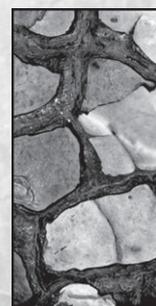


Evaluation of Waste Specific Environmental Impacts from Incineration

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INSTITUTE OF ENVIRONMENT & RESOURCES



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Preface

This thesis is the result of three years of reach at Institute of Environment & Resources at Technical University of Denmark (DTU) involving practical work in the field and the laboratories and modelling at the office. In the field I have been performing tests at incinerators and sampling of waste in large amounts. The sampling of wastes in the field and the lab has been challenging for me as I had to realize the principles of correct sampling according to the theory by Pierre Gy. As member of the research group, I have followed and contributed to the development of the EASEWASTE model. This has been an interesting and challenging task that has sometimes been frustrating but in comparison to the result worth the trouble. I have been involved in design, construction, testing and documentation of the incineration sub-model in EASEWASTE. Three of the incinerator tests (indirect analysis), which I performed, were part of projects carried out for a client with a predefined scope and with concluding reports. Applying, budgeting, planning, performing and reporting such projects have been a part of my PhD and have brought me knowledge on the balance of costumer and scientific needs. Furthermore it has forced me to work in accordance to a time budget which has been very instructive for me.

During the PhD work five articles of relevance to this thesis have been produced and they are present as below and as appendixes at the end of this thesis and referred to in the text as “Riber et al.” followed by their roman number (I-V). Additionally the past years of research have lead to the following public communication that covers as well presentations as published journal articles:

Included papers

- I Riber C., Fredriksen G.S. and Christensen T.H., 2005, Heavy metal content of combustible municipal solid waste in Denmark, *Waste Management & Research* 23, 126-132.
- II Riber C., Rodushkin I., Spliid H. and Christensen T.H., 2007, Method for fractional solid-waste sampling and chemical analysis, *International Journal of Environmental Analytical Chemistry*, 87, 5, 321-335
- III Riber C., Christensen T.H., Bhandar G.S., 2007a, Environmental assessment of waste incineration in a life-cycle-perspective (EASEWASTE), Submitted to *Waste Management & Research*.
- IV Riber C., Claus Petersen and Christensen T.H., 2007b, Chemical composition of material fractions in Danish household waste, Manuscript.
- V Riber C. and Christensen T.H., 2007c, Indirect Determination of Chemical Composition and Fuel Characteristics of Solid Waste, Manuscript.

Presentations

- Riber C., Research on waste incineration, Presentation at internal meeting at Babcock & Wilcox Vølund A/S, January, 2007
- Riber C., TOS and waste characterization, Presentation at internal meeting at Danish Standards, January, 2007
- Riber C., TOS and waste characterization, Presentation at workshop on Theory Of Sampling at Danish Standards, December, 2006

- Christensen, T.H., Riber, C., Chemical characterization of 45 material fractions in Danish household waste, Waste characterization seminar DTU, October, 2006.
- Riber, C., Christensen, T.H.: Sub-sampling of material fractions in household waste, Presentation at Waste characterization seminar DTU, October, 2006.
- Riber, C., Christensen, T.H.: Waste characterization by full scale waste incineration test, Presentation at Waste characterization seminar DTU, October, 2006.
- Riber C. Christensen T.H., Heavy metals in waste and ash products, Presentation at DAKOFA conference "Incineration versus co-combustion", April, 2006
- Riber C., Incineration Tests: Experience and conclusion from tests at the incinerators in Aarhus, Taastrup and Herning, Presentation at the Seminar APC products from waste incinerators at DTU, June, 2006.
- Riber, C., Incineration Tests in Aarhus, Presentation at internal DAFONET meeting, March, 2004.
- Riber C., Fredriksen G.S., Waste chemical composition and TFCs from incinerator green accounts, Presentation at internal DAKOFA meeting, September, 2003.

Publications in proceedings

- Riber C., Spliid H., Christensen T.H., Direct analysis of solid household waste and the related sampling variance, CEN Seminar on Chemical and Environmental Sampling in Brussels, April, 2005.
- Riber C., and Christensen T.H., Characterisation of waste types by direct and indirect analysis, Tenth International Waste Management and Landfill Symposium, Sardinia October, 2005.

Other publications

- Zeuthen J. H., Pedersen A. J., Hansen J., Frandsen F., Jappe H.L., Riber C., Astrup T., (2007), Combustion Aerosols from municipal waste, Combustion Science and Technology, 179:10, 2171 – 2198.
- Riber C., Christensen T.H., (2007), Waste incinerator characterization and environmental impact assessment, Report for Aarhus Municipality.
- Riber C., Christensen T.H., (2006), Waste chemical composition by indirect analysis, Report for Herning Municipality.
- Riber C. and Christensen T.H., (2006), Measurement of chemical composition in household waste and small combustible waste, Danish Environmental Protection Agency project No.1085, Copenhagen (In Danish but summary in English).
- Riber C., Kirkeby J., Christensen T.H., (2005) Life cycle inventory for the Aarhus incinerator, Report for Aarhus Municipality.
- Riber C., Fredriksen G.S., Christensen T.H., (2003), Hidden knowledge in Green accounts of incinerators, Stads og Havneingeniøren 2003:9.

Network participation

- DAKOFA: Working group on incineration
- Danish Standards: Working group on waste characterization
- Danish Standards: Working group on sampling

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My PhD work has been funded by various projects. Though the project did not have all funding in place from the beginning, we decided to proceed, and I would like to thank Professor Thomas H. Christensen for his persistent belief in the project and in my abilities throughout the project. I would also like to thank Thomas for his assistance throughout the project with a balanced amount of requirements, suggestions, encouragements and contributions and for creating an inspiring and innovative work environment.

The projects funding this study have been of major importance to the outcome and the organizations behind with which I in many cases have been collaborating closely with are all thanked (Miljøstyrelsen, DONG Energy A/S, I/S Vestforbrænding, I/S Amagerforbrænding, Århus Affald & Varme, Herning Kommune). Especially thanks to: Århus Affald og Varme, Hanne Rasmussen and Erik Damgaard; I/S Vestforbrænding, Poul-Erik Sørensen and Knudmoseværket, Ove Jespersen.

Without the physical help and assistance of lab technicians at the Institute of Environment & Resources, especially from Denny Viholt, the practical part of the project would have been a lot more troublesome. I would also like to thank my past and present colleagues: Thomas Astrup, Jiri Hyks, Gry Janniche and Trine L. Nidel for their practical help in performing tests at incinerators around Denmark. Last but not least I would like to thank Torben Dolin, Gry Janniche and Susanne Sønderstrup for their assistance in proof reading, commenting and suggesting changes to the draft of thesis and articles.

*I dedicate this work as well as so much else in my life
to my wonderful wife and my three fantastic boys.*

Summary

In Denmark, waste incineration was introduced more than 100 years ago and has for many years been the most important waste treatment option. However, what are the drawbacks of incinerating our waste and will waste management schemes ensure a sufficiently clean waste to secure only a minimum of emissions? This thesis aims at evaluating the impacts from incineration of waste on the environment but focuses on the impacts related to the waste. These impacts, called waste specific, are concluded to be as important to environment as the impacts from the process controlled only by the installed technology at the incinerator. Knowledge of the waste composition of material fractions and their chemical content is necessary in the evaluation process which was performed for 48 material fractions of Danish residual household waste. The mass of the primary sample in the direct analysis of the material fractions were not big enough to include rare items of importance to the elements in trace concentrations. Therefore indirect analysis using full scale incinerators was performed on all together 1400 tonnes of the same waste type and the result of the direct analysis was corrected to the result of the indirect analysis. Using the corrected waste analysis as input in a Life Cycle Assessment (LCA) on incineration showed that the most important contents of the waste are (in prioritized order):

1. Mercury
2. Energy
3. Arsenic

Up to half of the *Hg* content in the waste was concluded to be present in very rare items (1/800 tonnes) and this parameter was the most important single parameter in the environmental evaluation of a Danish incinerator.

Resume

For mere end 100 år siden blev affaldsforbrænding introduceret i Danmark og siden er forbrænding blevet en stadig vigtigere del af affaldshåndteringen. Spørgsmålet er hvad der er konsekvensen af vores brug af affaldsforbrænding og om vores affaldshåndtering leverer tilstrækkeligt rent affald til at sikre et minimum af miljøpåvirkninger. Denne afhandling har til formål at vurdere affaldsforbrændings påvirkninger på miljøet men med fokus på de påvirkninger der stammer specifikt fra affaldet. Disse affalds specifikke påvirkninger er fundet lige så vigtige som de påvirkninger der stammer udelukkende fra affaldsforbrændingsprocessen. For at vurdere de affaldsspecifikke påvirkninger må affaldets sammensætning af materialefraktioner og deres kemiske sammensætning være kendt. Dette kendskab er skabt gennem direkte analyse af 48 materialefraktioner fra rest dagrenovation fra danske husholdninger samt en indirekte analyse af 1400 ton af samme affaldstype. Den indirekte analyse blev gennemført fordi den direkte analyse blev konkluderet til at have en for lille primærprøve til korrekt at kunne repræsentere sjældne objekter i affaldet. Den direkte analyse blev korrigeret til resultatet af den indirekte analyse der inkluderede de sjældne objekter der er vigtige for bestemmelsen af flere elementer der er kritiske for analysen af påvirkninger. En livs cyklus analyse (LCA) blev udført på forbrænding af det korrigerede resultat af den direkte analyse. Resultatet af analysen viser at de følgende tre parametre i affaldet er af stor vigtighed for et forbrændingsanlægs samlede miljøpåvirkning (i prioriteret rækkefølge):

1. Kviksølv
2. Energiindhold
3. Arsen

Op mod halvdelen af affaldets kviksølvindhold konkluderes at stamme fra sjældne objekter i affaldet (1/800 tons) og netop disse objekter viste sig at være den vigtigste enkeltparameter i en miljøvurdering af et dansk forbrændingsanlæg.

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Riber C., Rodushkin I., Spliid H. and Christensen T.H., 2007, Method for fractional solid-waste sampling and chemical analysis, International Journal of Environmental Analytical Chemistry, 87, 5, 321-335

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V

Riber C. and Christensen T.H., 2007c, Indirect Determination of Chemical Composition and Fuel Characteristics of Solid Waste, Manuscript.

The appendixes are not included in this www-version but may be obtained from the Library at the Institute of Environment & Resources, Bygningstorvet, Building 115, Technical University of Denmark, DK-2800 Kgs. Lyngby (library@er.dtu.dk)

1 Introduction

For more than a century, waste incineration has played a major role in Danish waste management systems and processes have consequently developed considerably. From being simple open fires with great environmental impact and only mass and volume decrease as advantage it has evolved into an advanced process, almost entirely under human control /Kleis et al., 2003/. However, even today, the waste input seems to be the one parameter which we still cannot control, and which may have negative effects on the surrounding environment.

As waste amounts have increased with the increase in activity of our society and our awareness of the consequences to environment, so has also the importance of sustainable waste management become generally more widely recognized.. With modern tools for evaluating the impact on environment like Life Cycle Assessment LCA, we have become aware of impacts that we did not see before, and former principles, experiences and myths must therefore be abandoned. Complexity has increased with introduction of LCA to waste management but the level of understanding and reliability in our decisions has also increased dramatically.

Waste incinerators are traditionally acknowledged as main contributors to the impact from waste management on the environment. The question is why. It is the main purpose of this thesis to evaluate the influence of waste management on the environmental impacts from waste incineration. Not all impacts from waste incineration are related to the waste incinerated as some impacts are a result of the incineration process. Waste management options will change the composition of the waste and thereby influence the impacts on environment by incineration. However waste management will not change the processes at the incinerator. The relations between impacts and incinerated waste are called waste specific impacts and will be the main focus of this thesis.

