quite low, ranging from 12-25%. The main losses occurred during pre-treatment. The first step of pre-treatment is manual depollution, where hazardous compounds are removed as well as larger metallic parts, followed by shredding of the remaining waste. Shredding leads to the dispersion of metals to other fractions (especially to the plastic and ferrous fraction), where they are subsequently not recovered (Chancerel et al., 2008).

Table 12: The metal content of high-grade sWEEE (WEEE directive categories 3 and 4) and the recovery rates. [a]Chancerel et al. (2008), [b]Legarth et al. (2001), [c]Chancerel (2008), [d]Huisman (2003), [e]EASEWASTE (2010) (from Bigum et al. (I))

	Metal content high-grade sWEEE		Recovery rates %		
	Content	Unit	Pre-treatment[a]	Recovery process	Overall
Aluminium (Al)	33[c]	kg/tonne	86[c]	79[e]	68
Copper (Cu)	44[a]	kg/tonne	60	95[d]	57
Gold (Au)	22[a]	g/tonne	26	98[d]	25
Iron (Fe)	402[a]	kg/tonne	96	100[e]	96
Nickel (Ni)	3[b]	kg/tonne	100	90[d]	90
Palladium (Pd)	7[a]	g/tonne	26	98[d]	25
Silver (Ag)	313[a]	g/tonne	12	97[d]	12

The metallurgical recovery process of Ag, Au, Cu, Ni and Pd was modelled as based on the Swedish Rönnskär facility. The LCI data used was established based on the public green reports and a good correspondence with the company along with published ecoinvent data (Classen et al., 2007).

The Rönnskär facility is a multiple input–output system, as it extracts metals from both primary and secondary sources. An LCA on a multiple input–output system requires allocation of emissions and resource consumption as system expansion is not an option.

From a waste management perspective, a common technical aspect used for the allocation would be according to the incoming mass, as the "function" is to treat the amount of received waste. For the production of secondary and primary metals allocation, this could be based on the economic value of the outputs as the purpose of the system would be the recovery of resources, which is correlated to their economic value. Both allocation approaches are equally valid in this study, and both were, therefore, applied.

Mass allocation, as well as economic allocation, requires knowledge of the input and output flows to and from the different processes in the facility. These data are not always publically available and, in this case, required estimation based on available data on material consumption and waste generation. Additionally, economic allocation requires knowing the economic value of the metals. Metal prices from January 2006 were used for the economic allocation. However, prices on metals fluctuate. This can have an effect on the allocation and, hence, also the results of an environmental assessment. This is considered to be particularly important if an assessment wishes to determine the burden of the recovery of individual metals, e.g., kg CO_2 per kg Au. However, in this assessment, this was not done, and allocation was simply used to assess the bulk recovery of metals as opposed to primary production. The LCI data as presented in Bigum et al. (I) are allocated according to incoming mass.

Results

Table 13 shows the environmental impacts of recovering Al, Ag, Au, Cu, Fe, Ni and Pd from 1 tonne of high-grade sWEEE. The results are given according to individual impact categories and the type of allocation (mass or economic) which was used. The assessment showed significant environmental savings compared to primary mining and refining of the same metals. Only the environmental impact of the recovery of metals from high-grade sWEEE was evaluated, and the removal and subsequent treatment of hazardous components and plastic was excluded. The study does, therefore, not represent the overall environmental cost of treating 1 tonne of high-grade sWEEE but only the environmental benefits from the recovery and recycling of these specific metals.

Bigum et al. (I) also showed savings in resource consumption, when recovering high-grade sWEEE (Table 13). The highest savings in resource consumption was found for the precious metals (Ag, Au, Cu, and Pd), Ni and coal. The recovery of Al and Fe also constituted significant savings in resource consumption but to a smaller degree than the other metals.

