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The cases of Sweden and the Netherlands

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Department of Energy and Environment
Division of Environmental Systems Analysis
Chalmers University of Technology
SE-412 96 Gothenburg
Sweden
Telephone + 46 (0)31-772 1000

Cover:
Shipping of metal scrap and other waste. Rotterdam, the Netherlands. Photo by Henrikke Baumann, 2012.

Chalmers Reproservice
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EXECUTIVE SUMMARY

In this report, management practices and waste management and prevention policy that influence the environmental impacts from the metal packaging systems of Sweden and the Netherlands are described. Thereby, the report aims to target the environmental concerns related to packaging in Europe. The approach for reaching this aim has been to synthesise and spread knowledge about the, this far, little considered specific practices of for example the implementation of packaging recycling policy.

The environmental concerns related to packaging generally follow a *life cycle assessment* perspective and have since two decades been considered through not least an EU directive from 1994. The environmental concerns are related to the main materials commonly identified as the bases of metal packaging: steel and aluminium. Particularly prominent environmental issues are: energy use, global warming potential, raw material scarcity, and landfill space scarcity. *Recycling* (which in the packaging context and in this report refers to reprocessing in production processes but not for example energy recovery) is the promoted approach, and percentage wise large energy reductions have been stated to follow this approach.

This summary focuses on the report findings on metal packaging systems. The full report covers, in addition, method aspects.

Conclusions and recommendations for lowering environmental impacts from metal packaging

A few conclusions and recommendations for lowering environmental impacts from metal packaging systems are made based on the analysis of the results in this study, and they are summarised in the table further on in this summary. Conclusions are presented for both countries. The conclusions might be of relevance both specifically for the metal packaging systems, but also for waste management more generally. The conclusions are targeted for all actors in the respective systems, including producers, consumers, waste

management actors, and public authorities. For an overview, see also Figure 8 on page 52.

Sweden	The Netherlands	Details in chapter 4
Use complementary extraction from ashes: This can be a viable option besides the current sorting of household metal packaging waste at the waste source. However, the feasibility seems to depend on, for example, the effect this will have on the consumers' waste handling behaviour.		(k), p. 48–9
Increase the focus on aluminium: Aluminium packaging seems to be recycled at low rates and considerable environmental improvements can be reached if its recycling is increased.		(f), p. 46–7; (b), p. 45–6
Scrutinise official statistics: For example, recycling rates are calculated on consumption data that exclude a likely considerable amounts from not registered fillers and importers.	For example, assumptions of 100% recycling is used for one of the largest metal packaging waste streams.	(a)-(j), p. 45–8
Better avoidance of downgrading, in for example re-melting: Downgrading of material quality is related both to re-melting and to other waste handling processes. Downgrading is currently not monitored.		(h), p. 47; (l), p. 49
Improving relations between public and private actors could help: The presence of public-private conflicts have stalled improvements in monitoring data quality.	Material packaging fees were used for helping to cover national budget deficits, and the effects were, for example, that private actors were highly upset.	(m), p. 50–1
Opportunities for considerable improvements exist, particularly in Sweden, but are difficult to assess: Opportunities exist although the management of these metal packaging systems is viewed as successful compared to the management of other waste streams. Actually, this view might be part of the reason for the limited focus on them this far. The effects of changes are difficult to assess both due to aspects of each of them and due to their complex interdependences.		
It is easy to overlook factors of considerable importance: The combined material and management aspects of these systems seem to form a complex landscapes with many interrelations, such as between monitoring data quality, relations between public and private actors, and the aspect of downgrading of material quality through recycling.		(f), p. 46–7; (l), p. 49
Other practices not identified in this study can be of importance in addition: These practices include the actual possibility to use more metal packaging waste in the metal re-melting processes, potential environmental and other effects of conducting the re-melting in remote locations, the accounting of non-recycled metal packaging litter, and work on minimising packaging material per unit of packaging.		

Research approach

Due to the relatively similar and basic design of metal packaging in Sweden and the Netherlands, the method utilised in this study targets how organisational and governance practices in the metal packaging systems influence the packaging and its potential environmental impacts. This mix of, among other, humans, production and recycling equipment, and nature is here studied with a *socio-material* focus that accounts for both humans and non-human entities, in order to search for an as representative as possible description of the packaging systems. The use of socio-material approaches has become established in the social sciences and the humanities and arts, and has also been applied to relations between socio-technical systems and the natural environment. Further, methods have in this study been chosen and applied based on the specific conditions of this study, in line with the *grounded theory* approach, rather than selecting a pre-defined method set up. Specifically, discourse analysis and content analysis have been used as filters for text studies, interviews, and study visits. Further, two analysis frameworks have been introduced and applied in this study. The first one focusses the scopes covered in socio-material environmental findings. In this framework the concept of *product and management chains* is introduced. It covers material and energy flows according to the life cycle assessment perspective, the organisations directly handling these flows, as well as sector organisations. The second framework focusses on different socio-material characteristics of socio-material environmental findings. Regarding information sources, five interviews were performed, three in Sweden and two in the Netherlands, and two combined study visits were carried out in Sweden. Representatives for producer organisations and public authorities, respectively, were interviewed in both of the countries. The study was conducted between February 2013 and June 2014.

Material flows

As a first step of this study, an overview has been made of metal packaging flows and environmental impacts for Sweden and the Netherlands. The environmental part of this study on packaging systems follows the life cycle assessment perspective. This perspective is promoted by policy on the subject, not least at the EU level. Also, since large potential environmental impacts are caused by upstream processes in production of metals from virgin materials, these processes seem relevant to include when studying recycling for and waste generation prevention for these systems.

Regarding the basic flows of these systems, the annual metal packaging waste generation in 2011 was, from reported estimates, calculated to be 6.5 kg/capita in Sweden, and to be 11.6 kg/capita in the Netherlands. Generally, the metal packaging is divided between steel packaging and aluminium packaging. The shares of waste generation, by weight, for these fractions, were for Sweden in 2011 reported to be 57% steel packaging, and 43% aluminium packaging. For the Netherlands, the corresponding figures has in this study been calculated from estimates to be 89%, and 11%, respectively. Estimated overall metal packaging waste recycling were in 2011 for Sweden 75% and for the Netherlands 91%. Aluminium packaging was here from estimates calculated to in 2011 to be recycled at the rate 66% in Sweden and 63% in the Netherlands. For Sweden, aluminium packaging other than beverages is the smaller of two fractions of aluminium packaging but it was for 2011 here from estimates calculated to contribute to 0.6 kg/capita of the total 1.0 kg/capita of non-recycled aluminium packaging and to have a recycling rate of only 23%.

On potential environmental impacts for metal packaging systems, the apparent and in focus issues are energy use, global warming, raw material scarcity, and landfill space scarcity. Regarding energy use, recycling of the currently non-recycled aluminium fractions seems via proxy calculations for 2011 to have the potential to theoretically result in large percentage reductions of the two countries metal packaging systems' energy use, 64% for Sweden and 36% for the Netherlands, which is 13 and 8 times, respectively more than for steel. At the same time these aluminium fractions represent small shares of the annual waste generation in these metal packaging systems, and these metal packaging systems account for around 0.1% of each nations energy use. For global warming potential, the picture is similar, with corresponding aluminium fraction reduction potentials of 63% for Sweden and 37% for the Netherlands, and being 8 and 5 times, respectively, higher than for steel while corresponding to 0.1%–0.2% of the reported inland greenhouse gas emission from the respective two countries. For raw material scarcity, whether the steel or aluminium packaging is the largest impacting stream depends on how currently sub-economic aluminium resources are treated. The estimated per capita level of iron ore use for metal packaging in each of the two countries is from proxies here calculated to for 2011 have been around 0.4% of the total iron ore use per capita globally. The corresponding figures for bauxite are considerably higher – 15% for Sweden and 8% for the Netherlands. Regarding number of years that the here calculated reserves and resources of the two ore types will last is on the same order of magnitude, 80–110 for reserves, and 220–250 for resources, respectively. Finally, regarding landfill space scarcity, reported landfilling amounts are in total around 200 times higher than for non-recycled metal packaging in Sweden, and the corresponding Dutch figure is around 100 times.

Management practices

With these material flows, and additional ones presented in the more detailed main part of this report, in mind, two main groups of management practices with potential environmental influence were identified. The first of them covers potential ineffectiveness of public policy and data. The second is on conflicting arguments and perceptions. The two groups encompass:

For public policy and data:

- 5 issues on lack of reliability of data.
- 3 issues on lack of resolution in public policy and in data.
- 1 issue on lack of comparability internationally.
- 1 issue on lack of environmental accountability.

For arguments and perceptions:

- 1 issue on metals incineration.
- 1 issue on sorting in waste streams.
- 1 issue on public-private conflicts.