LCA converts the incinerator emissions to impacts on environment such as global warming or toxicity to humans and the first chapter of the thesis will therefore evaluate emissions from waste incinerators. Hereafter the input waste compositions are evaluated with specific focus on substances recognized of importance in the emissions evaluation. With the information on emissions and input waste composition, it will be possible to perform an LCA, which is done in chapter 3. The outcome of the LCA and the knowledge of emissions and waste composition will enable an evaluation of the waste specific impacts from waste incineration on environment. This evaluation will be of importance in future waste management evaluations, as it brings focus to the important characteristics of the waste when incinerated. Furthermore products can be compared for their potential effect when incinerated solely from knowledge on their chemical composition and heating value.

2 Emissions from Incineration

When evaluating the total impact on environment from a waste treatment technology, all possible and known impacts must be quantified to ensure a correct evaluation. The evaluation is based on present knowledge level on subjects like substances impact, inventory methodology and assessment methodology. Knowledge on substance environmental impact has developed over time, resulting in increased restrictions on waste incineration, last with the air emission limits on dioxin /DIRECTIVE 2000/76/EC/. The accumulation of experience in complex environmental evaluations of waste systems as for instance reported in Christensen et al. (2007) has lead to a development in the inventory requirements for waste processes. Today it is for instance required to include detailed descriptions of the energy producers that alternatively would produce the energy from the process. While the methodology of the assessment will be described later, the following will focus on substances and processes leading to potential significant emissions. The emissions to environment from incineration can be split in two categories: emissions directly connected to the waste content of a substance [waste specific] and emissions related to the process [process specific] either because the process limits the emission or because the process produces the emission. The terms of waste or process specific emission will further more indicate whether the emission could be lowered by removing a fraction of a given substance in the waste.

2.1 Process Specific Emissions

The fact that incineration of waste will lead to air emissions of Dioxin, NMVOC¹, CH₄, N₂O₂, CO, NO_x, HCl, HF, SO₂, CO₂, H₂O and dust. Furthermore, changing the waste to ensure the absence of these emissions is impossible. Contrary to the other substances, the emission of CO₂ and H₂O from the combustion and drying of waste is not controlled by any air pollution control (APC) system. Therefore their emission in this context is considered waste specific. Some might argue that emissions of NO_x, HCl and SO₂ would not occur if *N*, *Cl* and *S* were not present in the waste. This may be true. However, the emission to the air is controlled by the APC system and therefore not related to the waste content of the substances. The APC system is designed to be able to handle variations in the waste, and only variations outside design limits might lead to a waste specific emission.

Direct emissions from waste incineration to soil and groundwater have not been reported and are therefore not further described. Emissions to sea and/or surface water have in some cases been evaluated as process specific if system boundaries are expanded. Sea water emissions might occur if wet APC produces waste water that is discharged to the municipal sewers and the sewage treatment plant is considered within system boundaries. The treatment of the wastewater will not remove all substances and an emission has to be assigned to the incineration process. Emissions to surface waters occur if ashes from the process are used in construction works and substances leach from the ash. Research has shown no or very limited connection between ash substance content and leachate content why these emissions are considered process specific /Zomeren et al., 2004/.

¹ NMVOC; Non Methane Volatile Organic Compounds

Process specific emissions are defined as the measured emission in relation to the input mass of waste to the incinerator as further described in Riber et al. III. Most air emissions are measured continuously and are, in Denmark, reported annually in public available green accounts, which may be used as simple inventories like in Riber et al. I. Reduction of any process specific emission would require a technical solution regardless of the input waste and is not further described here.

2.2 Waste Specific Emissions

Waste potentially contains any substance that is used in consumer products, commercial business or industry, and the assessments of substances begin with an evaluation of which substances should be included. As incineration will destroy organic substances, only assessment of conservative substances², and of those, only some have impact potentials³, are of interest and focus will, in the following, be on conservative substances. When evaluating our knowledge of impacts to environment, it becomes clear that we do not have information on all substances emitted /Kirkeby et al., 2006/. The substances of most environmental concern are listed in Table 1 in alphabetic order. However in addition to the substances in Table 1, also iron (Fe) and aluminum (Al) scrap (in bottom ash) are of interest, as they have potential for recycling.

Table 1 Substances of environmental concern when incinerated.

• Antimony (<i>Sb</i>)	• Chromium (<i>Cr</i>)	• Molybdenum (<i>Mo</i>)
• Arsenic (<i>As</i>)	• Cupper (<i>Cu</i>)	• Nickel (<i>Ni</i>)
• Cadmium (<i>Cd</i>)	• Lead (<i>Pb</i>)	• Zink (<i>Zn</i>)
• Carbon (<i>C</i>)	• Manganese (<i>Mn</i>)	
• Chloride (<i>Cl</i>)	• Mercury (<i>Hg</i>)	

Impacts connected to substances in the waste can be of negative, positive or unknown influence on the surrounding environment. Waste specific emissions from an incinerator can be evaluated on their influence on the environment. Examples might be unwanted depletion of ozone or toxicity to aquatic organisms. It is only possible, with acceptable uncertainty, to predict effects from the produced products of the incineration process a limited number of years ahead. In LCA, 100 years are normally used as the assessment timeframe, although we know that this period does not account for the total potential emissions. To avoid leaving potential impact unaccounted for, Birgisdottir et al. (2007) introduce the concept of stored toxicity. This concept accounts for all residual potentials not emitted within the timeframe of the assessment.

Calculating a waste specific emission will include the transfer coefficient⁴ (TFC) of the substance to an output (e.g. air) multiplied by the waste content of the substance, as further described in Riber et al. III. The transfer to air is defined as measured substance mass in the air emission divided by the mass of substance input to the incinerator with

² Conservative substances; substances that are not destroyed or altered in the process

³ Impact potentials; a number stating the potential impact related to a reference, e.g. methane emission given as global warming potential in kg CO₂ equivalents

⁴ Transfer coefficient (TFC): a number describing the fraction of a substance in the waste found in a given output

the waste (see chapter 3.2). In theory, this methodology makes it possible to define TFCs for each material fraction in the input waste. However, the current level of knowledge does not yet allow for this. In the following, all material fractions are assumed to have the same transfer coefficient.

Air emissions

The only direct waste specific emissions always present from incineration is through the stack to the air as other outputs are usually either further treated before the emission occur or land filled. Depending on the substance and the APC technology applied, products like sludge, fly ash, waste water or spent activated carbon could contain the substances. If increased substance concentrations in the waste occur, the APC will not react to this; and even though a larger mass of the substance would be retained by the APC the air emission would still increase. If the APC react to increased substance concentration, the emission would be regarded process specific as for instance waste *Cl* content and HCl air emission.

If the APC reaches a maximum capacity of retaining a substance, the emission would not be estimated correctly with the used methodology as it relies on a constant transfer coefficient. This could for example be the case for *Hg* where the APC has a maximum capacity defined by the sorption capacity of the activated carbon used. If the waste concentration of *Hg* exceeds the sorption capacity of the activated carbon, the transfer coefficient to air will change and the emission increase. One way to assess these limitations of the APC is described in chapter 4.3.

Emission of C as CO₂ is only associated with a greenhouse effect if the origin of the carbon is fossil. Within the timeframe of a LCA (100 years), biogenic carbon is considered in balance and will not result in any greenhouse effect.

Non air emissions

The production of materials at the incinerator, like for instance energy or ash for road construction, will substitute for alternative production. Depending on the environmental burden of the alternative; the resulting environmental influence could be either positive or negative. Materials produced also have a financial value and this may as well be both positive and negative or not accounted for.

Production of bottom ash is waste specific as it is the result of the transfer of waste ash content. The bottom ash holds resources like aluminum and iron scrap that is, in most cases today, subject for recycling on commercial terms and with an environmental benefit associated. The recycling of the treated bottom ash in road constructions substitutes for gravel and Birgisdottir et al. (2007) states that the recycling results in no benefit or burden on the environment.

Fly ash alone or in a calcium mixture, depending on the APC technology, has no market value and is considered hazardous waste and is disposed off as such. In the Danish context, these ash products are stored in deep salt mines or likewise where they substitute for other fill. The Danish solution for these ashes is not believed to result in any environmental benefits or burdens. Other APC products like filter cake, sludge, spent activated carbon and gypsum are deposited like fly ash and evaluated as such. The fact that especially fly ash contains a lot of heavy metals and therefore works as a sink for these metals is not assigned any environmental value in this thesis, although it could be argued that it is a benefit to society.

Saved emissions

Energy production (heat or power) is the result of the waste energy content (carbon content) and accounted for by the heating value of the waste and a production efficiency (based on sold energy) as described further in Riber et al. V.

Electricity is in Denmark primarily produced at coal fired power plants. These are generally accepted to be the marginal⁵ production method for electricity, meaning that coal power production would be used to produce the last power needed. When a Danish waste incinerator produces electricity, the marginal process (here coal) is substituted and the avoided emissions can be credited the incinerator. When substituting the power plant it is necessary to take the production efficiency into account. However, if district heating is also produced at the power plant, this production must be left out, as the emissions must correspond to marginal production where no district heat is produced. CO₂ emission is avoided (substituted) when a fossil fuel fired power plant is substituted with waste incineration, but other emissions from the power plant might also be of importance, as concluded in chapter 4.2.

District heating is widespread in Denmark and has been so for more than a century /Kleis et al., 2003/. Waste incinerators therefore nearly always produce hot water for district heating. The district heating grids are usually local or regional as are the marginal processes. The local marginal could for instance be a coal or gas fired power plant or a biomass fired boiler. After the marginal process (plant) has been determined for the specific incinerator, the emissions of the process must be determined, and an assessment is made to show the amount of district heat production associated with emissions. For a given coal fired power plant, that uses seawater as coolant, it is by the plant owner estimated that, 50 % of the district heat production is gained as an increase in total efficiency /Riber C., et al. III/. Therefore only the remaining 50 % of the production is associated with fuel consumption and related emissions.

Substitution could also be negative to the incinerator (increase emissions) if the process substituted only cools on the same grid as the incinerator and at the same time is more electric efficient than the incinerator. This would result in a system deficit of electricity pr. produced energy unit of heat and that deficit would have to be accounted for by the incinerator with the grid marginal process (coal). Even though this case may seem theoretical, it is nevertheless the case in the city of Herning, Denmark, where a biomass and gas fired power plant is situated together with the incinerator. The service of volume reduction, hygienization, immobilization and similar to the waste by the incinerator is not credited environmentally as these services to society are not politically accepted.

⁵ Marginal means the small last/extra amount

3 Analysis of Waste

Waste specific emissions and impacts are a result of the waste content of a substance, and the success of any attempt to reduce the emission relies on detailed knowledge about the waste content. When investigating waste content of substances for evaluating waste specific emissions, it would be sufficient with a measure referring to the waste total content. However, if the waste specific emissions had to be reduced, the contaminant in the waste would need to be removed or lowered. To do this we would need to know the original contaminant material fraction. Subsequently it is necessary to measure the chemical content on all relevant material fractions of the waste by direct analysis as described in Riber et al. II. However, as argued later in this chapter, it will also be necessary to measure the waste chemical content by indirect analysis, using a full scale incinerator if heavy metals are to be included. When a waste specific impact is lowered by removal of a specific material fraction, other effects of removing the fraction also have to be evaluated. This could be effects on waste H₂O or ash content or heating value, which could be relevant in evaluating the effect on environment from the incinerator after the removal.

3.1 Direct Analysis

A direct waste analysis involves digestion and analytical quantification on the unchanged matrix of the waste. This is not feasible as the choice of chemical digestion procedures depend on unified physical properties to perform reasonable reproducibility and short term variability. Any mixed waste type must therefore be sorted prior to chemical digestion. However, the sampling also requires sorting of the waste. To reduce sampling errors, particle size reduction, mixing and mass reduction must be performed on the sample. These operations involve equipment that also relies on unified physical properties. Sorting the mixed waste into unified sub-samples will therefore be the first step in a direct analysis.

Not much scientific literature is available on methodologies of direct waste analysis in terms of chemical composition analysis of mixed wastes as household waste or Municipal Solid Waste (MSW). In Riber et al. IV, only two pieces of relevant comparative literature were found: Rotter S. (2001) and a study by the UK Department of the Environment (1995). The methodology described in these two studies is different in the sampling treatment compared to the one used here as further described in Riber et al. IV.