Table 13: Environmental assessment from recovering Al, Ag, Au, Cu, Fe, Ni and Pd from 1 tonne of high-grade sWEEE. The assessment includes the avoided burdens from primary production. Results are shown according to the allocation method used (mass or economic value) and given in PE (Bigum et al., I)

	Mass	Economic
Environmental impact categories		
Acidification	-0.25	-0.27
Ecotoxicity in Soil	-1.13×10^{-3}	-1.10×10^{-3}
Ecotoxicity in Water, Chronic	-7.83	-4.41
Global Warming 100 Years	-0.25	-0.38
Human Toxicity via Air	-0.98	-1
Human Toxicity via Soil	-0.26	-0.5
Human Toxicity via Water	-0.48	-0.25
Nutrient Enrichment	-0.05	-0.07
Photochemical Ozone Formation	-0.02	-0.04
Stratospheric Ozone Depletion	-1.01×10^{-4}	-2.16×10^{-3}
Resource consumption		
Aluminium	-5.07	-5.07
Brown Coal (Lignite)	-0.41	-2.07
Copper	-11	-11
Crude Oil	-0.21	-0.49
Gold	-14.6	-14.6
Hard Coal	-0.62	-0.91
Iron	-3.93	-3.94
Lead	-2.50×10^{-4}	-5.21×10^{-3}
Manganese	-1.44	-1.44
Natural Gas	-0.18	-0.43
Nickel	-12.3	-12.3
Palladium	-63	-63
Silver	-11.7	-11.7
Uranium	-0.2	-0.03
Zinc	-0.04	-0.01

Conclusion

The recovery of metals from high-grade sWEEE as opposed to primary production shows environmental savings on all environmental impact categories. The environmental benefits of recycling were most likely underestimated, since the data on the burdens from mining and refining of ore were incomplete. Especially the handling and disposal of tailings and chemicals were not properly quantified due to lack of data but are considered important (Engels, 2006). Underestimation of this would, however, not change the conclusion.

That the highest savings in resource consumption was found for the precious metals and Ni is particularly interesting, as the overall recovery (both in terms of bulk material and recovery efficiency) was higher for Al and Fe. This underlines the importance of focusing recovery targets on individual metals and not as an overall bulk amount. This also supports the conclusions of Bigum et al. (III) that there should be a focus on the recovery of precious and scarce metals rather than the traditional metals - Al and Fe. However, this conclusion is also influenced by the modelling tool (Klinglmair et al., 2013), which is discussed in Chapter 7.1.5 of this thesis.

7 Discussion

This chapter describes and discusses some of the many complexities, challenges and perspectives found to be particularly related to special wastes and LCA.

LCA studies on special wastes are commonly done on single products and from a product point of view (Johansson and Björklund, 2009; Socolof et al., 2005). Waste-LCA modelling on mixed special waste is, however, still a relatively new field and is, therefore, continuously being developed. Only a few waste-LCA studies have been done on special waste as a mixed waste type (Bigum et al., I; Bigum et al., II; Hischier et al., 2005; Huisman et al., 2007, Wäger et al., 2011). Because these waste types are generally quite complex, challenges still remain on how environmental issues with these waste types can best be included and reflected in LCAs.

7.1 Challenges of waste-LCA of special waste

7.1.1 Flows and waste routing

The producer responsibility on WEEE and batteries means that the wastes after collection become the responsibility of the producers. In Denmark, an initial pre-treatment of the special waste would be performed, resulting in material fractions sold for further pre-treatment and recycling. Special waste contains valuable metals, targeted for recovery. During treatment, additional waste flows (manually removed substances and items requiring special treatment, such as plastic etc.) will arise. These might not have a high economic value, if any at all, but would still require treatment. Due to the producer responsibility, producers have to absorb any costs of the treatment and management of special waste. This means that there is a financial incentive influencing the waste routing. The treatment facility willing to pay the highest price or, in case of low value fractions, treat the waste for the lowest amount, is most likely to get the bid. The WEEE directive sets requirements that the best available technology (BAT) should be used, but this does not change the financial incentive which, in practice, means that the waste routing will vary. The producer responsibilities also mean that the destination of the special waste is often considered a trade secret. This, in connection to varying waste routing, makes it difficult to track, and, hence, also to model the special waste flows.