First, for the lack of data reliability issues, the first issue is that so called free riders may make statistics look better than reality in Sweden. Free riders supply packaging to the market but are not paying for or helping to organise the recycling system. There is no formal obstacle for their packaging to enter the recycling system and to be accounted for there, while their supply to packaging waste generation is not accounted for. The second issue is that the statistics are difficult to follow up due to a change of the calculation of aluminium beverage packaging that are sorted into a recycling system where they are not supposed to be sorted. These two separate systems are the separate system with a refund for consumers that sort the cans into this system, and the general metal packaging sorting system designated for the remainder of household metal packaging waste. After the change, an estimated compensation is made of the statistics for the cans that are sorted into a system where they are not supposed to be sorted. The third issue is on that Swedish packaging statistics are not well defined and cross-checked. The fourth issue is on the calculation of bottom ash extraction for the Netherlands being difficult since other metal sources are present. The fifth and last issue is that the Dutch metal packaging statistics contain assumptions based calculations that are not scrutinised by otherwise rigorous authorities.

Second, for lack of resolution in public policy and in data, the first issue is on aluminium not being regulated separately despite being little recycled and resulting in several seemingly large environmental impact reductions percentage wise if fully recycled and with Swedish non-beverage packaging being a specifically potentially promising waste stream for increased recycling. The second issue is on a lack in the statistics resolution of the many different paths of collection and of sorting that exist in the Swedish metal packaging system while the Dutch system lacks separate reporting on steel and aluminium packaging, respectively. The third and last issue is on downgrading – that is, quality loss through for example recycling – not being accounted for in general for packaging.

Third, for lack of comparability internationally, the issue is that country comparisons are difficult since statistics calculation methods differ between countries.

Fourth, for lack of environmental accountability, the issue regards the current approach of focusing on different packaging materials separately and on packaging separated from other environmentally impacting activities. The question is whether this approach is an effective one for reaching decreased environmental impacts, or whether it only leads to sub-optimisation.

Fifth, for metals incineration, the issue is on whether complementary extraction from ashes is encouraged and feasible in Sweden.

Sixth, for sorting in waste streams, the issue is that it has been focussed considerably on in Sweden previously despite the relatively basic problem of copper and tin impurities then being spread between different metals packaging recycling systems.

Seventh, and finally, for public-private conflicts, the issue is on possibilities of targeting conflicts in the Swedish general packaging system, and in the Netherlands of using packaging fees for covering budget deficits. Regarding the Swedish issue, an aspects that might be handled through inspiration from the Dutch system is that the Swedish system currently is heavily divided between public and private actors and that the public actors recently have been suggested to manage a drastically larger share of this system. These suggestions are now causing severe disagreements that also hinders the development of more accurate statistics generation on packaging.

Analysis

For further analysis, first, the findings of this study were related to the main processes of the product and management chains and their environmental impacts. The starting point was that this study did not cover all aspects and that several were indicated but not treated fully in the report. Keeping this in mind, the outcome of the analysis pointed to a rich and complex landscape, and which resulted in additional observations. A combination was found of considerable potential opportunities for lowering environmental impacts and a difficulty to assess the effects of such measures. Further, the complexity seems to add to the risk of easily missing vital aspects when managing the systems. Finally, five particularly prominent potential improvement areas of the systems were pointed out. These are in Sweden (re-

)introducing complementary extraction of metal packaging waste from incineration ashes, an increased focus on aluminium packaging, and on downgrading of material quality, improving relations between public and private actors, as well as scrutinising of the production of official statistics.

In addition, this study has been compared to the scopes and characteristics of earlier socio-material environmental studies. Regarding scopes, this study has a larger coverage of issues related to policy design and policy processes. This should not be surprising since this is a topic where policy is and has been used to a comparably large extent. Nevertheless, the issues of this study, like earlier studies, are in all cases related to socio-material relations both along product and management chains as well as within the nodes of these chains. Interactions between different product and management chains and with external factors are also present in one of the findings of this study.

Regarding socio-material characteristics that are prominent in the findings of this study, a few remarks can also be made. This study differs only clearly in one of these characteristics from earlier socio-material environmental studies. A multitude of drivers is in this study found to be a more common feature. In common with the previous studies, frequent characteristics include: that both social and material agency is seen, imperfectness of interactions, the importance of where the boundary of activities considered is set, as well as non-environmental drivers.

Keywords

Metal packaging systems, life cycle assessment (LCA), management practices, policy, recycling, Sweden, the Netherlands, socio-material, packaging, environmental, and management.

PREFACE

This study has been made possible to carry out through the funding provided by J. Gust. Richerts Stiftelse with 180 kSEK, and the funding application granted by Ragnar Sellbergs Stiftelse with 100 kSEK. I am very grateful for their interest in this project.

The study has been carried out between February 2013 and May 2014.

The study has been conducted by PhD candidate Mathias Lindkvist (Division of Environmental Systems Analysis, ESA, Department of Energy and Environment, Chalmers University of Technology, Gothenburg, Sweden; and guest researcher at the Industrial Ecology Programme, Faculty of Engineering Science and Technology, Norwegian University of Science and Technology, NTNU, Trondheim, Norway), project leader and associate professor Henrikke Baumann (ESA; and guest researcher at the Department of Public Administration, Faculty of Social Sciences, Erasmus University Rotterdam, Rotterdam, the Netherlands), and waste management researcher and assistant professor Maria Ljunggren Söderman (ESA; and IVL Swedish Environmental Research Institute, Gothenburg, Sweden).

The study would also not have been possible to carry out without the time and effort invested by interviewees, study visit facilitators, and other informants. This was most prominently, but not only, provided by: Thord Görling (Packaging and Newspaper Collection Service, Förpacknings- och tidningsinsamlingen AB, FTI, Smögen and Stockholm, Sweden), Peter Hammarstrand (IL Recycling, Gothenburg, Sweden), Christina Jonsson, Cecilia Mattsson, and Catarina Östlund (Swedish Environmental Protection Agency, Naturvårdsverket, Stockholm), Robert-Jan ter Morsche (Stichting Kringloop Blik, SKB, Zoetermeer, the Netherlands), and Marcel Verweij (Inspectie Leefomgeving en Transport, ILT, Arnhem and Utrecht, the Netherlands).

Trondheim, Norway, 23 June 2014

CONTENTS

1. INTRODUCTION

In this report, findings are presented from a study on how management practices and policy influence the environmental impacts from metal packaging systems. The study has covered management practices related to policies on waste management and waste prevention, for the metal packaging systems in Sweden and the Netherlands. Thereby, the aim has been to target the environmental concerns related to packaging in Europe (cf. Directive 94/62/EC 1994, 2013). More specifically, the study has focused on synthesising and spreading knowledge about the little considered *practices* of implementing packaging *recycling*[1] policy. This focus is motivated by a review, in this study, of publicly available material on this aspect. This review revealed that the aspect was only touched upon in the material that addressed it and which was most prominently found in two reports by Greenblue (2011) and PRO Europe (2011).

Regarding environmental impacts, these have been of widespread concern for packaging and packaging waste in Europe since the early 1990s (cf. Directive 94/62/EC 1994, Rouw & Worrell 2011, SOU 1991:76). The concern is most clearly manifested through an *EU directive* from 1994 (see Directive 94/62/EC 1994, 2013). Through this directive, improvements and harmonisation of packaging recycling and prevention are requested. The environmental issues concerned are generally derived from and considered through applying a *life cycle assessment* perspective. Life cycle assessments consider material and energy flows along entire *product chains*, from raw material extraction, via, among other, production, transports and consumption, to terminal waste management or recovery (core publications on life cycle assessment include Baumann & Tillman 2004, European Commission 2010, Finnveden et al. 2009, Guinée 2002, ISO 14040:2006, 14044:2006). Environmental issues focused on regarding packaging systems are, among other: energy use, global warming potential, raw material scarcity, landfill space scarcity, and noxious substances used in packaging materials (e.g., Directive 94/62/EC 1994, pp. 10–11, 2013, pp. 3–4, Rouw & Worrell 2011, p. 483, Tillman et al. 1991).

Actions taken in response to these concerns have prominently included extensions of already previously existing policies on and recycling systems for beverage packaging to include all packaging. Investigations have been carried out, and policies, and waste prevention and recycling systems or have been set up and have become established in almost all European nations. (cf. Directive 94/62/EC 1994, PRO Europe 2011)

[1] The term recycling is in this report used in accordance with the definition of it in the EU directive on packaging and packaging waste (see Directive 94/62/EC 1994, 2013). In this definition, recycling covers "reprocessing in a production process of the waste materials" (Directive 94/62/EC 1994, p. 13) and excludes energy recovery (Directive 94/62/EC 1994, p. 13).