The direct analysis of this study involved analysis of 48 material fractions, hand sorted from residual household waste as described in Riber et al. II and IV. The names of the fractions are shown in Table 2 and will in the following be referred to in quotation marks. The following steps of direct analysis in the methodology described in Riber et al. II can be summarized as follows:

- Coarse size reduction by shredder (10-15 mm)
 - Mixing and mass reduction by riffle splitter
- Size reduction by crushing or cutting mill (3-4 mm)
 - Mixing and mass reduction by riffle splitter
- Fine size reduction by disc mill or cutter mill (0,1-1 mm)
 - Mixing and mass reduction by riffle splitter

The used equipment must fit to the material properties of the fractions and in Table 3 the used equipment to treat each material fraction in household waste is shown.

Table 2 Names of material fractions investigated in direct analysis of residual household waste.

Vegetable waste	Dirty paper	Wood	Aluminum containers
Animal waste	Dirty cardboard	Textiles	Aluminium foil
Newsprints	Kitchen tissues	Shoes, leather	Metal like foil
Magazines	Soft plastic	Rubber	Metal containers
Advertisements	Plastic bottles	Office articles	Other of metal
Books and phonebooks	Other hard plastic	Cigarette butts	Soil
Office paper	Non-recyclable plastic	Other combustibles	Rocks, stones and gravel
Clean paper	Yard waste	Vacuum cleaner bags	Ceramics
Paper/cardb. containers	Animals etc.	Clear glass	Cat litter
Cardboard	Nappies	Green glass	Other non-combustibles
Carton with plastic	Cotton sticks	Brown glass	Batteries
Carton with Al foil	Other cotton etc.	Other glass	Residual

Like the sampling, also the digestion procedures must be fitted to the material properties of the individual fractions. The digestion procedures used for each material fraction of the waste is described in detail in Riber et al. II and briefly shown in Table 3.

Analyses of sampling variance for two material fractions in Riber et al. II show that for most substances, the sampling method has been satisfactory except for the first shredder step for the one fraction. Some substances showed significant variance and a Kolmogorov-Smirnov test argued that this step significantly added variance. Riber et al. II states that this variance was caused by items of high concentrations that are smaller than the shredding diameter and therefore not affected by the shredding. Another explanation could be cross contamination or contamination of the sample by the shredder. When variance added to the over all waste is, because of the low concentration levels of the fraction, relative small and insignificant to the overall result is to be expected. The direct analysis described in Riber et al. IV on residual household waste concludes that the main part of the heavy metals originates from inert fractions:

- *Cd, Ni, Mn* and *Hg* are mainly in the waste as the result of the presence of “Batteries” in which these metals are present in very high concentrations
- *Cd* originates mainly from “Batteries” but is also represented by other the non combustible fractions.
- *Pb* is mainly present in the fraction “Other metal”.
- *Mo* is mainly present in “Other non combustible”.

Table 3 Details of sampling operation and analytical digestion procedures /IV/

Sample fractions	Sub-sampling		Analytical digestion	
	Grain size reduction	Mass reduction	Reagents	Digestion Conditions
Glass, soils, ceramics, other non-combustibles	Rotary disc mill (Siebtechnik IS100A, Mülheim an der Ruhr, D)	Riffle splitter (Rationel Kornservice RK12, Esbjerg, DK)	5ml HNO ₃ + 5 ml HF	Open vessel, Temperature 120°C, 72 h at re-flux conditions, evaporation, dissolution in 7M HNO ₃
Metals and metal containers (including Al), batteries	Titanium drilling	Simulated 1. dimensional sampling	10ml HCl+ 3ml HNO ₃ +0.1 ml HF	Open vessel, Temperature 120°C 24 h at re-flux conditions
Food waste, all types of papers, cartons, cardboards, textiles, garden waste, soils, wood, cigarette butts, vacuum cleaner bags, other combustibles	Cutting mill (Retch SM 2000, Haan, D)	Riffle splitter (Rationel Kornservice RK12, Esbjerg, DK)	5 ml HNO ₃ + 0.5 ml H ₂ O ₂ +0.03ml HF	Microwave-assisted digestion, 600W, Temperature 140°C, P < 1400 KPa , 1h
All types of plastics, shoes, leather, rubber	Cutting mill (Retch SM 2000, Haan, D)	Riffle splitter (Rationel Kornservice RK12, Esbjerg, DK)	5 ml HNO ₃ + 0.03ml HF	Microwave-assisted digestion, 1200W, Temperature 180°C, P < 5200 KPa , 1h
Glass, soils, ceramics, other non-combustibles	Rotary disc mill (Siebtechnik IS100A, Mülheim an der Ruhr, D)	Riffle splitter (Rationel Kornservice RK12, Esbjerg, DK)	0.2 g LiBO ₂	C crucible fusion, Temperature 1050°C, 45 min, dissolution of the bead in 0.7 M HNO ₃

The remaining heavy metals that do not necessarily originate from inert fractions are: *Cr*, *As*, *Cu* and *Zn*. *Cr* in the waste originates mainly from “Shoes & leather”, “Green glass” and “Textiles” all together these fractions account for 81 % of the *Cr* in the waste. Waste content of *As*, *Cu* and *Zn* are for reasons that will be described in chapter 3.5 not allocated to any specific material fractions.

Riber et al. IV also concludes that 30 % of the carbon is fossil and that most of the heating value originates from biogenic carbon fractions. The fraction “Non recyclable Plastic” contains nearly all the PVC of the waste and 60 % of the waste *Cl* content but also 21 % of the heating value.

Compared to the German study (Rotter, S., 2001) the concentrations found and their distribution between fractions are in correspondence. It is concluded that the sampling method of the German study leads to a wider spread of the heavy metal between the fractions than the one used in this study. The origin of the *Cd* content is not the same in the German study, where the majority of *Cd* is found in hard PVC and in this study is mainly found in batteries. No explanation for this difference has been found.

3.2 Indirect Analysis

Unlike direct analysis, the indirect analysis of solid waste is used to analyze products of a process in which the matrix of the waste can be alternated. Processes may vary, but will usually involve combustion where evaporation of water and oxidation of the carbon results in up to 80 % loss of mass and corresponding volume. The mass and volume reduction allows for large primary samples and produces more homogeneous products for secondary sampling. Waste incinerators are used mainly because they are designed to treat large masses of waste with a wide range of physical properties. While the main purpose of the indirect analysis is to identify waste chemical composition, the method also makes it possible to determine fuel characteristics and changes in transfer coefficients at the incinerator.

In the literature indirect analysis has been used to identify chemical composition of more or less well defined waste types but also to conclude the resulting transfer of substances considering a given waste and incinerator. The method was first introduced scientifically as a waste characterization method by Brunner et al., 1986. Both Belevi et al. (2000) and Morf et al. (2000) work with the complex relations of waste composition, incinerator operational conditions and the transfer coefficients of substances (TFC). But the indirect method has also been used to conclude on waste chemical content in e.g. Schemayer et al. (1995), Belevi (1998) and Bliksbjerg et al. (1994). The study by Terkildesen et al. (1994) focuses only on a very specific waste type (CCA treated wood) and does not attempt to determine a waste chemical composition. Terkildesen et al. (1994) only concludes on the change in TFC and output contaminant concentrations related to the waste type.

The methodology of indirect analysis is described in detail in Riber et al. Vand, the main points of which will be discussed in the following. The main prerequisite for a indirect analysis is a waste incineration plant that fits the given analysis in terms of size and type of air pollution control (APC) system. The amount of waste that is available for the analysis may limit the choice of incinerator size and the accessibility of output products may also pose a problem. It is of great importance that sampling of the output products is in correspondence with the theory of sampling by Pierre Gy, (1998), as incorrect sampling will otherwise bias the final result significantly. When sampling output products, it is important to take the retention time within the system into account to ensure that all samples can be related to the same test waste. As a minimum, the retention time must be quantified for the bottom ash, as this usually holds the longest retention time. The system must have been consuming the test waste for at least two times the average retention time of the bottom ash to avoid contamination with the previous combusted waste. In addition to taking samples of output products, a series of data must also be collected during the test, to enable the calculation of mass, substance and energy balances. This process is further described in V.

The waste chemical composition can be calculated through simple mass and substance balance, and the water content of the waste is calculated based on an assumption of the waste C/H content relation, as described in Riber et al. V. Calculations of the heating value can be carried out on the basis of energy production data, heat losses in the flue gas and an estimated convective loss of furnace and boiler. Riber et al. V argues that the heating value of solid waste is best reported as the lower heating value of the water and ash free waste (LHV_{waf}), as the LHV on wet waste overestimates the importance of the water to the energy content. Riber et al. V furthermore argues that the incinerator efficiency is best reported as the sold energy

referred to the higher heating value as this is usable also for the new condensation technologies.

In addition to the two incinerators described in Riber et al. V, this study has also includes a third incinerator used in an indirect analysis. The following presents a brief description of the characteristics of the third incinerator in Herning, Denmark. In contrast to/unlike the first two incinerator lines used for indirect analysis, the Herning incinerator is still in operation and fulfills all present legislative requirements for air emission. The grate is air cooled and the furnace capacity is seven tonne pr. hour. The APC system first has a dust filter, then an acid reactor, followed by a semidry reactor, including activated carbon addition and a bag house filter. At the end of the process, a flue gas condensation unit removes some of the steam and produces hot water for the local district heating system. The main boiler is equipped with a natural gas fired super heater and produces both electricity and heat for district heating.

In Aarhus, household waste was analyzed, and in Taastrup, both household waste and small combustible waste from recycling stations were analyzed. The household wastes were collected in different container sizes at the households and the small combustible wastes were tested in both spring and fall. The Herning test was performed on household waste from Herning municipality. The collection of a specific material fraction called “Problematic waste” at the household is unique for Herning. Besides providing the chemical composition of the household waste in Herning the test also evaluated whether the removal of “Problematic waste” from residual household waste incinerated had a significant effect on the chemical composition. An overview of the tests performed is presented in Table 4 including information on where the tests have been performed, the waste type used and amount etc.

Table 4 Indirect analysis performed from 2003-2006 and information on waste types and incinerator mass distribution / Riber et al. V/.

Waste type	Small combustible		MSW	Household waste			
	Spring	Fall		Aarhus	Aarhus	Odense	Cph.
Waste origin	Taastrup	Taastrup	Aarhus	Aarhus	Taastrup	Taastrup	Herning
Incinerator							
Waste input (tones dry)	84.6	60.4	856	785	50.5	109	48.2
Water content (wet waste)	16.7 %	20.6 %	28.4 %	32.8 %	22.2%	29.0 %	20.3 %
Fly ash (dry waste)	2.53 %	3.48 %	1.80 %	2.46 %	3.83 %	2.29 %	2.61 %
Bottom ash (dry waste)	20.7 %	24.0 %	21.9 %	20.1 %	24.1 %	22.0 %	16.4 %
LHV _{waf} (kJ/g)	21.6	24.5	20.0	16.7	16.6	15.3	16.7

Cph. abbreviates Copenhagen. Brackets denote the reference mass.

The conclusions on the indirect analysis on household waste are that household waste chemical composition is not a function of the collection container size. On the other hand, the specific collection of problematic waste at source lowers the concentrations of key heavy metals significantly, as shown in Figure 1. The small combustible waste from recycling stations has significantly higher heavy metal concentrations compared to the household waste as shown in Figure 2.

Household waste content of *Hg* is a function of three occurrences in the waste, as stated in Riber et al. V; background concentrations in multiple fractions (0-1 mg/kg), rare items like button cell batteries (1-2 mg/kg) and very rare items of elemental *Hg* (2-4 mg/kg). Examples of very rare items could be thermometers and similar objects. Such

examples occurred once per app. 800 tonnes with an *Hg* emission in the range of 40-800 g pr incidence. The occurrences were seen as peaks of *Hg* in the emitted flue gas at the Taastrup incinerator, as this was equipped with continuous *Hg* emission measurements, as can be seen in Figure 3. Up to half of the average *Hg* content in the household waste could be represented by very rare items in the waste.