7.1.2 Composition and content

As shown in Figure 4, special waste is highly heterogeneous, and the variation in composition of the special waste types is significant. This makes it difficult to include special waste in waste-LCAs, as the notion of an average composition is not really applicable for these waste types.

Chemical composition analyses of the special waste types as a mixed fraction are demanding and expensive. Also, very detailed analyses measured on one mixed batch of special waste, might not be representative of the next, due to the difficulties of obtaining a chemical composition analysis within an acceptable uncertainty level (Morf et al., 2007). Conducting chemical analyses of mixed special waste streams on very large samples over a longer period of time could be one way to improve the data background. The data would be valuable and might possibly be used as generic data if this is supplemented with a sensitivity analyses on the influence of varying composition, as in Bigum et al. (III). Data on the chemical composition of mixed special waste could also be used for modelling best/worst case scenarios. However, even a strengthened data background for the chemical composition of special waste would not completely solve the issue with heterogeneity of the waste.

Another possibility for determining the composition of special waste is to use available literature as was done in Bigum et al. (III). Using literature data has the benefit of being easily available. Product-based compositional data are increasingly being published, as LCAs can be used as a tool to evaluate and optimise products (Duan et al., 2009; Choi et al., 2006). Using product-based compositional data could be used to represent a mixed waste stream. This would also be a means to ensure that the special waste management industry is able to adapt to the incoming special waste, due to technological development changes. However, as shown in Bigum et al. (III), the combination of the current available literature data is still not fully able to account for the content of the special wastes. Increasing the amount of available data would require that producers register the content of their products when placing their products on the market. This should then preferably be done according to standardized material fractions, so generic databases could be established.

Developing better and more complete datasets are decidedly necessary, as the environmental assessments (in particular the toxic categories and abiotic resource depletion) are directly related to chemical composition of the special waste fractions (Bigum et al., III).

7.1.3 Treatment technologies

In order to establish a more thorough understanding of the current management of special waste, there is a significant need for better knowledge of the special waste treatment technologies. This should include individual metal recovery rates, as the recovery of these metals was found to be environmentally significant and thus a significant parameter in environmental assessments (Bigum et al., I).

Pre-treatment

There are numerous pre-treatment facilities for WEEE in Europe, and the treatment that WEEE undergoes can vary (except for the mandatory removal of certain components as set by the WEEE directive). As with the difficulties with determining the waste routing, it is likewise difficult to determine which treatment technologies are used and what treatment steps. In addition to this, the treatment processes generate additional waste flows that also require treatment. These will also vary and possibly be mixed with non-WEEE waste types requiring the same treatment. This makes it difficult to track the treatment of special waste and to allocate the environmental burdens. Determining which treatment technologies are typically used and, thereby, also to establish both specific and generic LCI data on the special waste treatment technologies can be a challenge. Add to this that information is often considered propriety, as the treatment facilities are in direct competition with each other due to the nature of the producer responsibility.

Bigum et al. (I) included an example of a pre-treatment facility for sWEEE. The pre-treatment facility results in eight outputs, where two of them, i.e., the hazardous substances (the manually removed substances required to receive special treatment by the WEEE directive) and the plastic waste fraction were not included in the environmental assessment. The plastic fraction was excluded due to lack of knowledge on its waste routing and subsequent treatment. The substances requiring special treatment could be excluded due to the study setup (Bigum et al., I). The study setup meant that the treatment of these substances would take place both in the investigated system (secondary production) and the substituted system (primary production), as the manual removal of these compounds is determined by law. If wanting to assess the environmental burdens of the management of WEEE (e.g., with a functional unit being "treatment of 1 tonne of WEEE"), the treatment of the hazardous compounds should be included in the assessment. These compounds are most likely to be hazardous and require extensive treatment. It can, therefore, be suspected that the management and treatment of this fraction also results in high burdens. If data on the treatment of these substances could not be obtained, considering a worst- and best-case scenario could be an alternative.