However, despite that these effort already have been running for two decades, the reported packaging recycling levels still vary considerably between the EU member states and even the figures themselves seem in need of being scrutinised. Of the 26 of the states that have reported overall packaging waste recycling rates for the year 2011, as of 10 December 2013, these range between 69% and 81% for the five countries with the highest rates and between 41% and 51% for the five countries with the lowest rates. Regarding the reliability of the figures, they have in several cases considerably exceeded the logical maximum of 100%. The figures for Denmark range between 115% and 175% for glass packaging recycling between 2006 and 2009, with no further explanations provided in direct connection to these statistics. (Eurostat 2013b)

Also, in general there seems to be little action based on knowledge exchange between packaging systems in different European countries. Since the packaging products themselves are fairly similar and basic regarding design, such as cardboard boxes and glass bottles, it seems as if the search for a more practically applicable understanding of these systems ought to include other parts of them. This includes the organisation of the technical systems for handling the packaging and the governance and management of these systems. (cf. Directive 94/62/EC 1994, Greenblue 2011, PRO Europe 2011, Rouw & Worrell 2011). Further, in order to strive for a held together understanding of such systems, it seems important to direct sufficient resources to studying the management practices that link policy to the actual material flows.

In this report, we present a pilot study on technical, organisational, and policy aspects of European packaging systems regarding one of the six packaging materials groups from the perspective of environmental performances. The study covers metal packaging systems in the two EU member states Sweden and the Netherlands. Metal is one out of six main categories of packaging used in the EU directive on packaging and packaging waste (Directive 94/62/EC 1994, 2013). We have identified substantial differences between the metal packaging systems of these otherwise similar countries regarding both reported recycling levels and overarching management practices. Recycling rates are reported to lie between 75% and 78% during 2009–2011 for Sweden, while the corresponding figures for the Netherlands are 87% and 91% (Eurostat 2014c). The management of the two systems differ regarding for example the use of *sorting at the waste source*, that is, the product user's disposal of metal packaging into a waste stream that is designated for only containing metal packaging waste, or metal packaging waste and one or a few other waste materials. This is the main approach used for managing the large waste stream of metal packaging waste from households in Sweden while it is only marginally used for the corresponding waste stream in the Netherlands. (cf. Naturvårdsverket 2012a, Nedvang 2012) Thus, it seems to be a case of relevance in itself, while features that are applicable to other waste management also might be found from this study.

The subsequent main part of this report consists of three chapters. First, in chapter 2, the research approach of this study is presented by describing conceptual starting points, the methods that have been used for carrying this study out, the search for and characteristics of information sources, and delimitations and definitions applied. Second, in chapter 3, an overview is presented of the metal packaging and metal packaging collection and recycling systems. These are delimited to mainly cover the flows of the Swedish and Dutch consumption from a life cycle assessment perspective. Third, and finally, in chapter 4, the specific management practices of potential environmental significance that have been found in this study are presented and analysed.

2. RESEARCH APPROACH

In this chapter, concepts and methods used in this study are presented. In sub-chapter 2.1, the use of a general approach of combining studies of organisational and technical aspects is motivated and presented. In sub-chapter 2.2, previous similar applications of this approach are described. In sub-chapter 2.3, the general approach to methods for this study and the specific methods used, are outlined. In sub-chapter 2.4, the path of choosing and the characteristics of the information sources used for this study are explained. Finally, in sub-chapter 2.5, delimitations and definitions, on for example which metal packaging that is covered in this study, are presented.

2.1. Socio-materiality

Both material entities, such as metal cans, and human actors seem to play central roles in the metals packaging systems as well as in other packaging systems. Therefore, it has been considered to be feasible to use a combined research approach, where both material and technical aspects, such as properties of the materials used in the packaging, and human and social aspects, such as management practices, are taken into account. Such *socio-material* study designs have over the last few decades become applied to an increasing extent by scholars in the social sciences and in the humanities and arts. They have considered material aspects to be important for reaching relevant research results in a broad range of fields, such as studies of the practice of scientists, business administration, finance, and literature and film. As a consequence, these aspects have been conceptually added to these scholars' core areas of social and cultural aspects, and often in an integrative way where no clear borders are drawn between different entities. (e.g., Åsberg et al. 2012, Barad 2007, Callon 2001, Harman 2011, Holbraad 2011, Latour 1987, 2005, Miller 2005) Of these approaches, the open ended *actor-network theory* (e.g., Latour 1987) seems to lie closest to environmental studies of technical systems through its initial formation in the research field *science and technology studies* and its further development in studies of management practices (cf., e.g., Baumann 2004, 2008, 2012, Callon 1998, Czarniawska 2000, 2004, 2005, Czarniawska & Hernes 2005, Latour 1987, 2005). It thus forms a logical bridge to this study, but three additional socio-material approaches are here also used as starting points. The general feasibility of performing socio-material environmental management studies has already been found to be satisfactory through previous studies (Brunklaus 2008, Brunklaus et al. 2009, 2010, Lindkvist & Baumann 2010, 2013), and three of the four socio-material approaches used here have been successfully tested as analysis tools for these types of studies (see Lindkvist & Baumann 2013).

A few more specific characteristics of both actor-network theory and the three additional socio-material approaches have been considered to be specifically relevant to have in mind for this study and for similar studies. These characteristic are described in the following, and they are summarised in Table 1, further on in this sub-chapter. The three additional approaches here used as a basis are *object-oriented ontology* (e.g., Harman 2011), *agential realism* (e.g., Barad 2007), as well as Martin Holbraad's (2011) research on close human encounters with material entities – an approach that he labels as *pragmatological*. From these four approaches, three characteristics have been considered in relation to the types of entities covered by socio-material studies, while four characteristics of socio-material interactions have been identified.

Table 1. Overview of environmentally relevant socio-material characteristics that have been derived from four socio-material approaches.[1), 2), 3)]

			Socio-material approach			
			Actor-network theory	Agential realism	Object-oriented ontology	Pragmato-logical approach by Holbraad
Entities	(1)	Socio-materiality	X	X	X	X
	(2)	Both social and material agency	X	X	X	X
	(3)	All levels from micro to macro	(X)		X	
Interac-tions	(4)	Imperfectness of interactions			X	
	(5)	Open-ended and close study				X
	(6)	Boundary choices (incl. time)	X			
	(7)	Mutually excluding practices		X		

[1)] Figures within brackets refer to descriptions in the body text further on in this sub-chapter.
[2)] X = central characteristic to respective approach.
[3)] (X) = semi-central characteristic to respective approach.

Regarding entities, the general consideration of both *social and material aspects* are central to all of the four socio-material approaches (1). They are, not least, central since both humans and material entities are considered to matter in the sense of exercising agency in these approaches (2). (Barad 2007, Harman 2011, Holbraad 2011, Latour 1987). Further, object-oriented ontology, and to some extent actor-network theory, point out that micro, macro, and every level in between are of potential relevance to consider (3) (Harman 2011, Latour 1987).

On interaction related characteristics, it is through object-oriented ontology argued that information transfer between entities is incomplete, distorted or during certain periods of time non-existing (4) (Harman 2011). This reasoning can be further nuanced using Holbraad's (2011) position that material entities can be reasonably well understood by humans if they are approached through an open-ended and thorough approach (5). From actor-network theory can further be brought a potentially generally applicable finding about that conclusions of a study are considerably dependent on the scope of it (6) (Latour 1987). The case that Latour (1987) presents most explicitly in this regard deals with that whether a person was seen as successful or not depended on the time frame during which that person's actions were considered. Finally, in agential realism, Barad (2007) presents a combination of, among other, ontology and epistemology, where certain socio-material practices exclude other practices (7).

2.2. Socio-materiality in environmental studies

Recently, *environmental socio-material studies* have also been performed on relations between socio-technical systems and the natural environment. In this section, a general selection of findings from these studies are described with the intention of providing an analysis approach for both this study and for similar studies. An overview of the findings is presented in Table 2, in the following sub-chapter, where they are there grouped using a *scope framework* on the coverage of these findings. This framework is described further in that sub-chapter, as well.

In order to further systematise the findings presented in this section, the overview of environmentally relevant socio-material characteristics introduced in the previous section is here further developed into a *characteristics framework*. The framework and its relation to these findings is presented in Table 3, in sub-chapter 2.3, and the framework is also further described in that sub-chapter. Each of the characteristics presented in the preceding sub-chapter was identified in one or more of the findings, and all of these characteristics are therefore included in the framework. Four additional characteristics were derived from the findings, and they are also included. In addition, they are introduced through the descriptions further on in this sub-chapter.

The environmental socio-material studies considered in this sub-chapter have been carried out through either the approach *environmental assessment of organising* (Baumann 2004) or through its closely related research programme *organising for the environment* (Baumann 2008). They have in most cases used a life cycle assessment perspective as one of their bases.