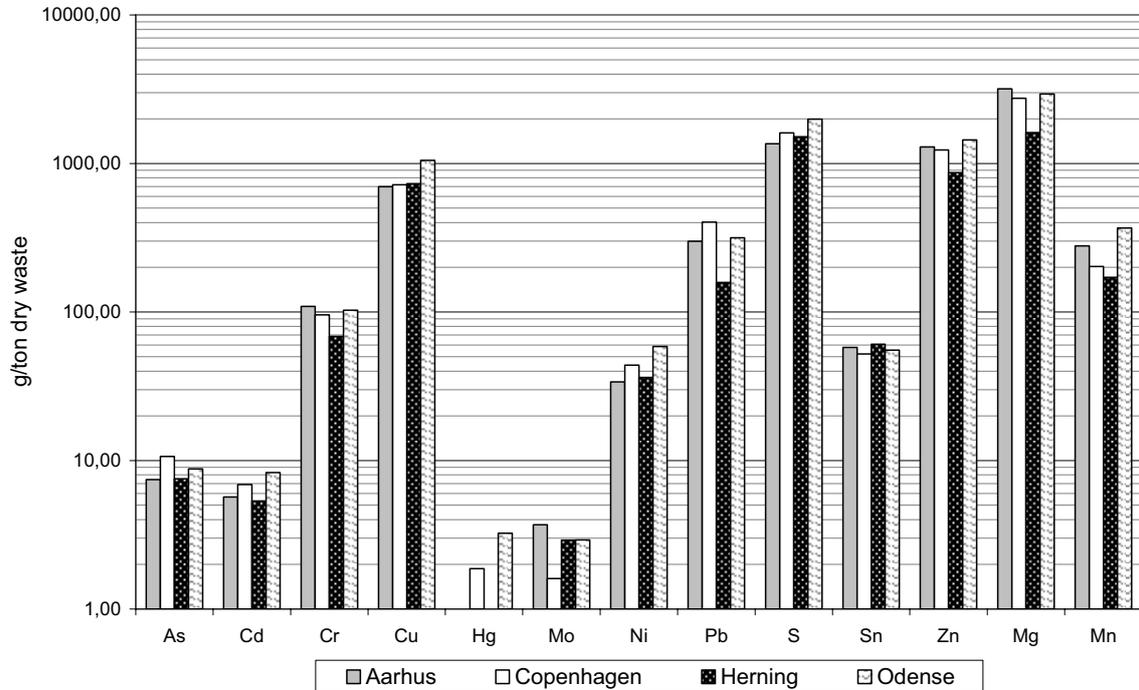


Figure 1 Comparison of dry waste content of some contaminants in four Danish collection schemes.

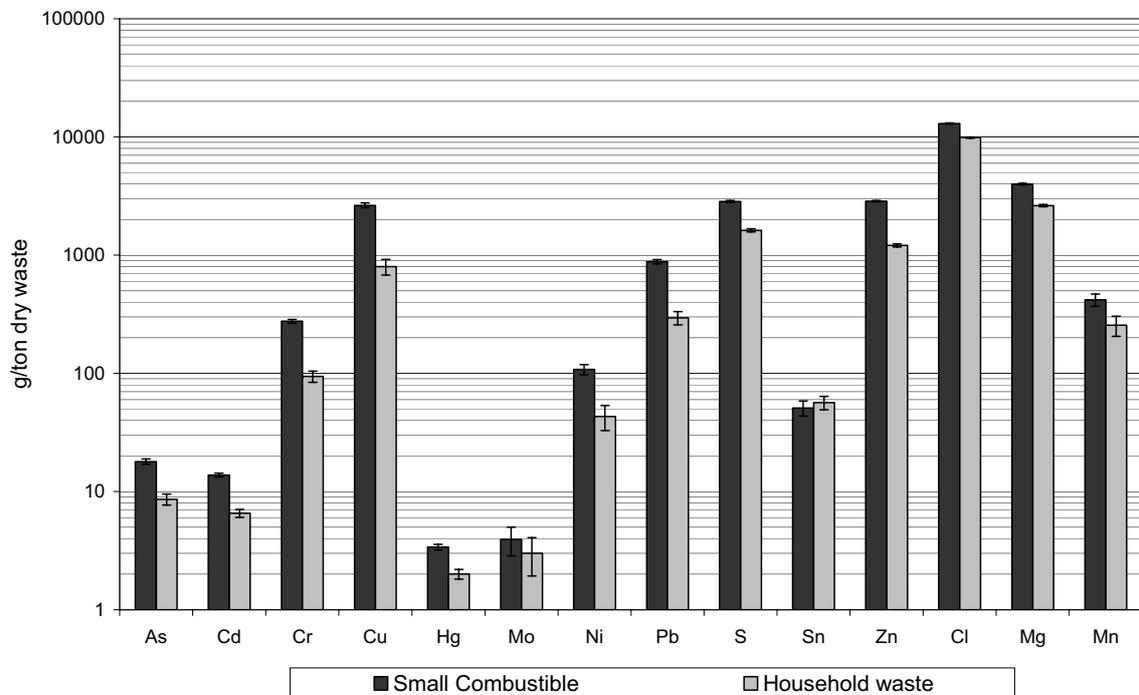


Figure 2 Comparison of average Danish residual household waste content of some contaminants measured by indirect analysis and combustible waste less than one meter (Small) from recycling stations / Riber et al. V/.

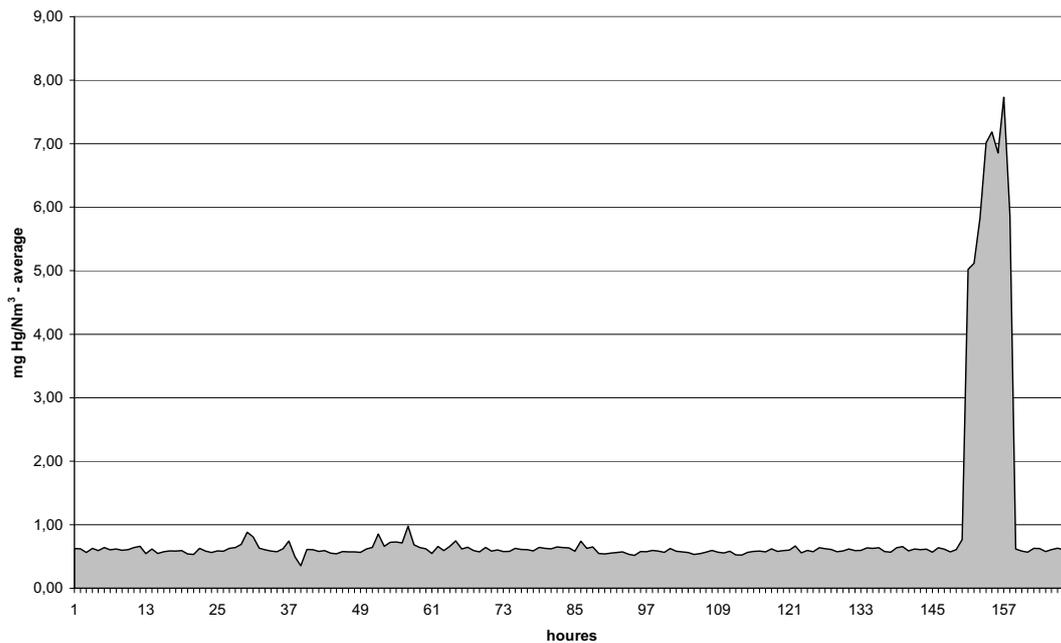


Figure 3 Emission of Hg pr normalized cubic meter (Nm³) at the Taastrup incinerator measured continuously and expressed as averages emission pr. hour / Riber et al. V/.

3.3 Fuel Characteristics

Fuel characteristics are defined as the physical properties of the fuel (waste) that will affect the combustion process and energy production of a waste-to-energy plant. The four indirect analyses of household wastes and two indirect analyses of small combustibles made it possible to define fuel characteristics for the two waste types. In Table 5 the most important parameters are shown and the following can be concluded in comparison to MSW:

Household waste

- Low LHV_{waf} (15.3-16.7 GJ/tonne) and high water content (22-33%)
- Delays dryout, ignition and burnout in the furnace
- Potential CO emission and bottom ash TOC

Small combustible waste

- High LHV_{waf} (22-25 GJ/tonne) and low water content (17-21 %)
- Speeds up dryout, ignition and burnout in the furnace
- Potential CO emission

Riber et al. V concludes that the PVC content of the household waste is the most important reason for variations in LHV and Cl content.

3.4 Transfer Coefficients

Transfer coefficients (TFCs) are the relation between the input mass of a substance and the different output masses. The calculations of the coefficients will therefore be affected by both mass, substance and energy balance. Transfer coefficients are a common estimate of incineration performance and have been used by for instance Belevi (1995) and Terkildsen et al. (1994) to evaluate the effect of burning different waste types.

Belevi et al. (2000) and Morf et al. (2000) both state that transfer coefficients are dependant on waste content. Morf et al. concludes that the average TFC is mainly a function of substance physical-chemical properties and only variations are related to waste input concentration. Riber et al. V concludes that TFCs are significantly influenced by the waste type combusted shown by indirect analyses of two waste types at the Taastrup incinerator. From the four tests at the Taastrup incinerator the transfer to bottom ash is shown in Figure 4 for a number of selected substances. The TFC of mixed wastes as MSW are therefore a result of the different TFCs of the waste types mixed and will vary with the mixture. The average TFC is therefore both a function of substance physical-chemical properties and the occurrence of rare items and concentrations in the mixture of waste.

It is widely recognized [Morf et al. (2000), Bliksbjerg (1995)] that incinerator technology and operational conditions are important factors to the resulting TFCs. To exemplify this transfer of *Hg* to outputs are shown in Figure 5. The first and second transfers in Figure 5 are from the test in Taastrup with household waste and small combustible waste respectively. The third transfer in Figure 5 is from Aarhus and the fourth is from Herning, and they both represent household waste. Figure 5 clearly shows that the TFCs are influenced by waste type, operational conditions and incinerator technology.

Table 5 Correction of the result of direct characterization of household waste in order to obtain correspondence to the total concentrations determined by the indirect characterization. The “indirect” data are averages of three independent determinations / Riber et al. IV/.

	Direct	Indirect	Difference	Relative difference	Corrected fraction
As	2.7	9.0	6.2	231%	Residual
Cd	13.1	7.0	-6.1	-47%	Batteries
Cu	173	822	649	375%	Residual
Hg	0.07	2.56	2.48	3380%	Batteries
Mo	12.3	2.7	-9.5	-78%	Non combustibles
Ni	191	45	-145	-76%	Batteries
Pb	1650	340	-1310	-79%	Other metal
Zn	905	1320	417	46%	Residual
Cl	5780	8440	2654	46%	Non Recyclable Plastic
Mn	730	283	-447	-61%	Batteries

Unit: g/ton dry waste.

Relative standard deviations on results of the three indirect measurements were in average 22%.