Traditional pre-treatment facilities might not be suited for the identification and separation of the precious and scarce metals in special waste and will primarily focus on the traditional metals such as Al, Cu and Fe, which can be recovered in bulk amounts. This is most likely related to the relatively easier recovery of the traditional metals, the high concentration of the traditional metals in the waste, the value of these, and because the recycling targets are weight based (CEC, 2012). Recovering in bulk amounts is supported by the existing legislation. However, the total amount of metals contained in WEEE is also important (Oguchi et al., 2013), and the recovery of precious and scarce metals was, in Bigum et al. (I), shown to have significant environmental relevance, actually exceeding those of the traditional metals. From a resource point of view, and based on the findings in this thesis, recovery efforts and recycling targets should, therefore, aim at incorporating the recovery of the precious and scarce metals.

Metallurgical recovery

There are only a few metallurgical recovery facilities in Europe. The establishment of generic LCI data sets for the metal recovery from special wastes should, therefore, be possible. Detailed analyses of the facilities are most likely available, but it can be difficult to obtain access to data on the performance of a given facility. The metallurgical recovery facilities are multi-functional technologies, which recover metals from primary as well secondary sources. This can be an issue with regard to allocation when modelling metal recovery from an individual input, e.g., special waste rather than the mixed input and would require very detailed knowledge on the flows throughout the system (Stamp et al., 2013).

7.1.4 Allocation of burdens

Environmental assessments on systems with multiple inputs and/or products, such as waste treatment technologies for special waste, require a method for distributing the burdens and benefits between these two factors. This is particularly the case with special wastes, which contain metals. The ISO standards recommend that systems expansion be used to address the issue of distributing the burdens, but system expansion is rarely possible when conducting LCAs involving metals. Bigum et al. (I) addressed this issue by allocating according to both mass and economic value and found little difference in the allocation methods when taking a system perspective. Had the study aimed at determining the environmental burdens of the recovery of the individual metals (if, for instance, wanting to compare the burdens and benefits of the different metals in relation to each other), the choice of allocation method would have had a significant influence. The mass allocation approach would have put the majority of the burdens on the metals recovered as a bulk rate, and the economic allocation would have placed the majority of the burdens on the metals with the highest economic value (calculated on the basis of the actual amount recovered as well as the metal price). Both the mass and economic allocation methods rely on the chemical composition of the special waste fraction and are thus subject to the varying composition in the mixed special waste fractions and primary ores. Varying composition and metal prices will influence allocation, and the results of an LCA study. This means that results of an LCA on special waste where allocation has been performed, in reality only apply for the specific composition and metal prices used in that study. This means that preferably unallocated LCI data should be used, and that these should be allocated according to the specific system being investigated. However, since allocation is often required, when it comes to metals, and it can be difficult to obtain unallocated LCI data, using allocated data should be done with care and with a thorough check of the background for these data. This also means that generalisations and conclusions based on waste-LCAs studies could be difficult, and the assumptions and background of the studies should always be carefully investigated and assessed.

The influence of varying composition and metal prices on special waste-LCAs is something that could benefit from further investigation and assessment.

7.1.5 Assessment of resource depletion

Assessment of resource depletion in LCAs is related to the notion of how to assess resource scarcity. The scarcity issue, however, is not clearly defined.

“Rare” as a concept simply means “occurs less frequently than others” and is a highly subjective concept. The issue of “what is considered rare metals” was discussed by Behrendt et al. (2007). Behrendt et al. (2007) used three parameters to define rare or scarce resources. The first is related to the high value of a resource, as it can be considered that prices reflect the relationship between supply and demand. Secondly, the availability of the resources from known reserves, which are considered recoverable under current technical extractive abilities, with the price of the resource is a factor. And finally, the availability of the metals with respect to supply stability, and the political situations in the countries where the metals are extracted, was considered. The three criteria for availability mean that metals can be defined differently depending on the criteria used, as only some metals fulfil all three.