Three of these studies that are here covered were carried out within one PhD project. First, in a project on environmental impacts of management practices, residential properties management was studied. In one case study a *caring* type of organising was found to be less environmentally impacting than an *emergency-driven* one when comparing the amounts of energy use and water use between two in other respects similar housing estates. The emergency driven approach was in the case the consequence of an organising inspired by hotel management. (A) (Brunklaus 2008) From this case, non-environmental drivers as a seemingly relevant factor has been added to the list of socio-material characteristics (8). Second, in a case of comparing passive houses to conventional buildings, the generally less environmentally impacting passive houses were found to be as impacting as conventional houses if the electricity provided to the households were supplied from non-renewable sources. The origin of the electricity for materials production also became of higher environmental importance for the passive houses. This pointed to considering not only overall technology and management choices but also considering the case dependent sub-aspects, as well as the potential need for increased co-operation along the product chains to synchronise practices by residents, construction firms, subcontractors, and municipalities. (B) (Brunklaus et al. 2010) Third, regarding organisational processes, internal management indicators, particularly of bottom-up types, have (recently) been found to be lacking in relation to environmental goals. Such indicators have been seen as a necessity for achieving actual implementation that result in lowered environmental impacts. On the other hand, the common focus on reporting to external stakeholders was found to only have fulfilled the purpose of communicating an environmental awareness. (C) (Brunklaus et al. 2009)

A few further studies based on or related to these approaches have revealed additional socio-material environmental findings. It has been pointed out that the natural environment can set the frame for organisational patterns rather than traditional boundaries of authorities in the case of watershed management (D) (Adolfsson 2005, 2007). Further, in a case of litter cleaning an organisationally long chain of actions for handling this environmental issue was found to result in little actual cleaning (E) (Lundberg 2008). Based on this finding, the number of steps in a chain of actions is added to the already covered socio-material characteristics (9). In another study, it has been shown how operationally critical breakdown situations may spur less environmentally impacting management practices of increased co-ordination, in a case on cement production (F) (Lindkvist & Baumann 2010, 2013). In a further cement production case, repeated feedback loops between industry and regulative bodies have been indicated to assist in industry launched environmental initiatives that aimed for emissions levels considerably below requested levels (G) (Lindkvist & Baumann 2010). To account for this finding, the frequency of interactions has been added as a socio-material characteristic to consider (10). Further, not changing management practices and staff responsibilities in a growing organisation has been found to be a possible reason for increased discarded residues, in a case on bakeries (H). In this bread case, environmental consequences were outlined for a small company that both was diversified and used one supply material in large amounts. In addition, the company was being forced by increasing currency instability at the same time as its management did not have any particular focus on life cycle environmental impacts. This resulted in a change to a potentially more environmentally impacting supplier alternative. (I) (Lindkvist & Baumann 2013) In order to reflect this finding, a socio-material characteristic on multiplicity of drivers has been introduced (11).

A few summarising comments can be made about these studies of environmental aspects related to socio-material systems. The life cycle assessment perspective indicates the importance of considering material and energy flow relations from raw material extraction, via, among other, production, transports and consumption, to terminal waste management or recovery. These flows are connected to human actions along these chains, but these actors may lack the knowledge about the environmental effects along these flows caused by their actions, and actions at different points in the chains may be un-co-ordinated or loosely co-ordinated and may counteract each other. (Baumann 2008) Further, the findings listed in this sub-chapter indicate a broad spectrum of practices of environmental relevance. The ability of considering case specific practices of importance thus seems to be a central tenet of this type of studies. Also, two more general considerations seem to be worth taking into account. First, the potentially different findings that will result from different choices of boundaries regarding time, space, types of activities studied and other relevant factors seem in need of specific consideration. Second, practices at different levels of scale and the potentially complex interdependence between these ought to be taken into account if present.

2.3. Methods

The in the previous sub-chapters described socio-material philosophy and findings have been used as the basis for this study, while a flexible and when needed iterative approach was used for selecting the methods applied in this study. This type of open-ended study design lies in line with the grounded theory method approach (see Glaser & Strauss 2006).

In practice, two main method tenets have been integrated. The first one is selected approaches from material and energy flows studies for a *material part of the study*. The second is qualitative techniques for an *organisational part of the study*. These are combined in the two analysis frameworks already referred to in the preceding sub-chapter.

First, for the quantitative flows studies, life cycle assessments have been used as the basis, and this use has in some cases bordered to the related method *material flow analysis*. Life cycle assessment, conceptually presented in Figure 1, further on in this sub-chapter, is a tool for modelling material and energy flows systems. These flows cover product chains, which include chains of both products and services. They consist of supply chains, consumption phases and waste management, and their associated (potential) environmental impacts. (Baumann & Tillman 2004, Finnveden et al. 2009, ISO 14040:2006, 14044:2006) It has been applied to specific products and services, and also to groups of products, the latter being the case for packaging (cf. SOU 2001:102; Tillman et al. 1991). The focus of life cycle assessments lies generally on the environmental impacts caused by providing a certain amount of a function to for example consumers, which is described through the *functional unit*. In practice, basic and still representative units are used, since the precise function of a product or service often varies between for example different users, and due to the high complexity of detailed models of actual phenomena. Therefore, the functional unit is typically for example one kg of tomatoes consumed, or one square metre of roofing provided during one year. (See, e.g., Baumann & Tillman 2004) In the case of life cycle assessments of packaging, one kg of packaging material is the functional unit usually used (see, e.g., SOU 2001:102). For the use of life cycle assessment in this report, the concepts *cradle-to-grave*, *cradle-to-gate*, and *gate-to-gate* are here also outlined. Cradle-to-grave refers to covering the environmental impacts directly related to all of the technical systems in the product chain. Cradle-to-gate refers to covering environmental impacts from the beginning of the technical system of the product chain but only until the stage of a chosen intermediary process of the technical systems in the product chain. Finally, gate-to-gate refers to covering environmental impacts from one process in the product chain. (cf. Baumann & Tillman 2004, Finnveden et al. 2009, and ISO 14040:2006, 14044:2006)

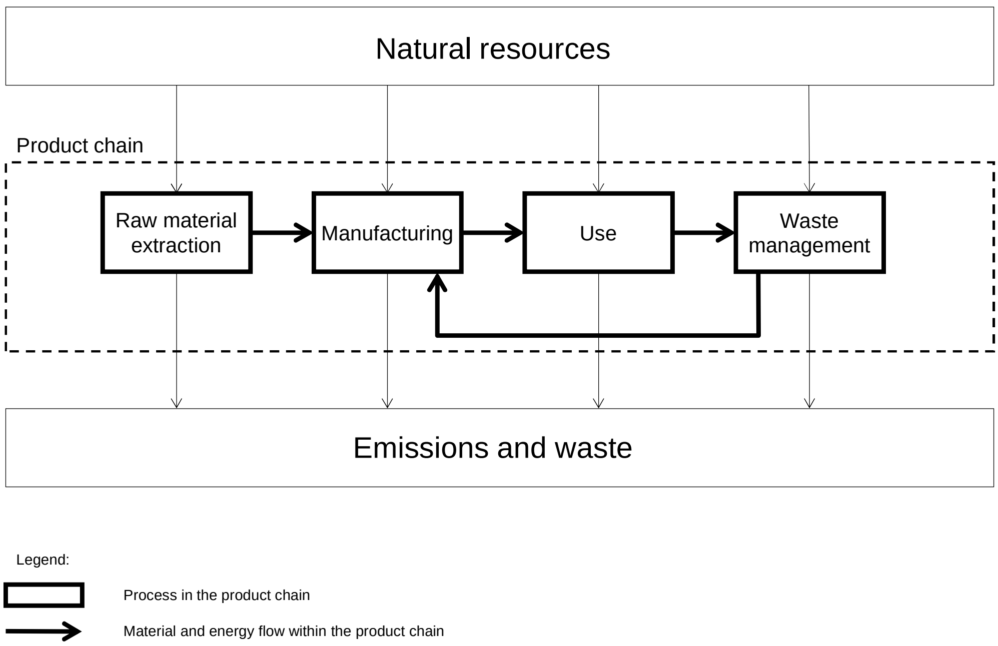

Figure 1. Simplified conceptual presentation of the life cycle assessment perspective. Material and energy flows along the product chain and their related flows are modelled. The related flows cover environmental impacts through use of natural resources and through emissions and waste generation.

In this study, life cycle assessments are in two ways bordering to material flow analysis. This regards the applying of it to a product group, and regards when the scope is the consumption within a certain geographical area. Material flow analysis is generally used to quantify the different flows of a certain substance or material, either globally or for a certain geographical region or for certain consumption, production and waste chains (Bringezu & Moriguchi 2002). However, life cycle assessments of product groups differs from material flow analysis by, in addition to one main material, tracing flows of other related materials and substances and considering their (potential) environmental impacts. Thereby, life cycle assessments also explicitly considers for example global warming potential of carbon dioxide emitted from production processes, and the environmental impacts of substances added to the main product material during production.