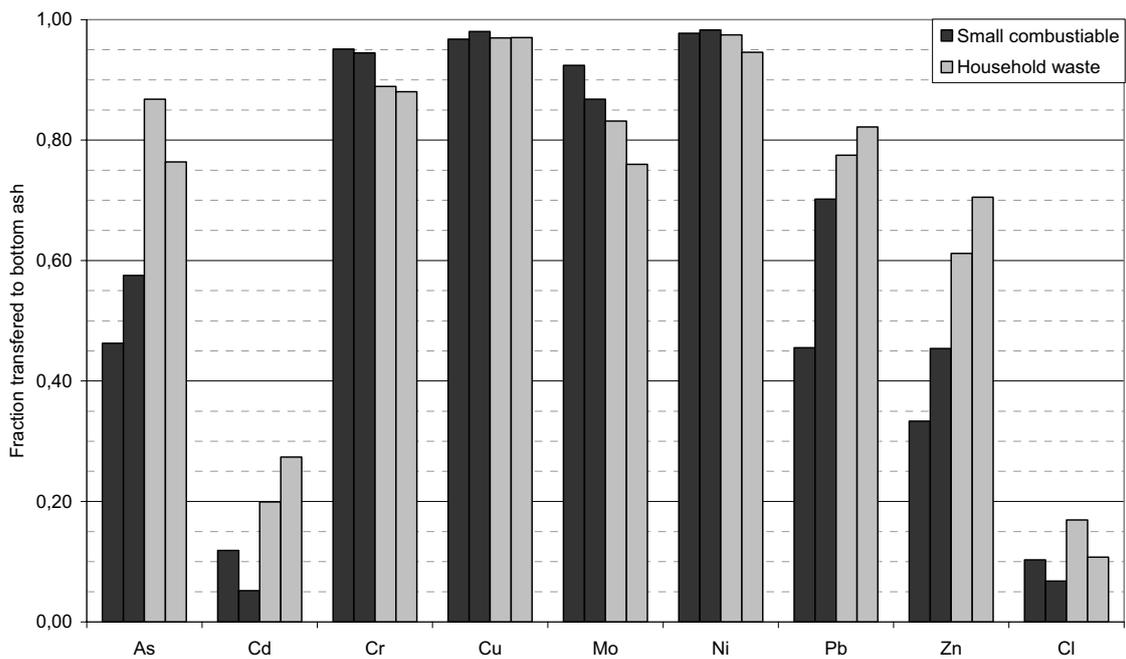
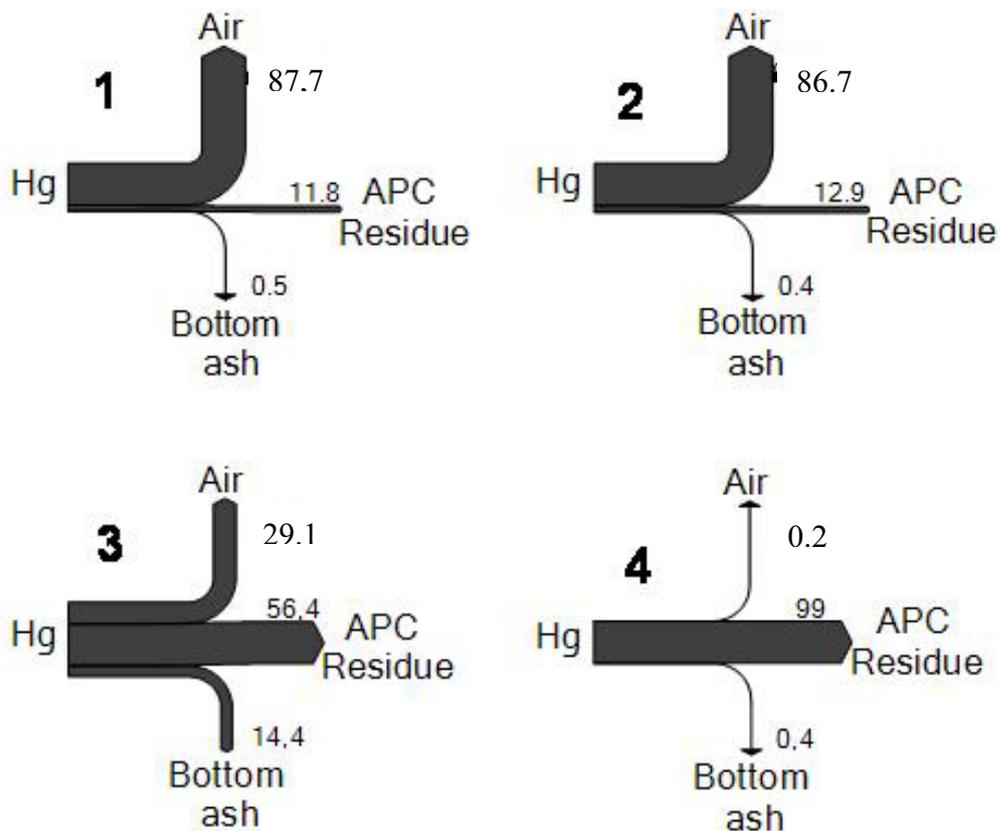


Figure 4 Comparison of transfer of some substances to bottom ash at the Taastrup, Denmark



incinerator when burning two different waste types /Riber et al. V/.

Figure 5 Transfer of Hg in four cases;

- 1) household waste in the Taastrup incinerator
- 2) small combustible waste in Taastrup
- 3) household waste in the Aarhus incinerator

4) household waste in the Herning incinerator.

3.5 Corrected Direct Analysis

As the indirect analysis includes 10-100 times more waste than the direct analysis it is assumed to be more representative regarding the total content of the waste. Therefore it is relevant to correct the content of substances in the direct analysis to correspond the results of the indirect analysis. Because of variance in results of the indirect analysis, only differences greater than 20 % are subject to corrections and the substances corrected can be seen in Table 5, together with the difference corrected. Corrections are defined as either missing content (direct < indirect) or excessive content (direct > indirect). If a single fraction dominates the distribution of a substance in the direct analysis the correction is made here. If this is not the case and other arguments can not be found to place the correction it is made in the “Residual” fraction indicating the missing relation to a material fraction. The arguments for the performed corrections from Riber et al. IV are summarized here:

Corrections for missing content

- *Cl* was corrected in “Non recyclable plastic” because the analytical method was not complete for the organic bound *Cl* in this fraction, as reported by Rotter (2005).
- *Hg* was added to the “Batteries”, as this was where the majority of *Hg* was found, and because batteries were expected to be the main source of *Hg* in the waste.
- *As*, *Cu* and *Zn* were not dominated by one fraction and therefore the addition was done in the “Residual” fraction.

Corrections for excessive content

- *Cd*, *Ni* and *Mn* were reduced in “Batteries” as this was the major source of the elements.
- *Pb* was found mainly in the fraction “Other of metal” and the subtraction was done here.
- *Mo* was subtracted in the fraction “Other non combustible” that contains the major part of *Mo*.

Riber et al. IV concludes that because of the major correction of *Hg* and other elements in trace concentrations, it is not advisable to conclude on waste concentrations of trace elements only based on direct measurements. The corrected chemical composition of residual Danish household waste distributed on 48 material fractions and as total content is shown in Riber et al. IV.

4 LCA modelling of Waste Incineration

Life Cycle Assessment (LCA) is recognized in the literature to be useful in five ways /Wenisch et al, 2004/:

- Identification of important impacts in a pre-design phase
- Comparison between alternative designs in a design decision phase
- Ranking present impacts in prioritizing optimization of the existing process
- Evaluation of the process compared to a reference in a management phase
- Benchmarking performance against existing or new alternative processes in a future optimization perspective

Modeling of the environmental impact from waste incineration is best performed by a full life cycle assessment as this includes all known impacts on the environment on a regional to global scale. In recent years, this argument has been widely recognized by many authors, such as Morselli et al. (2005); Chevalier et al. (2003); Hellweg et al. (2001) and Eriksson et al. (2007). In addition to direct emission by the incinerator, impacts from processes (products, energy or raw materials) consumed by the incinerator, or processes that are saved as a direct consequence of the use of the incinerator, are also included. The LCA could also include construction and demolition phase of the incinerator, but these phases are often regarded insignificant when taking into account the sum of impacts generated at the incinerator. Not only does the incinerator impact environment as a result of the combustion of the waste and cleaning of flue gases, it also produces energy that would otherwise have been produced by other process. All together the implied savings are called substitutions in the LCA nomenclature and are accounted for as the (negative) emissions from the saved process /Moselli et al., 2005/.

The use of LCA in evaluating the environmental impact associated with incineration of one tonne of waste requires a complete inventory of the interactions between the surrounding environment and the incinerator. As mentioned above these interactions include emissions as well as products, raw material or energy consumed. However, it is also necessary to gain an understanding of the transfer of substances within the incinerator and knowledge of the chemical composition of the waste. Emissions from the incinerator are, as mentioned above, categorized according to what controls the emission: process or waste. In the inventory of an incinerator the process specific emission are given as relative to input mass, where waste specific emissions are presented as a transfer coefficient. In reality this is a simplification, as many emissions are both controlled by process and waste content. However, for the sake of simplicity, emissions will in the following be regarded as depending only on one factor for simplicity reasons. The methodology and software described later in this thesis, are, however, also designed to cover mixed process and waste specific emissions for the same substance.

4.1 Methodology

For computation of environmental assessments of waste management systems, the software EASEWASTE has been developed, and in a sub-module of this software an incinerator can be assessed. The model assists in converting data of consumption, production, substance transfers and process emissions into an inventory of emissions /Kirkeby et al., 2006/. Other models exist that could possibly do the same, and some literature describes the application of for instance TEMP (Ecobilan, France); Morselli et al (2005) and implementations in SimaPro; Eriksson et al. (2007) and Schmidt et al. (2007). Previous examples of the purposes of modeling waste incineration has been: comparison of district heating alternatives by Eriksson et al. (2007); comparison of waste treatment options by Schmidt et al. (2007) and evaluation of APC technology by Chevalier et al. (2003).

The calculations performed by the EASEWASTE incineration sub-model are given in detail in Riber et al. III but falls in the following categories:

- Output mass calculation relying on ash transfer coefficient and corrected for water content.
- Air emissions (waste specific) calculated by transfer coefficients.
- Energy production calculated by the waste energy content, an energy production coefficient and the saved process emissions.
- Output chemical composition calculated by the substance transfer coefficients and by the output mass.
- Emissions to all environmental compartments⁶ (process specific) calculated as the given rate of emission and the specified emissions profile.
- Emissions from used processes or raw materials calculated by the given consumption rate and the emissions inventory of the process.

The methodology of life cycle impact assessments is based on the transformation of all the emissions of the life cycle inventory for one functional unit to only a limited number of impact categories. The emissions are converted into a common unit in each category by an emission impact factor matrix (F_e), as described in Kirkeby et al. 2006 and Wenzel et al., 1997, and are schematically shown in Figure 6. The categorized emissions in the impact categories are normalized by the total environmental load of all activities in society for an average person (F_p) /Wenzel et al. 1997/. The main purpose of this step is to make the sum of emissions in each impact category comparable to the sum of other categories.

The emission factors defining the conversion from emissions to impact categories follows the EDIP97 methodologies described in e.g. Kirkeby et al., 2006 and Wenzel et al., 1997 where also the person equivalent is described further. Person equivalents (abbreviated PE) are the unit of the normalized impacts.

⁶ Air, soil, surface -, ground - or sea water.

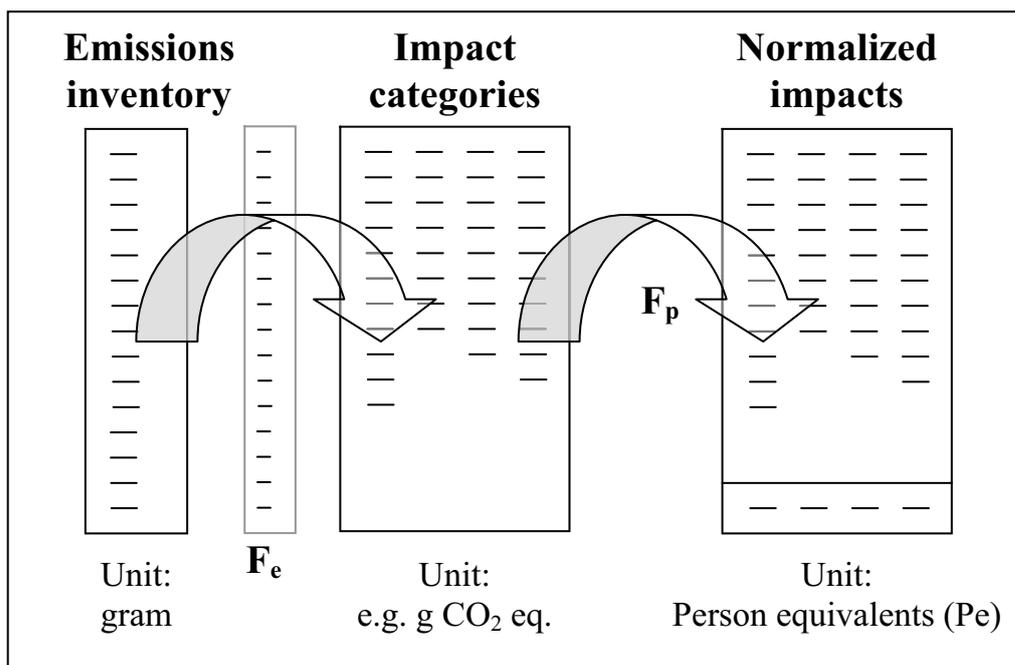


Figure 6: Conversion of emissions to potential environmental impacts in relation to a person equivalent.