Schüler et al. (2011) defined "critically" according to demand growth (increase in demand of certain elements due to green growth and the EEE-industry), supply risk (physical scarcity as well as political-economic risks in the producing country), and recycling restrictions. Recycling restrictions included issues related to dissipative applications, limitations to and lack of suitable recycling technologies, and also lack of economic incentives supporting recycling. In the end, Schüler et al. (2011) concluded that there is a consensus between the different approaches for certain elements and could, therefore, be defined as being in supply risk.

The European Raw Materials and Supply Group identified a list of critical raw materials for the EU (ECEI, 2010). The criteria used did not include the issues of price nor geological scarcity but simply used supply risk as a factor.

Supply risk was related to two types: a) the political-economic stability of the producing countries; and b) an "environmental country risk," where the supply could be influenced by the countries taking actions to protect their environment. The European report did not consider geological scarcity to be an issue with the argument that only a small percentage of the Earth's crust have so far been explored, and that there is a large potential for discovering new deposits. In reality, this means that ECEI (2010) considers the geological availability to be infinite. Considering the geological availability to be infinite is an issue that is often discussed. Similar to this argument is the argument that geological scarcity is not an issue because increasing prices on materials when they start becoming scarce, will lead to exploitation of the economic reserve bases or mineral resources, which were previously not deemed economically feasible (McKelvey, 1980).

From an environmental point of view, the extraction and recovery of resources comes with high environmental cost, both in the form of energy use (cumulated energy demand), material demand (cumulated raw material demand), by-products and waste products (total material requirements) that need to be managed (Koch and Kohlmeyer, 2009; Bigum et al., I). Simply mining more of the target metals, would come with high environmental cost, which should also be a factor when talking about sustainable use of resources. Finally, the simple and modest notion of keeping resources in circulation for future generations, is increasingly being included in environmental thinking (Braungart and McDonough, 2008).

LCAs include resource depletion and scarcity by using "characterisation factors." In the case of resources, and due to the issues presented in the beginning of this chapter, these characterisation factors may vary substantially (Klinglmair et al., 2013). The relative ranking of resource depletion impacts differs between methodology, and the degree of scarcity of the individual resources also varies relative to each other (Klinglmair et al., 2013). Klinglmair et al. (2013) specifically mention the discrepancies between the EDIP 97 (Wenzel et al., 1997) and the CML (Oers et al., 2002) (used in Bigum et al. (I) and (III) respectively), and that a resource's relative importance strongly depends on the LCA modelling tool. This is of particular importance in relation to Bigum et al. (I) and (III), as one of the purposes of the Bigum et al. (I and II) studies was to provide the necessary background to be used to evaluate resources targeted for recovery. The findings by Klinglmair et al. (2013) highlights that LCA as a tool might not be ready to do this, or at least that consensus on which tools to use should be reached. The ILCD recommendations by the JRC (2010; 2011) might be the first step in achieving this. As a consequence of this, the assessments of the individual resources, conducted in Bigum et al. (I) and Bigum et al. (III), should not be compared directly as different modelling tools were used. However, the broader general conclusions, which can be drawn from both, is that the environmental importance and benefits of recovering metals is significant.

Another important issue, relating to modelling of resource depletion in LCAs, is whether to use mid-point or end-point models. Mid-point models reflect the early stage of a cause and effect chain, where the end-point indicator attributes these to the environmental issue (or area of protection) giving cause for concern (e.g., human health, natural environment or natural resources) (ISO, 2006a; JRC, 2011; Hauschild et al., 2012). Presently, the ILCD is only able to recommend one mid-point model for the evaluation of resources, and this is the CML 2002 method (Oers et al., 2002). No end-point level can be recommended, as the ReCiPe model is still considered to be under development. That the ILCD recommended method to LCA practitioners includes the issue of scarcity, this strengthens resource scarcity as a valid impact category and means that it will most likely be further elaborated on and incorporated in future LCAs.