Second, qualitative analysis has in addition been applied to texts, interviews and study visits. The techniques *discourse analysis* and *conversation analysis* have specifically been used as interpretation filters. Through discourse analysis it is argued that speech and other activities mainly are framed by a limited number of trends and conventions. Words, statements and reasoning are aligned to them for fulfilling a purpose and for being conceived as meaningful. Conversation analysis is similar but focuses on the more detailed level of speech order. Its focus lies on, for example, the possibilities and constraints that a spoken phrase in a discussion between two or more persons puts on the phrases that may follow. (Silverman 2006)

Third, and finally, the two integrative frameworks, already referred to, are here briefly described. The objective of the first of these frameworks is to indicate how socio-material environmental findings relate to the life cycle assessment perspective with associated *actors*, and to policy. Concepts in this scope framework are presented in Figure 2, further on in this sub-chapter, and the framework is applied to the findings described in the preceding sub-chapter in Table 2, further on in this sub-chapter. Life cycle assessment is a relevant analysis unit due to its general importance in environmental assessments (cf. Baumann & Tillman 2004) and due to its central role in the packaging discourse, as already pointed out. It has here been extended to include, first, the actors directly handling the material and energy flows, such as producing companies and citizens consuming the goods. Taken together, one such actor and the process of the product chain that it handles is in this report denoted a *node*. These nodes together with sector organisations has been used as the delimitation of the here introduced *product and management chain* concept. The inclusion of sector organisations in this concept is motivated by their often considerable role in supply chain related issues. (cf. FTI n.d. c, SKB 2007) In this framework, a further distinction is made regarding whether findings particularly concern relations between different product and management chains, along them, or within nodes inside them. Policy is included as a unit of analysis, but not within the product and management chain. Its inclusion in the analysis has been prompted by its generally prominent role in the environmental discourse and due to being a starting point for this study. A division is made between the *contents* of policy documents, and the *processes* around policy for, for example, enforcement, since such a division has been found to be relevant based on the findings in the previous sub-chapter. Relations that do not clearly fall into either the product and management chain or policy perspective are handled through an analysis unit for external factors, a unit of analysis that was also pointed out as significant through these findings.

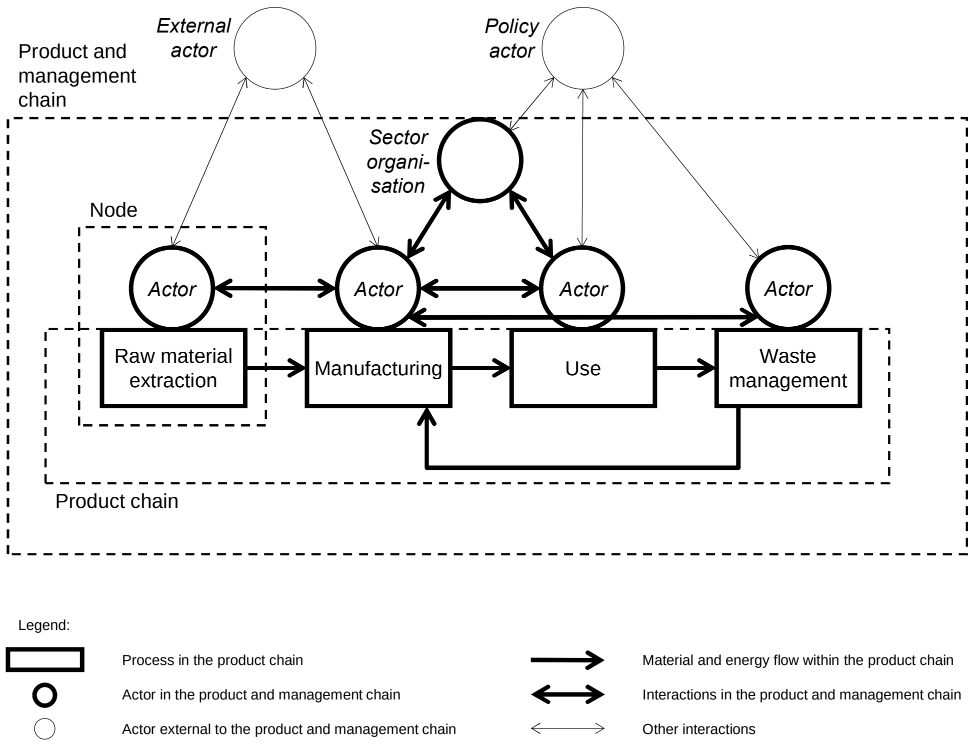

Figure 2. Simplified conceptual presentation of the scope framework. Combination of the life cycle perspective, presented in Figure 1, and socio-material actors that are clearly related to the product chain. Natural resources, and emissions and waste are, as presented in Figure 1, part of the framework, but are not displayed in this figure in order to display the additions to the life cycle assessment perspective clearly.

Table 2. Findings from environmental socio-material studies – grouped according to the scope framework presented earlier in this report. [1)]

	Scopes clearly covered in finding					
	External	Policy		Product and management chains		
		Con-tents	Pro-cesses	Between chains	Along chains	Within nodes of
(D) The environment setting the frame	X			X	X	X
(A) Caring or emergency-driven acting	X				X	X
(C) Need for bottom-up indicators	X				X	X
(I) Company pressured by currency instability	X				X	X
(G) Contact with authority repeatedly			X		X	X
(E) Organisationally long action chain				X	X	X
(B) Other impacts from new type of building					X	X
(H) Organisational stability during growth					X	X
(F) Emergency situation leading to change						X

[1)] Letters within parentheses refer to descriptions in the body text in sub-chapter 2.2.

The second framework used in this study is a framework on environmental socio-material characteristics. Its outline, origins, and its application to the socio-material environmental findings presented in the preceding sub-chapter are presented in Table 3, further on in this sub-chapter. As already mentioned, this framework is based on the characteristics outlined in sub-chapter 2.1, with additions from the socio-material findings from earlier environmental studies. Due to these added characteristics, additional groupings have been made to the outline used in sub-chapter 2.1, within interactions related characteristics. The sub-divisions used are: quality, quantity, and drivers, respectively.

Table 3. Characteristics framework, for environmental socio-material characteristics – origins and its application to socio-material environmental findings, respectively. 1), 2)

				Derived from		Findings (of in total 9) that each characteristic applies to	
				4 socio-material approaches	Socio-material environmental studies	Clearly	Partly
						%	%
Entities		(1)	Socio-materiality	X		89	11
		(2)	Both social and material agency	X		78	
		(3)	All levels from micro to macro	X			78
Interactions	Quality	(4)	Imperfectness of interactions	X		78	
		(5)	Open-ended and close study useful	X		11	44
	Quantity	(6)	Boundary choices (incl. time)	X		78	
		(9)	Number of interaction steps in a chain		X	56	
		(10)	Frequency of interactions		X	78	
		(11)	Many drivers		X	11	
	Drivers	(7)	Mutually excluding practices	X		44	11
		(8)	Non-environmental drivers		X	78	

1) Figures within parentheses refer to characteristics described in the body text in sub-chapter 2.1 (characteristics 1–7), and in sub-chapter 2.2 (characteristics 8–11).
2) More detailed data used for calculating the percentage values are presented in Appendix A, in Table A.1.

2.4. *Information sources*

Regarding information sources, this study is based on publicly available material, interviews and study visits. The process of searching for and the characteristics of these sources are here briefly described, and an overview of the interviews and study visits is presented in Table 4, further on in this sub-chapter. Initial studies of publicly available material were complemented by interviews with key persons with an overview of the Swedish system. These interviews were performed in order to search for potentially different perspectives on the findings from these initial studies, and since the aspects studied through this project were little covered by the available public material. Assisted by the results of these initial interviews, it became possible to select representative study objects at a more detailed level of the systems in both Sweden and the Netherlands. Additional interviews and two combined study visits were identified and carried out, and additional objects of publicly available material were studied.

Table 4. Overview of interviews and study visits performed in this study.