4.2 Cases Modeled

To illustrate the possible outcome of LCA on waste incineration, three cases covering three aspects of implementation of such analysis are presented in the following:

1. Technology comparison between the present Aarhus incinerator and the Herning Incinerator with flue gas condensation
2. The consequence of the recent modernization of the Aarhus incinerator and of a further energy production optimization.
3. Comparison of treatment by incineration at the Aarhus incinerator of two types of waste.

The three cases independently cover one of the ways in which it would be advantageous to implement LCA, as earlier described (design phase, management phase and optimization).

Case 1 Technology comparison

The technologies of the Herning and Aarhus incinerators were described above under indirect analysis and they are both included in EASEWASTE with their inventories and substance transfers. This case study involves only treatment of residual household waste and the corrected chemical composition on 48 material fractions were used in the modeling. The main purpose was to evaluate incinerator performance and identify the best choice of technology and the important factors for optimization. The Aarhus incinerator was modeled in 2006 and the Herning incinerator in 2005, and the result as impacts in person equivalents per category are given in Figure 7. The Aarhus incinerator is located in a power and district heating grid together with a coal fired power plant that is substituted, whereby fossil CO₂ emission is saved. As mentioned above, the Herning incinerator is located in a district heating grid together with a 50/50 biomass and natural gas fired power plant. The fact that a deficit in grid power arises and that half of the energy production (from biomass) will not result in saved impact, leads to the positive impact on global warming by the Herning incinerator. The nutrient enrichment and acidification categories are not affected to the same degree, and the two incinerators perform almost equal. The Herning incinerator performs best in the toxic categories as seen in Figure 7. There are two main reasons for this: firstly (and most importantly), almost no *Hg* is emitted due to flue gas condensation. Secondly substitution of the biomass fired power production saves dioxin and *Cd* emissions. Both plants have advantages and drawbacks and the weighting of impact categories will affect which plant is assessed to have the best environmental profile.

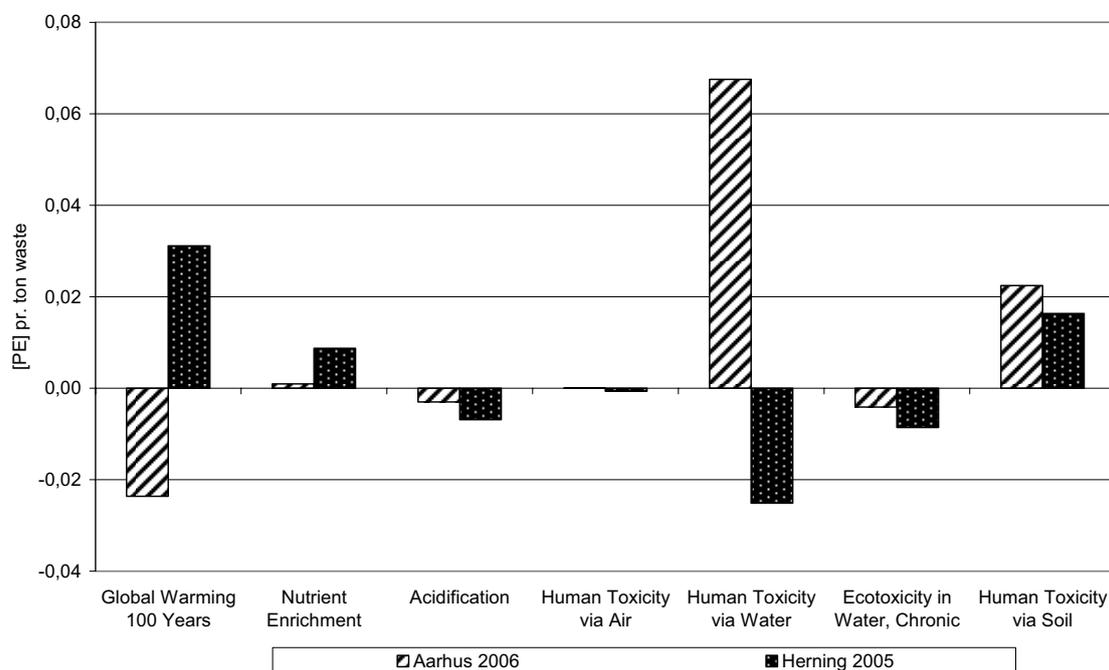


Figure 7 Impacts to environment based on LCA on two Danish incinerators burning residual household waste

Case 2 Consequence of modernization

In the years 2003-2006, the Aarhus incinerator was subject to a technology modernization preparing the incinerator to meet new emission limits and optimize energy production. The incinerator had one line demolished and a new and bigger one built with more efficient energy utilization and state-of-the-art wet APC technology. The wet APC systems in the two remaining lines were rebuilt into semidry systems with a new reactor technology, but no optimization of energy production. In addition to the modernization of the Aarhus incinerator, an energy production optimization scenario on the two old lines was also modeled. In the scenario called Aarhus** it was assumed that the energy production efficiency would be comparable to the new line and in all other aspects identical to the Aarhus 2006 incinerator. In Figure 8 the impacts in person equivalents per category are shown for both toxic and non toxic impact categories. The modernization introduced in the incinerator has converted environmental burdens in all the non toxic categories into savings and a further optimization in scenario Aarhus** is seen to be very beneficial. As shown in Figure 8, the toxic categories have also become lower by the modernization. Human toxicity via water and soil has decreased in particular. This is mainly caused by the reduction of *Hg* emission by introduction of a dioxin removal unit, but further energy optimization does not seem to have a significant effect.

The optimization of the Aarhus incinerator gave major savings in potential environmental impacts. Non toxic impacts were lowered as a result of energy optimization and toxic categories were lowered due to less *Hg* emission. However, there still seems to be potential for optimization in both category groups.

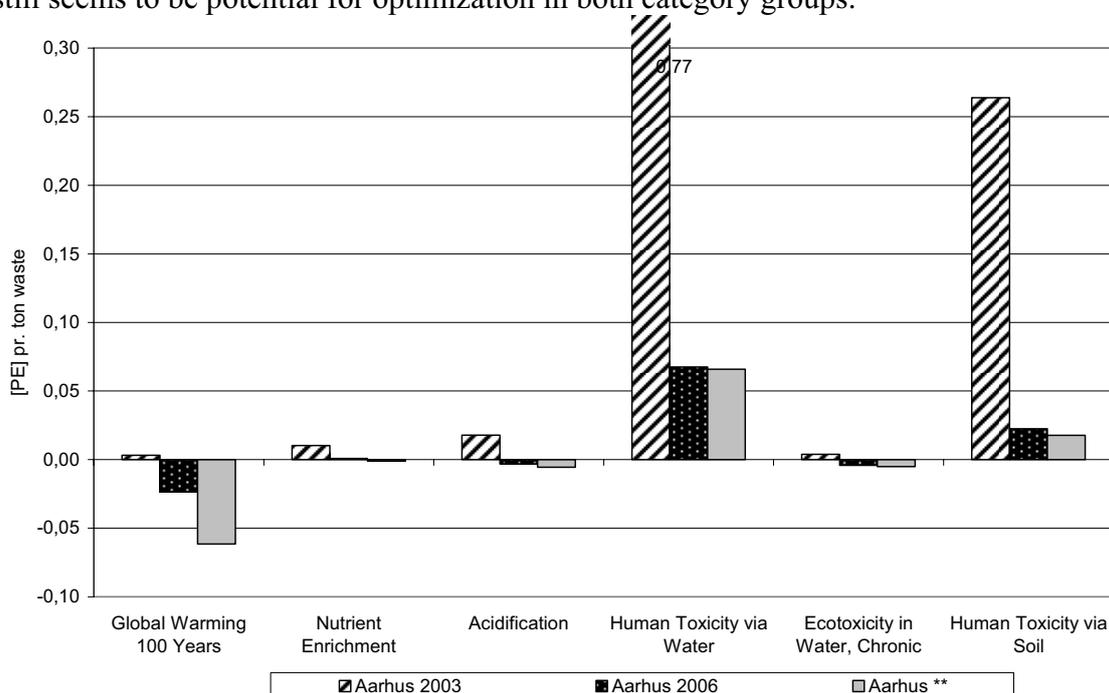


Figure 8 Impacts to some environment from LCA on the modernization of the Danish incinerator in Aarhus burning residual household waste

Case 3 Waste type influence

In the Aarhus incinerator, both residual household waste and municipal solid waste (MSW) were tested and the chemical composition and physical properties are used in this case study. Additionally, the MSW was modeled with double heavy metal content, simulating a contamination or a reduced cleaning in the APC of the incinerator. For simplicity reasons the incinerator transfer coefficients are kept constant for all three waste types. The inventory and substance transfer for the Aarhus 2006 incinerator (measured with MSW) is used and the results for the toxic categories are given in Figure 9. The non toxic categories gave an unchanged impact for two mixed wastes but a 9 % decreased impact on global warming by the residual household waste caused by lower energy and higher water content. Unsurprisingly, the toxicity of the MSW with double heavy metal content is almost the double for the human toxicity via water and soil compared with normal MSW, whereas the ecotoxicity remains almost unchanged. One result that was unexpected was that the incineration of household waste showed a 5 times higher impact on human toxicity via water and soil (compared to MSW). The reason for this is the chemical composition of the waste. The *Hg* content of residual household waste was 4 times the one in MSW, and furthermore the lower heating value (LHV) gave less substitution which corresponds with the large increase in impact. The *Hg* content and LHV of a waste type are concluded to be very important for the environmental burden associated with incinerating the waste in Aarhus.

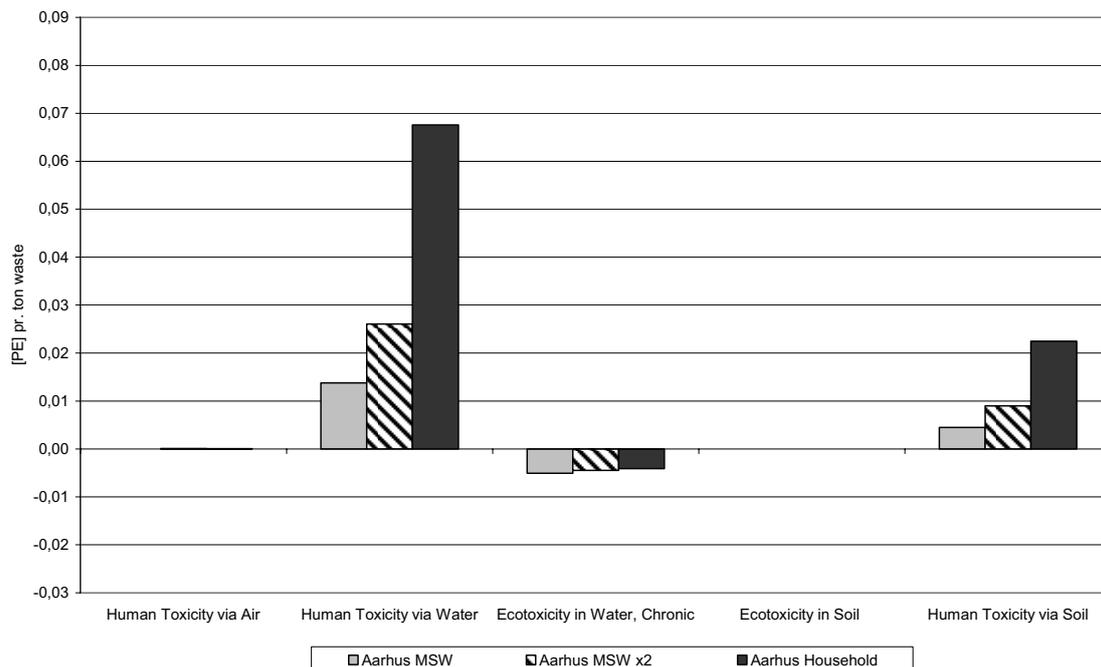


Figure 9 Impacts to toxic categories from LCA on three waste types incinerated at the Aarhus incinerator, Denmark: Municipal solid waste (MSW), MSW with double amount of heavy metals (MSWx2) and residual household waste from corrected direct analysis (Household).