How to practically evaluate resource scarcity in LCAs is still an issue that is widely debated within the LCA community without reaching final consensus

(Klinglmair et al., 2013). The methodology, therefore, continues to be developed, often with different backgrounds (Yellishetti et al., 2009). Even the issue of whether or not to include resource scarcity in environmental assessments is still being debated with the argument that impacts of the extraction of resources is related to the extraction, not the scarcity itself (Klinglmair et al., 2013). A general discussion on how to assess resource scarcity is greatly needed for a broader consensus and further scientific development in order to ensure that LCA is applicable and accepted as a decision-making tool for environmental assessments, as well as a base for political strategies.

7.2 Perspectives in waste-LCA of special waste

In order to ensure that LCA as a methodology is applicable to waste management, it is necessary to be aware of its limitations and what influences the results. The generated information and results are neither complete, objective nor accurate (Ekvall et al., 2007). Waste management of special wastes is very complex, and LCA as a methodology has the potential to include many of the aspects and parameters in the evaluation. The LCA methodology can benefit from incorporating certain issues related to special wastes, which would further ensure that the methodology continues to be considered useful as a waste-LCA tool that can support sound decision making.

As discussed in this thesis, resource evaluation in connection with special wastes is one issue that is highly related to LCA as a methodology.

One aspect of resource evaluation entails that the actual evaluation of the recovered resources needs to be strengthened by improving the scientific bases for assessing the quality of the recovered resources and the environmental aspects connected to this. This includes accounting for impurities, contaminants and residuals from the recovery processes, thus allowing for a better understanding of the environmental impacts connected to these (Astrup et al., 2013). This should also be supported by a better understanding of the environmental impacts of the extraction of primary resources, as these result in significant environmental impacts (Bigum et al., I; Koch and Kohlmeyer, 2009).

Another aspect of resource evaluation is the choice of system boundaries, when assessing resource depletion in waste-LCAs. The system boundaries in waste-LCAs traditionally only include the waste phase, and any environmental impacts prior to the waste phase are disregarded (the zero burden approach). The zero burden approach basically means that the waste management sector cannot be held accountable by any inherent environmental burdens contained in the waste. Taking a zero burden approach has merits, as the responsibility of the contained resources lies with the producers in the design and manufacturing phase, which the waste management sector has little influence on. However, the zero burden approach is applied even though many of the "consumed" resources are still present in the waste and, if recovered, will result in a saving. Unrecovered resources will, on the other hand, not give a negative value to the resource depletion impact category and will, in an assessment, not be considered a burden. This approach means that the issue with the unrecovered resources is not included in waste-LCAs and is, hence, not a part of the decision-making process. This leads to a risk that choices regarding waste management options might not be fully enlightened and that, despite it receiving political focus, the issue of resource depletion is missed. Waste-LCAs should also be able to include accounting for the waste-specific resource depletion when making long-term waste management decisions. If it does not, the waste-LCAs will, in reality, only reflect the present technologies, with risks being used merely to optimize the existing waste management policies.

Bigum et al. (III) used the "resource burden approach" to quantify the loss of unrecovered resources and included waste-specific resource depletion in addition to the resource recovery. Although other challenges with regard to how to actually account for and assess resource depletion still have to be addressed, it shows that inherent resource depletion does play a role and should be included. Inclusion of waste-specific resource depletion should be part of environmental assessments and the overall consequences of waste management. This would lead to an increased focus within the waste management sector on the recovery of other resources than those traditionally recovered (carbon, ferrous and non-ferrous metals).

8 Conclusions

This thesis provided necessary background information for the understanding of special waste management and the environmental concerns related to these. It also addressed the issue of environmental assessment of special waste in an LCA context and contributed to a broader understanding of what to include in waste-LCAs of special waste, as well as the challenges and perspectives of this. The main findings of the research can be summarised as follows:

- Misplacement of special waste with residual household waste was found to be significant, amounting to 3,484 tons of WEEE and 538 tons of batteries per year. The misplacement of WEEE and batteries were, however, not found to be able to account entirely for consistent differences between what is marketed and what is collected. This suggests that there are flows of special wastes that are still unaccounted for.