Country	Type of activity	Interviewee or study visit guide		Date of activity, all during 2013	Description
		Organisation [1)]	Name		
Sweden	Interview	Packaging and Newspaper Collection Service	Görling, Thord	22 Oct.	Screening interview from the perspective of industry
		Swedish Environmental Protection Agency	Jonsson, Christina	24 Oct.	Screening interview on data aspects of packaging statistics
			Östlund, Catarina	24 Oct.	Screening interview from the perspective of the national agency
	Study visit	Packaging and Newspaper Collection Service	Görling, Thord	26 Nov.	Visits to three intermediate scale *unmanned collection facilities* (in Swedish: '*återvinningsstationer*') for household packaging waste, Gothenburg, Sweden
				26 Nov.	Visit to intermediary storage for household packaging waste, after collection and before recycling, at IL Recycling, Gothenburg, Sweden
The Netherlands	Interview	Stichting Kringloop Blik	Ter Morsche, Robert-Jan	28 Nov.	Screening interview from the perspective of industry
		Inspectie Leefomgeving en Transport	Verweij, Marcel A.P.	5 Dec.	Screening interview from the perspective of the national agency

[1)] Organisation names and acronyms:
Packaging and Newspaper Collection Service
= Förpacknings- och tidningsinsamlingen AB, FTI.
Swedish Environmental Protection Agency
= Naturvårdsverket.
Stichting Kringloop Blik
= SKB.
Inspectie Leefomgeving en Transport
= ILT.

Regarding characteristics of the sources, the publicly available material includes reports from and webpages of for example governmental agencies and umbrella organisations representing industry, and academic publications. Regarding interviews and study visits, for the entire project, five face-to-face interviews and two study visits were performed and the interviews were voice recorded. Each of these meetings and visits lasted between approximately 40 and 110 minutes. Three of the interviews and both of the visits were performed focussing on and with representatives for the Swedish system, while two of the interviews were performed focussing on and with representatives for the Dutch system. Representatives both of national governmental agencies and of organisations governed by commercial actors in the product chain upstream of consumption were interviewed in each of the countries.

2.5. Delimitations and definitions

Delimitations for this study are made regarding allocation, regarding definition of different terms, and regarding different foci selected for the study. Allocation regards which metal packaging to allocate to each country (1). Definitions are handled for packaging (2), for metal packaging (3), for waste generated (4), and for the term recycling and related terms (5). The foci concern which parts and aspects of the metal packaging systems to cover in more depth for each of the countries (6), as well as the temporal focus (7).

First, on allocation, generally, the packaging consumed in the respective countries are used as the starting point. This means that products with or of metal packaging sold to final buyers in the country but which through its upstream product chain are subject to import are covered. The corresponding exports are, on the other hand, not covered since these are seen as allocated to the country for their final consumption of the packaged and packaging products. This follows the allocations made by EU, Swedish and Dutch policy and official reporting on packaging (e.g., Directive 94/62/EC 2013, Naturvårdsverket 2012a, p. 60, Nedvang 2012, p. 20). The metal packaging waste collection to recycling systems have been of specific focus in this study, as outlined in the introductory part of chapter 3, and therefore it is here relevant to state that the processes between the sales of final metal packaging products and metal waste collection have not been indicated to be an issue and is not reported in the EU data available (see Eurostat 2012). Therefore, for each of the two countries here studied, sales of final metal packaging products have been treated as covering the same packaging that is generated as waste and waste collected within the respective countries.

Second, the metal packaging coverage considered in this study follows the general packaging definitions from the public policies on packaging and packaging waste in Sweden and the Netherlands (see SFS 2006:1273, VROM 2005). These in turn follow the EU directive on packaging and packaging waste. This directive covers the functions of containing, protecting, handling, delivering, and presenting products, except for road, rail, shipping and aviation containers which are excluded. Further, it covers primary, secondary and transport packaging. Primary packaging refers to packaging that supports one unit of a product. Secondary packaging covers group packaging supporting several units of a product. Finally, transport packaging is packaging that is designed to support and facilitate handling and transport of products or products contained in secondary packaging. (Directive 94/62/EC 1994)

Third, regarding the packaging fraction metal packaging, the approach used in this report follows the main division of the packaging and its waste that has been made in the EU directive on packaging and packaging waste. It is there divided into six groups based on the material base: glass, plastics, paper and cardboard, metals, wood, and other (Directive 94/62/EC 1994).

Fourth, metal packaging *waste generated* will be referred to by that term in this report. In the EU reporting this term is used. However, it seems to be the same amounts as those *placed on the markets* during the corresponding year in Sweden and the Netherlands, and in their national reporting they use terms equivalents to the term placed on the market – '*tillfört marknaden*' in Swedish, and '*op de markt gebrachte*' in Dutch. (See, e.g., Eurostat 2014c, Naturvårdsverket 2012a, p. 60, Nedvang 2012, p. 29)

Fifth, since there has been found to exist a potential confusion about how the term recycling and related terms are used, it will here be clarified how the term is used in this report. It is here used according to the definition applied in the EU directive on packaging and packaging waste. The corresponding terms used in the Swedish and Dutch versions of the directive are '*materialutnyttjande*' and '*recycling*', respectively (Directive 94/62/EC 2013, p. 8, Direktiv 94/62/EG, p. 8, Richtlijn 94/62/EG, p. 8). Recycling covers, according to this directive, "reprocessing in a production process of the waste materials for the original purpose or for other purposes including organic recycling but excluding energy recovery" (Directive 94/62/EC 2013, p. 8). *Energy recovery* corresponds to the Swedish term '*energiutvinning*', and to the Dutch term '*terugwinning van energie*' (Directive 94/62/EC 2013, p. 8, Direktiv 94/62/EG, p. 8, Richtlijn 94/62/EG 2013, p. 8).

This use of the term recycling and its Swedish and Dutch corresponding terms is subject to two potentially confusing issues: the unclear relation between the Swedish term '*materialåtervinning*' used in another EU waste policy as corresponding to recycling and the Swedish term 'materialutnyttjande', and the use of '*återvinning*' as the Swedish term corresponding to the term *recovery*. The term recycling is defined in the EU directive on waste as "any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations" (Directive 2008/98/EC, p. 10). The corresponding terms used in the Swedish and Dutch versions of the directive are 'materialåtervinning' and 'recycling', respectively (Directive 2008/98/EC, p. 10, Direktiv 2008/98/EG, p. 10, Richtlijn 2008/98/EG, p. 13).

A comparison between the use of the terms 'materialutnyttjande' and 'materialåtervinning' in the Swedish ordinance on producer responsibility for packaging and its use in the official reporting on packaging and packaging for the year 2010 seems to give a confusing result. The term 'materialutnyttjande' is used in the ordinance (SFS 2006:1273), while the term 'materialåtervinning' is used where the official reporting for 2010 refers to this ordinance (Naturvårdsverket 2012a, pp. 18–20). At the same time, with regards to goals for waste from construction and demolition, this latter report uses 'materialutnyttjande' as a concept that includes a broader range of operations than 'materialåtervinning', while not stating how the terms differ (Naturvårdsverket 2012a, p. 20).

Further, the term recycling, as used in this report, differs from the term recovery, which in the EU directive on packaging and packaging waste is defined (through the EU directive on waste) as covering, among other, energy generation from for example a fuel and different forms of recycling (Directive 2006/12/EC, 2008/98/EC, p. 24, 75/442/EEC 2003, Annex IIB, 75/442/EEC 2006, 94/62/EC 2013, p. 8). The corresponding terms used in the Swedish and Dutch versions of this directive are 'återvinning' and *'terugwinning'*, respectively (cf. Directive 94/62/EC 2013, p. 8, Direktiv 94/62/EG, p. 8, Richtlijn 94/62/EG, p. 8). The term 'återvinning' is in everyday Swedish language used as a translation of the English term recycling and its use here as not corresponding to that term may therefore be confusing.

Sixth, regarding the focus in this study with respect to the two countries considered, it has differed between them. This is due to their differing ways of organising the management and reporting on packaging and packaging waste, and since relevant findings were sought for rather than comparability as a means in itself.

Seventh, and finally, the general temporal focus of the project has been to use the present, the near past, and the near future as the point of departure, and to relate it to relevant information on the packaging systems during approximately the last two decades. Initially a two decades retrospective was planned to be used as the frame. However, at a relatively early stage of the study, it became clear that the most fruitful results, both practically and academically, seemed to be achieved by relating to near time situations. Three main reasons for such a focus were at hand. First, there was found to be a lack of detailed descriptions of the past packaging and packaging recycling systems in the publicly available sources. Second, since a central part of the study was interviews, discussions, and study visits with key actors in the packaging systems, this study could potentially influence actors that have an actual possibility to impact the systems. This would, however, require a certain amount of focus on current conditions and events, in order to be of relevance for these actors. Third, a focus on present activities allowed for a deeper study, since the present and near present information lacking in publicly available sources could be discussed and observed more easily and accurately than such information of more remote temporality.

3. MATERIAL FLOWS OF METAL PACKAGING

In this chapter, an overview is given of the metal packaging and metal packaging collection and recycling systems in Sweden and the Netherlands. The purpose of this overview is two-fold. It aims to provide a source of information in itself, regarding the general characteristics of these systems. Further, it is designed to supply the in chapter 4 presented management practices of potential environmental relevance with a frame against which they can be understood and evaluated.