4.3 Modeled Waste Specific Impacts

The Aarhus 2006 incinerator from case study no. 2 also described in Riber et al. III was used to evaluate the influence on the overall incinerator environmental performance of waste specific impacts. Furthermore the essential waste specific were investigated to determine which environmental compartment was responsible for the given impact. The environmental compartment was then investigated to determine which single material fractions are responsible. Last the material fraction was investigated to state which substance was responsible. This kind of investigation on results of an LCA case study will in the following be referred to as “two way” analysis. The term describes the fact that the LCA is used both ways; first a modeling where input data results in an assessment; secondly a investigation of which of the input parameters gave the important impact in the assessment.

The relative contributions from processes or waste to the impact categories of the Aarhus incinerator are shown in Figure 10. The figure indicates that five categories are affected by the waste specific emissions, and in three of the cases, the impact categories are significantly affected: Global warming, Human toxicity via water and Human toxicity via soil. The impact on global warming is solely caused by air emissions of CO₂ originating primarily from the fossil C content of the waste material fraction “Non-recyclable plastic”. The relative contributions by single substances to the toxic impacts categories are shown in Figure 11. As Dioxin and NO_x are not waste specific they have therefore been disregarded here. The impact on both human toxicity via water and soil are caused by air emissions of Hg originating primarily from the waste material fraction “Batteries”. The two remaining impact categories with a significant effect are Human toxicity via air caused mainly by Pb from the material fraction “Other of metal” and ecotoxicity in water caused by Hg in “Batteries”.

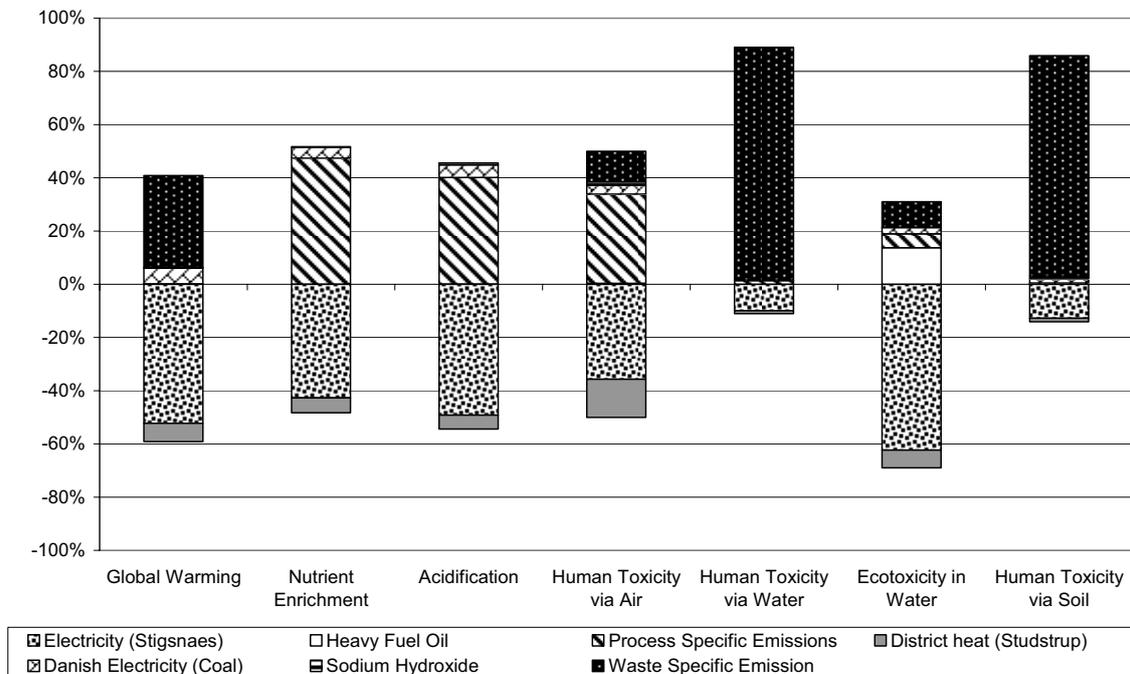


Figure 10 Relative contributions to some environmental impact categories from processes and waste combustion at the Aarhus incinerator, Denmark /Riber et al. III/.

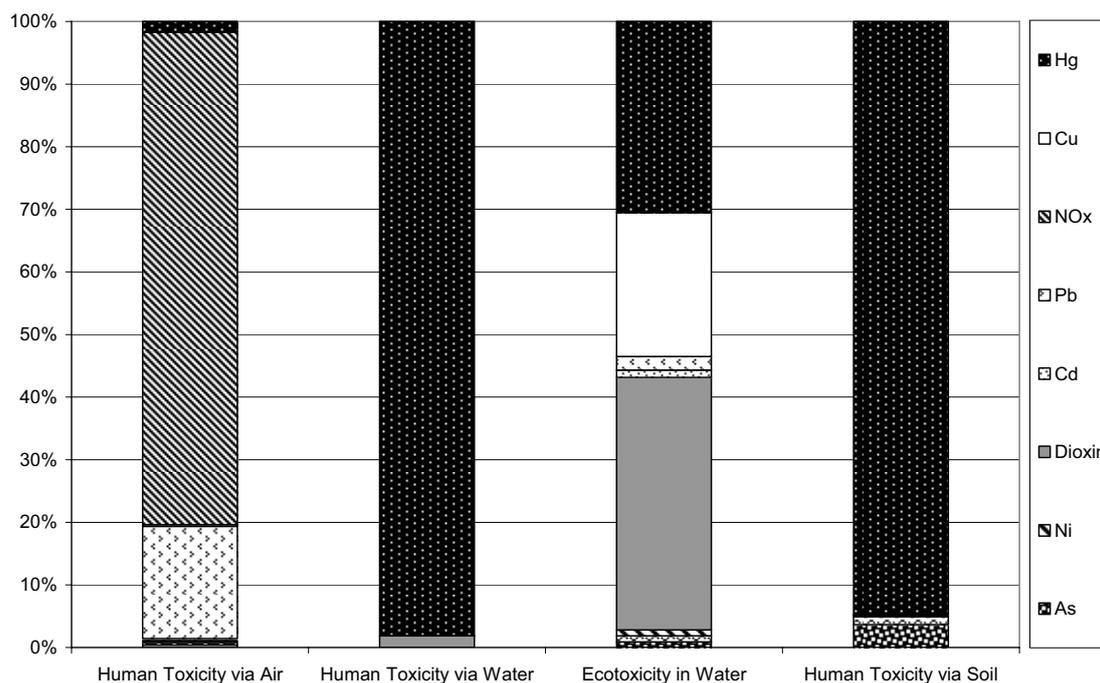


Figure 11 Impacts to some toxic categories from single substances at the Aarhus incinerator, Denmark

The negative relative contributions in Figure 10 are the result of substitution of energy from the coal fired power plant as earlier described. The substitution is a result of the power and district heat production at the incinerator and the productions are a function of technology efficiency and energy content of the waste. The substitution impact is therefore both process specific, as the outcome relies on technology efficiency, and waste specific, as it will change when waste energy content is altered. In the following, the substitution is treated as a waste specific impact. Altogether, the impact of the substitution processes at the Aarhus incinerator amounts to $-1.6E-1$ PE, counting all single impacts together.

The importance of single waste specific parameters can be evaluated by their summated impact across categories, hereby weighing categories equal, as this expresses their cumulative potential impact. The most important toxic impacts are listed in Table 6, where also the cross category sum has been stated. The non toxic categories have not been included, as only one waste specific substance in one category was present, and fossil C contributes to global warming with $8.6E-3$ PE. Table 6 reveals that *Hg* is the single most important substance to the toxic impact categories. If process specific impacts are disregarded (in italics in Table 6), *Hg* appears as more than 100 times more significant than second most important waste specific impact. *As*, *Cr*, *Cu* and *Pb* are also significant, relative to the total impact, although not as much as some of the process specific impacts (*NO_x* and Dioxin).

In conclusion, the waste specific impacts from incineration of waste can be ranked according to their total potential impacts and thereby their importance to the environmental profile.

1. Waste energy content (LHV_{waf})
2. Hg content (mainly found in the “Batteries” material fraction)
3. Fossil C content (mainly found in the “Non recyclable plastic” material fraction)
4. As and Cu content (material fraction not determined)
5. Cr and Pb (inert material fractions and leather)

Table 6 Important potential impacts to four toxic impact categories and the non weighted sum from LCA on incineration of one tonne of household waste at the Aarhus incinerator.

Substance Name	Human Tox. via Air	Human Tox. via Water	Ecotox. in Water	Human Tox. via Soil	Sum of Tox.
Mercury (Hg)	3.9E-06	7.8E-02	4.1E-04	2.3E-02	1.0E-01
<i>Nitrogen Oxides (NO_x)</i>	<i>1.8E-04</i>	<i>1.9E-07</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>2.7E-02</i>
<i>Dioxin</i>	<i>1.6E-07</i>	<i>1.5E-03</i>	<i>5.4E-04</i>	<i>3.9E-05</i>	<i>2.1E-03</i>
Arsenic (As)	8.8E-07	1.6E-07	1.2E-05	8.8E-04	8.9E-04
Chromium (Cr)	5.7E-07	2.5E-06	1.3E-05	3.1E-04	3.2E-04
Copper (Cu)	4.0E-10	2.9E-06	3.1E-04	1.4E-06	3.1E-04
Lead (Pb)	4.2E-05	2.7E-05	2.9E-05	1.7E-05	1.1E-04
Nickel (Ni)	3.6E-08	2.5E-09	1.3E-05	3.1E-05	4.4E-05
Cadmium (Cd)	4.5E-07	2.7E-06	1.6E-05	8.4E-06	2.7E-05
<i>Carbon Monoxide (CO)</i>	<i>9.0E-07</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>9.0E-07</i>
<i>Unspecified Particles</i>	<i>6.7E-07</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>6.7E-07</i>
<i>Sulphur Dioxide (SO₂)</i>	<i>3.6E-07</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>0.0E+00</i>	<i>3.6E-07</i>

Unit PE (Person Equivalent)/ton. Process specific impacts are in italic.

LHV_{waf}

The importance of the LHV as a waste specific impact is as previously described a subject for discussion, and it’s importance in relation to the total impact depends on both technology and location of the incinerator in the energy production grids. This is addressed in case study one where the energy efficient Herning incinerator has a substitution process, which results in a much larger global warming impact than the Aarhus incinerator. Comparing impacts internationally will create problems like described above, as the opportunity to sell energy is not always the same.

It is important to keep in mind that the waste energy content is expressed as LHV_{waf} and thereby a function of both heating value of the waste and the water content. The influence of waste energy content on the impacts from energy production is limited by the possible and realistic changes in the heating value of the waste. Therefore, the heating value should not be set as the most important parameter.

Mercury

The Hg content was described in chapter 3.2 and it was concluded that up to half of the Hg content could be present as rare items with high concentrations. It could be expected that Hg in high concentrations will not distribute as the Hg in background concentrations, as the controlling process of APC is sorption. Firstly, sorption capacity could be exhausted, resulting in no sorption and secondly, concentrations could be so

high that sorption reaction will not have time to occur. High concentration of one analyte could also theoretically influence sorption equilibrium resulting in reemission of other already sorbed substances. The analysis of the Taastrup incinerator showed that without a dedicated APC system for *Hg*, only 40 % of *Hg* is retained by simple condensation. This suggests that with a reduction of sorption capacity to zero, 60 % emission is to be expected. If half of the waste content emits 60 % the emitted amount of *Hg* at the Aarhus incinerator would be underestimated by app. 300 % if emissions are only based on measurement excluding the very rare incidences. However, comparing this with the evaluation of transfers of the Herning incinerator, as illustrated in Figure 5, could give a different result. The Herning incinerator removes 99.8 % of the *Hg* content because of both activated carbon and flue gas condensation (leading to *Hg* condensation caused by the temperature drop). The emissions from the Herning incinerator may not be affected by rare items with high *Hg* concentration to the same degree, as the condensation unit may act as an extra filter, even though sorption capacity is used.