- There are improvements to be made for collecting special wastes in Denmark. Information material should focus on addressing specific items that are frequently misplaced. These are small household appliances, electric tooth brushes, wrist watches, clocks, cables, headphones, toys, flashlights, bicycle lights and lamps.

- LCIs for the recycling and recovery of metals in special waste were established. It was found that using unallocated LCIs should always be preferred and then be allocated according to the specific system being investigated.

- It was found that the lack of knowledge, uncertainty and variations related to the composition of special waste is particularly important, as composition is directly related to the environmental impacts. In particular, the content of rare, precious and hazardous metals should receive the greatest focus.

- Knowledge of the special waste treatment technologies (efficiencies, elemental partitioning, etc.) was also found to be an important issue related to waste-LCAs of special waste having a direct influence on the results of an environmental assessment.

- The depletion related to unrecovered resources was found to be significant and should be included in the resource depletion impact category. A resource burden approach should be taken when conducting waste-LCAs, where special waste is a factor. Overall, it could be concluded that special waste streams should be collected and recycled separately from the other waste streams due to both hazardous pollutants in the wastes and because of the significant environmental benefits of recovering resources in the wastes. In particular, precious and scarce metals should be recovered.

9 Further research and perspectives

In connection with environmental assessment of special waste and hazardous waste types, the following topics and aspects were identified as needing further research:

- A study on citizen behaviour with respect to source segregation would be beneficial. The study should aim at gaining knowledge on what makes people use the dedicated collection schemes for their special waste, and what makes them misplace it along with their residual household waste. Of particular interest would be to investigate WEEE and the hypothesis on "recognition," where some special waste types and items seem not to be viewed as WEEE that require separate collection.
- An evaluation of the different dedicated collection schemes with respect to the efficiency of the schemes should be conducted. This would be highly beneficial as a means for improving the collection of the special waste types.
- A study on the flows of special waste streams aiming at determining other unknown flows of special waste should be conducted. Knowledge on alternative waste disposal routes is required, when seeking to improve collection of the special waste types.
- Comprehensive elemental composition analysis of special waste types, both on a product level and a mixed waste fraction, are challenged by the heterogeneousness of the special waste types. There is a most important need for better data on this, either via direct sampling or by product information from the producers. Especially, the inclusion of scarce and precious metals should be included in analysis, due to their high environmental and economic relevance.
- Increased knowledge and data on the special waste treatment technologies regarding recovery rates, efficiencies and environmental impact (emissions of substances to air, water, soil, as well as energy consumption, etc.) is required in order to evaluate if the existing management and treatment of special waste is sufficient, and how technologies could be improved. Currently, limited data on this are available, which makes it difficult to evaluate and perform environmental assessments on special waste management.

- The environmental burdens of the primary production of metals contained in special waste need to be better addressed in order to be able to quantify the potential environmental savings of recycling.
- Allocation of the environmental burdens and savings are a key factor for environmental assessments of special wastes. Allocation is dependent of the composition of the special waste streams as well as the value of the recovered fractions. The influence of these aspects on LCA should be evaluated, in order to gain knowledge on the robustness of LCA results related to special wastes.
- Incorporation of issues related to resource depletion, including the continued work with updating characterisation factors to represent depletion, and benefits of resource recovery in waste-LCAs, need to be further improved.
- Finally, consensus on LCA methodologies, including characterisations factors, for assessing resource depletions needs to be obtained.

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In this online version of the thesis, the papers are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from:

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The Department of Environmental Engineering (DTU Environment) conducts science-based engineering research within four sections:
Water Resources Engineering, Urban Water Engineering,
Residual Resource Engineering and Environmental Chemistry & Microbiology.

The department dates back to 1865, when Ludvig August Colding, the founder of the department, gave the first lecture on sanitary engineering as response to the cholera epidemics in Copenhagen in the late 1800s.

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