The material and energy flows considered in this study are, as described in the previous chapter, included based on the life cycle assessment perspective. For these metal packaging systems in each of the two countries material recycling is the type of management focussed and reported on in order to create a less environmentally impacting system. Thereby, their product and management chains consist of three basic parts, where also transports are considered. In Figure 3, further on in this chapter, these parts and their main processes are graphically presented. The first part is production of raw steel and aluminium and other metal packaging components from virgin materials, reaching for example from steel production upstream to iron ore mining. The second part stretches from basic packaging components to final use, via packaging production, and for the metal packaging where it applies filling and retail. The third part covers waste management leading to recycling, as well as the steps leading to creation of the waste from metal packaging that is currently reported to not be recycled. Current waste management in the two countries leading to recycling includes for each stream of metal packaging waste at least two processes from the following process types: sorting, collection, and solid waste incineration with extraction of residues from metal packaging waste in the resulting ashes.

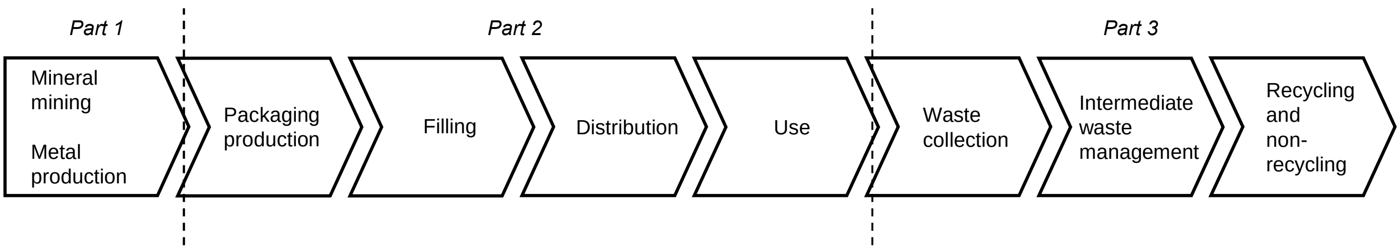

Figure 3. The three parts and their main processes in the metal packaging systems in Sweden and the Netherlands.

In the remainder of this chapter further information is presented about these product chains and the related environmental flows. First, an overview is given of the basic flows, such as metal packaging waste amounts generated per year. Second, the available information on methods and paths used between metal packaging waste generation and its recycling is outlined. Third, and finally, the environmental aspects of the metal packaging systems are covered.

3.1. Basic flows

In this sub-chapter, the basic flows of the Swedish and Dutch metal packaging systems are presented. This includes yearly amounts of metal packaging waste generated; the division between steel and aluminium packaging; the different types of metal packaging products produced; amounts of metal packaging waste and of packaging waste in relation to total waste flows, respectively; and recycling amounts and levels. Overviews and additional remarks on data accuracy for estimated amounts of packaging waste generated, recycled, and non-recycled are also provided in Table 5, Figure 4, and Figure 5, further on in this sub-chapter.

Table 5. Estimated amounts of metal packaging waste for the year 2011. [1), 2)]

	Total per country		Total per capita		Share of the waste generated for respective country	
	Sweden	The Netherlands	Sweden	The Netherlands	Sweden	The Netherlands
	kton	*kton*	*kg*	*kg*	*%*	*%*
Waste generated	61	193	6.5	11.6	100	100
Recycled	46	176	4.9	10.6	75	91
Non-recycled	15	17	1.6	1.0	25	9

[1)] Data based on Eurostat (2014c, e), Naturvårdsverket (2012a, p. 60), and Nedvang (2012, p. 20, 77–79). Packaging statistics from 2012 and onwards not yet published, as of 27 February 2014.
[2)] On inaccuracy issues of this data, see chapter 4.

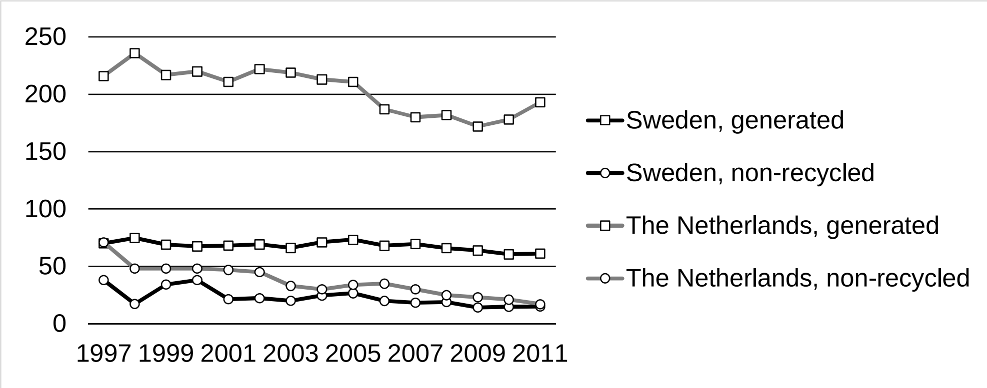

Figure 4. Estimated metal packaging waste amounts per year. Waste generated represents the sum of recycled and non-recycled amounts. Amounts in kton.[2,3,4]

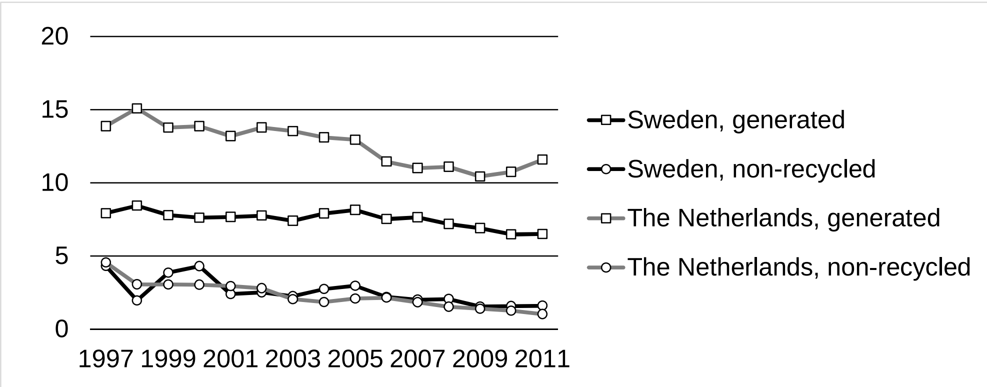

Figure 5. Estimated metal packaging waste amounts per capita and year. Waste generated represents the sum of recycled and non-recycled amounts. Amounts in kg.[5,6,7]

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[2] Based on data from Eurostat (2014c), Naturvårdsverket (2012a, p. 60), and Nedvang (2012, p. 20, 77–79). Packaging statistics for 2012 and onwards not yet published, as of 27 February 2014.

[3] The data is also presented in Appendix B, in Table A.3.

[4] The amounts are extracted from official statistics, but should nevertheless be seen as approximations only. This is due to the general inaccuracy of metal packaging statistics. Large amounts of reuse of metal packaging were reported in Sweden during the late 1990s (cf. Naturvårdsverket 1997, Bilaga 2, pp. 6–8, 1998, p. 31, 2000, p. 41), for example, 22 kton for 1996 (cf. Naturvårdsverket 1997, Bilaga 2, pp. 6–8), but this is not included for any year in the statistics that this figure is based on, partly since the reuse in different systems were calculated using different and incommensurable methods (see Naturvårdsverket 2001, p. 18). The total Swedish amounts for 2004 and 2005 includes amounts of estimated packaging from producers that are not part of the national collection and reporting system – 5 kton for 2004, and 7 kton for 2005. See also chapter 4 on data inaccuracy.

[5] Based on data from Eurostat (2014c, e), Naturvårdsverket (2012a, p. 60), and Nedvang (2012, p. 20, 77–79). Packaging statistics for 2012 and onwards not yet published, as of 27 February 2014.

Amounts of waste generated

Amounts of metal packaging waste generated is here outlined both for the latest reporting year and in retrospect for a longer time period. Both additional overviews of and remarks on data accuracy for estimated amounts of packaging waste generated are provided in Table 5, Figure 4, and Figure 5, earlier in this sub-chapter. Regarding total metal packaging waste generated recently, the reported estimated amounts for 2011 were 61 kton for Sweden, and 193 kton for the Netherlands. Based on these figures, estimated per capita amounts generated during that year have been calculated to be 6.5 kg/capita for Sweden, and 11.6 kg/capita for the Netherlands (Eurostat 2014c, e, Naturvårdsverket 2012a, p. 60, Nedvang 2012, p. 20).