One way to evaluate *Hg* emissions correctly when modelling in the future could be to define a new material fraction containing the part of the *Hg* content in the waste present in concentrations so high that they will affect the efficiency of the APC. This fraction should then be given a TFC reflecting a decreased efficiency of the APC. This is possible in EASEWASTE as each material fraction could be assigned a specific TFC.

Fossil Carbon

The carbon content of the waste is distributed to fossil and biogenic carbon and the fossil C mainly originates from the material fraction “non recyclable plastic” which corresponds to 30 % of the total C content. The impact from fossil C is 8.6E-3 PE, as earlier mentioned, but this fraction also represents 21 % of the LHV_{waf} and thereby it saves impacts from traditionally energy production amounting to -3.4E-1 PE. It therefore seems to be beneficial to incinerate the PVC in the Aarhus incinerator, assuming that this would not lead to other emissions than fossil CO₂. Furthermore, as the legislative requirements are today, incinerating fossil CO₂ will dilute the other air emissions by the increase in CO₂ content of the flue gas. The placement of fossil C as the third most important parameter does not take into account the positive effects of the carbon and should therefore be ranked lower.

Arsenic and Copper

The importance of *As* and *Cu* to the human toxic categories and ecotoxicity is evident and it is therefore important to find the sources. However, since the corrected waste material fraction characterization in chapter 3.5 did not point at a specific material fraction as the main source, it remains unknown what causes the impacts. *As* has previously been used in wood preservation and it could be a important source of *As* in the waste although it has not been proven. *Cu* is present as a pure metal in a lot of products such as electronics and the *Cu* with no fraction of origin in the direct analysis could be cables, switches and transformers all containing large quantities of pure *Cu*. The environmental impacts related to handling and storage of ashes have not been included in this study but any relation between waste *Cu* content and the leaching (emission) of *Cu* to surface waters must be taken into account. TOC in bottom ash is known to accelerate the *Cu* leaching and *Cu* is among the most important emissions in the Ecotoxicity to water category. This parameter may be of great importance to the

overall environmental impact assessment of the incinerator and should consequently be investigated further.

Chromium and Lead

Unlike our knowledge about *As* and *Cu*, the corrected waste material fraction characterization has revealed the origin of *Cr* and *Pb*. *Pb* mainly originates from the inert fraction “Other of metal” and *Cr* was found to primarily in “Green glass”, “textiles” and “Shoes and leather”. Whether *Pb* and *Cr* transfers are the same for all of the fractions containing the substances is presently not known but seems a reasonable assumption. If this is indeed the case, the incineration of the inert fractions “Other of metal” and “Green glass” will only be a burden to environment as no substitution is present. As these fractions are not subject to any specific material recycling after incineration, it seems unadvisable to incinerate these fractions. The textiles, shoes and leather fractions will need further investigation before it can be determined whether they have net positive or negative potential impacts.

5 Conclusion

Life cycle analysis provides estimates on the potential environmental impact by a given functional unit, defining the service of a product or process. In this thesis the functional unit is incineration of one tonne of waste, and the focus has been on impacts related to the input waste. This focus has been chosen because of two questions of importance: Firstly, what is the potential of minimizing environmental impact in existing waste treatment systems without installing new or changing technology? Secondly, by which substances in the waste do waste management influence the impact on environment from waste incineration?

The modeling of impacts was performed using the EASEWASTE model and the sub-model of an incinerator. It has been demonstrated that waste specific impacts from incineration to environment play an important role compared to the process specific impacts. Further more they offer an opportunity for decision makers to improve environmental performance alone by source separation. The modeling makes it possible to link impacts on the environment to specific material fractions of the waste, but this requires that the waste input is characterized into material fractions before treatment.

A direct measurement of the chemical composition of 48 material fractions from residual household waste was conducted and to validate the result, an indirect measurement of the same waste type was also performed. The indirect measurements were performed using full scale incinerators and were conducted in three tests, all together covering 1400 tones of residual household waste. Corrections to the direct measurement were necessary because the primary sample mass was too small to represent rare items of importance. For the substances *As*, *Cu*, *Cd*, *Hg*, *Mo*, *Ni*, *Pb*, *Cl*, *Mn* and *Zn* the direct measurement were corrected to meet the measured content of the indirect method. The studies showed that the direct method was not suited to evaluate the total waste concentrations of rare trace elements. *Hg* was concluded to be influenced by so rare items that even the indirect analysis did not provide a satisfactory result. To obtain any result, longer test periods than 24 hours would have been needed.

LCA case studies showed great differences between the environmental impact of different incinerators as well as treated waste types. This serves as proof that a simple hierarchy principle will not cover local differences and therefore is not suited to evaluate waste management options involving waste incineration. The LCA on the waste specific impacts concluded that the most important parameters in the waste are (in prioritized order):

1. *Hg* content (in the “Batteries” material fraction but also as thermometers etc.)
2. Waste energy content (LHV_{waf})
3. *As* and *Cu* content (material fraction not determined)
4. *Cr* and *Pb* (inert material fractions, leather and textile)
5. Fossil *C* content (in the “Non recyclable plastic” material fraction)

It is also concluded that the prioritized order will be affected by the local conditions and the technology of the incinerator. The impacts can in some cases be related to specific material fractions, but not always. It is concluded waste management should

aim sorting inert fractions from the waste for incineration and that a direct campaign against *Hg* containing batteries, thermometers and switches in the waste for incineration is of major importance.

6 Future work

The research performed in my Ph.D. has pointed to three areas that would be interesting for future development and research: Direct waste analysis methodology, indirect waste analysis methodology and environmental assessments of waste incineration by LCA modeling. The main purposes of the future development would in my opinion be:

- Explore the influence on the transfer coefficients at the incinerator by single material fraction. It would be interesting to study whether *Cr* in stainless steel distributes in the same manor as *Cr* in Leather or paint? Another interesting question would be whether the influence is important to the environmental evaluation of single fractions management.
- Investigate from which material fraction the important substances in a LCA originate from. Today some substances of importance still have unknown origin, and this makes it impossible to remove the substances from the waste.
- Understand the influence of rare items on transfer coefficients at the incinerator. In this connection it would be interesting to study whether the high concentration levels in the rare items distribute different than the average distribution, and what effects this could have on the environmental assessment.
- Prove any relations between waste input characteristics and the quality of incinerator output products for recycling, for instance bottom ash leaching.
- Quantify the presence and effect of rare items for substances important to the outcome of the environmental assessment. It is important that we know the mass of waste pr. rare item pr. substance so that we can design the primary sample mass as small as possible.
- Include modeling of the output products and evaluate the importance of this step to the incinerator overall evaluation. *Cu* leaching from bottom ash might be of importance.
- Improve the indirect methodology with respect to sampling and retention time measurements to shorten test time and thereby save resources.

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8 Appendixes

I

Riber C., Fredriksen G.S. and Christensen T.H., 2005, Heavy metal content of combustible municipal solid waste in Denmark, *Waste Management & Research* 23, 126-132.

II

Riber C., Rodushkin I., Spliid H. and Christensen T.H., 2007, Method for fractional solid-waste sampling and chemical analysis, *International Journal of Environmental Analytical Chemistry*, 87, 5, 321-335

III

Riber C., Christensen T.H., Bhandar G.S., 2007a, Environmental assessment of waste incineration in a life-cycle-perspective (EASEWASTE), Submitted to *Waste Management & Research*.

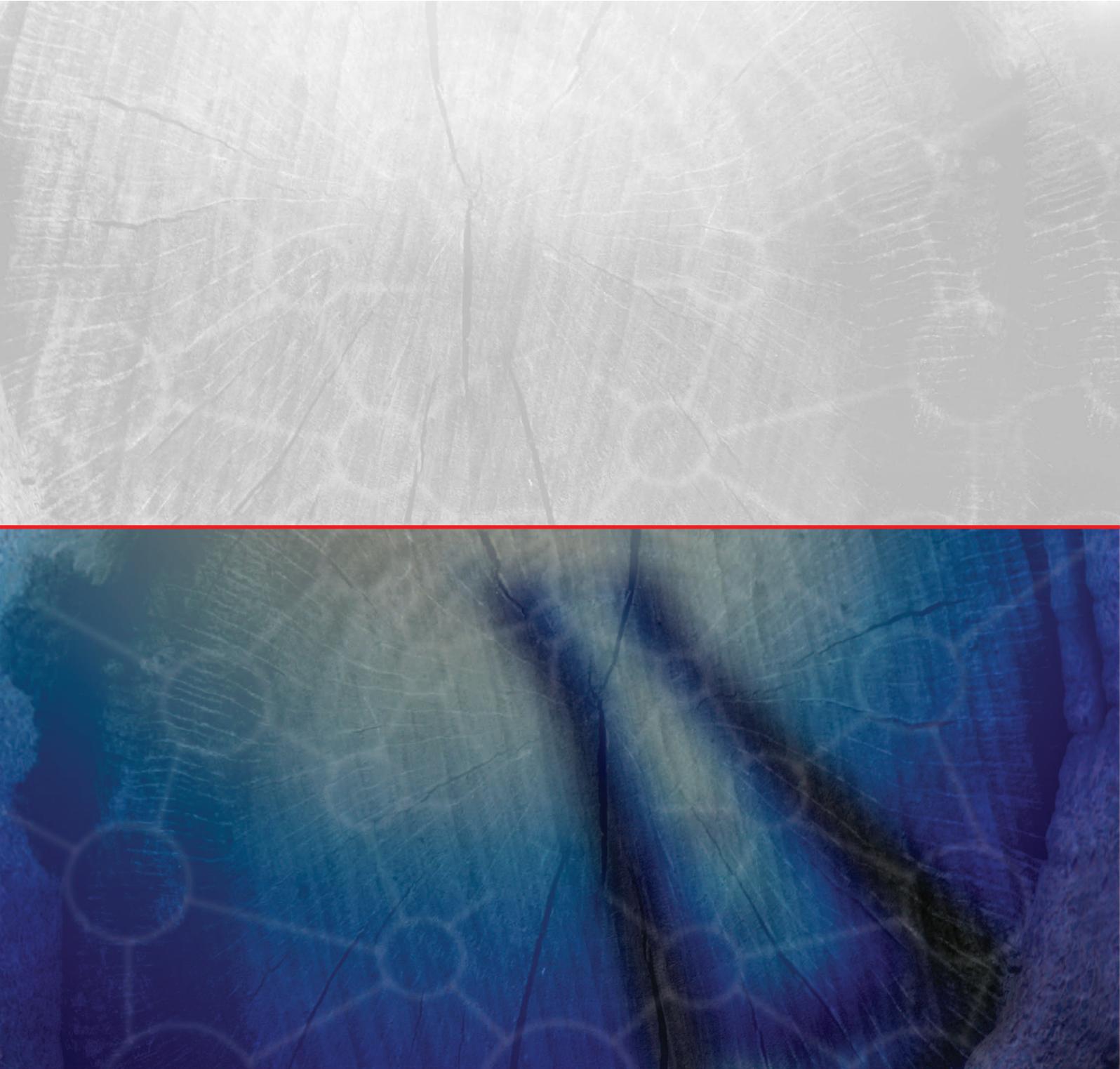
IV

Riber C., Claus Petersen and Christensen T.H., 2007b, Chemical composition of material fractions in Danish household waste, Manuscript.

V

Riber C. and Christensen T.H., 2007c, Indirect Determination of Chemical Composition and Fuel Characteristics of Solid Waste, Manuscript.

The appendixes are not included in this www-version but may be obtained from the Library at the Institute of Environment & Resources, Bygningstorvet, Building 115, Technical University of Denmark, DK-2800 Kgs. Lyngby (library@er.dtu.dk)

A microscopic image of plant tissue, showing a network of veins and circular structures. A red horizontal line is drawn across the middle of the image.

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