Regarding the annual estimated generated amounts of metal packaging waste from a longer time perspective, the reported estimates seem to indicate a slight decreases for Sweden and for the Netherlands between 1997 and 2011, but due to data inaccuracy issues these indications should be interpreted with caution. The estimated Swedish amounts were on average 71 kton/year during 1997–1999, 62 kton/year during 2009–2011, and varied between 60 kton/year and 75 kton/year during 1997–2011. The estimated Swedish per capita amounts were on average 8.1 kg/year during 1997–1999, 6.6 kg/year during 2009–2011, and varied between 6.5 kg/year and 8.4 kg/year during 1997–2011. (Eurostat 2014c, e, Naturvårdsverket 2012a, p. 60) The estimated Dutch amounts were on average 220 kton/year during 1997–1999, 180 kton/year during 2009–2011, and varied between 170 kton/year and 240 kton/year during 1997–2011. The estimated Dutch per capita amounts were on average 14 kg/year during 1997–1999, 11 kg/year during 2009–2011, and varied between 10 kg/year and 15 kg/year during 1997–2011. (Eurostat 2014c, e, Nedvang 2012, p. 20)

Steel and aluminium

Generally, a materials division of metal packaging is made between steel and aluminium packaging (cf. SKB 2007, SOU 2012:56). For Sweden, estimated steel packaging waste generated in 2011 was reported to amount to 35 kton and to 3.7 kg/capita, which represents 57% of the country's estimated amount of metal packaging waste generated that year. The corresponding figures for aluminium were 26 kton, 2.8 kg/capita, and 43%. (Eurostat 2014b, c, d)

For the Netherlands, the amounts and shares of packaging waste generated of steel and of aluminium are here estimated based on calculations using reported estimates. Shares of steel and aluminium packaging are reported for household waste and for waste from organisations, respectively. Of the household metal packaging waste amounts, 78% of the weight is report-

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[6] The data is also presented in Appendix B, in Table A.4.

[7] See remarks about inaccuracy issues of the data in the footnotes of Figure 4.

ed to consist of steel packaging and the remainder of aluminium packaging. Of the metal packaging waste from organisations, the packaging for logistics purposes is reported to be steel packaging. In the residual metal packaging waste from organisations, the weight share of aluminium has based on market information been reported to be 10% of the aluminium weight share reported for household metal packaging waste. (Nedvang 2012, p. 78) For the calculations made in the following, it has been assumed that these shares for metal packaging waste from organisation also apply to the metal packaging waste from organisations that is sorted at the waste source and that is not packaging for logistics purposes.

The subsequent calculations are based on that the Dutch amount of household metal packaging waste is estimated to be the at the waste source sorted amounts not being specified as either being packaging for logistics purposes or being specified to originate from organisations plus the amounts not sorted at the waste source that are not from organisations. According to this procedure, the household packaging waste amounts sorted at the waste source becomes 8 kton according to Nedvang (2012, p. 78, 'Tabel J.2'). The amount of metal packaging waste not sorted at the waste source was reported to be 93 kton for 2011. Out of this amount, 3.5 kton has here been calculated to originate from organisations. This is based on that the 14 kton of the metal packaging waste from organisations that is estimated to be sorted at the source of waste and that is not for logistics purposes is assumed to be the stated 80% of the metal packaging waste from organisations that is sorted at the waste source (see Nedvang 2012, pp. 78–79). The amount of metal packaging waste from households thus becomes 98 kton. The amount of metal packaging waste from organisations not being transport packaging then becomes 18 kton, and is calculated as the sum of the in this paragraph introduced 3.5 kton and 14 kton. Applying the in this sub-sub-chapter presented shares of steel and aluminium, steel packaging waste generation in 2011 in the Netherlands amounted to 170 kton, 10 kg/capita, and to 89%, while the corresponding figures for aluminium are 22 kton, 1 kg/capita, and 11%. (Eurostat 2014c, e, Nedvang 2012, pp. 77–79) As a comparison, a report from 2007 from the sector organisation Stichting Kringloop Blik (SKB) stated that the Dutch metal packaging consisted of 90% steel packaging and 10% aluminium packaging (SKB 2007).

Types of products

Regarding types of products, information has here been combined from official statistics, official annual packaging reports, and from other estimates.

In Sweden, for steel packaging, a report published in 1994 could be used for an approximation of the relations between generated waste amounts of different product groups. However, no far reaching conclusions should be made from the estimated and possibly outdated figures from that report.

Total yearly amounts were estimated to be 66 kton (whereof 6 kton were reused). Thereof, 40 kton of tin coated consumer packaging was estimated to have been generated as waste in 1992, which is 60% per weight of this estimated total Swedish steel packaging waste generation. This amount of consumer packaging included food cans; approximately 12 kton of paint buckets, corresponding to 18% of the estimated Swedish steel packaging waste generation; cans for lubrication oil and for other chemical-technical products; as well as lids and bottle caps mainly for glass containers. In addition to these small containers, approximately 26 kton (whereof 6 kton were reused) of industrially used barrels was reported to be used annually, which represents 40% of the estimated Swedish steel packaging waste generation. (Statens naturvårdsverk 1994, pp. 22–32)

Regarding Swedish product types of aluminium packaging, 17 kton of beverage packaging waste made of aluminium were reported to have been generated in 2010 (Naturvårdsverket 2012a, p. 69). This represents 68% of the total estimated amount of aluminium packaging waste generated during that year in Sweden, (Naturvårdsverket 2012a, p. 60, 69). For other products of aluminium packaging the corresponding figures were 8 kton and 32% (Naturvårdsverket 2012a, p. 60, 65).

In the Netherlands, the metal packaging for logistics purposes was reported to generate 58 kton of waste in 2010. This represents 32% of the estimated Dutch metal packaging waste generation that year. Of this packaging, 32 kton was reported to be steel barrels, 20 kton was reported to be larger buckets, and 6 kton was reported to be intermediary bulk containers (IBCs). Their shares of the estimated Dutch metal packaging waste generated in 2010 were 18%, 11%, and 3%, respectively. (Nedvang 2011) Regarding Dutch household metal packaging waste generated in 2010, Stichting Kringloop Blik stated that it approximately could be divided into 65% of food cans, 20% of beverage packaging, and 15% of containers for paint and other containers, respectively (SKB 2013). An estimate of these product groups' shares of the total metal packaging waste generated in the Netherlands can be calculated based on the 100 kton of metal packaging stated to be consumed by Dutch households during 2010. When this amount is related to the total estimated Dutch metal packaging waste generation reported for 2010, their shares, by weight, become 37%, 11%, and 8%, respectively, and 56% in total. (Nedvang 2011, SKB 2013)

Waste amounts generated compared to other waste

In order to relate metal packaging waste to other waste streams, the amounts of it generated and the amounts of total packaging waste generated are here compared to the generation of other waste. These figures are intended to indicate (metal) packaging waste's share of the amounts needed by society to be waste managed.

The amounts, by weight, of metal packaging waste estimated to be generated are here compared to corresponding figures for all waste and for all packaging waste generated in Sweden and the Netherlands, respectively. The figures are also presented in Table 6, further on in this sub-sub-chapter. Of the total amounts of waste reported to be generated within the respective countries during 2010, estimated metal packaging waste amounted to 0.05% for Sweden, and to 0.15% for the Netherlands. Of total packaging waste amounts estimated to be generated within the respective countries during the same year, metal packaging estimates amounted to 4.8% for Sweden, and to 6.5% for the Netherlands. The, for 2010 estimated, total packaging waste share, of total waste generation, amounted to 1.1% for Sweden, and to 2.3% for the Netherlands, respectively. It can be noted that a large part of the total amounts of waste was reported to consist of waste from mining, quarrying, and construction: 84% in Sweden and 66% in the Netherlands. If these waste sources are excluded, the metal packaging estimates represented, by weight, 0.31% of the remaining Swedish waste, and 0.43% of the remaining Dutch waste. Correspondingly, estimated total packaging waste represented 6.5% of the remaining Swedish waste, and 6.6% of the remaining Dutch waste. (Eurostat 2013a, Naturvårdsverket 2012a, p. 60, Nedvang 2011, p. 21–22)

Table 6. Comparison of amounts by weight in different waste streams to metal packaging and to packaging, respectively. For 2010. [1)]

		Sweden	The Netherlands
		%	%
Estimated metal packaging waste generation's share of …	total waste	0.05	0.15
	total waste excluding mining and quarrying, and construction	0.31	0.43
	packaging waste	4.84	6.53
Estimated packaging waste generation's share of …	total waste	1.06	2.28
	total waste excluding mining and quarrying, and construction	6.49	6.64

[1)] Data based on Eurostat (2013a), Naturvårdsverket (2012a, p. 60), and Nedvang (2011, p. 21–22). Statistics from the EU not available for more recent years than 2010 for total waste, for mining and quarrying, and for construction, as of 27 February 2014

Recycling

In this sub-sub-chapter, the reported figures on metal packaging recycling and non-recycling are presented. First, the latest reported amounts and rates are outlined. This is followed by an outline of the longer time perspectives on non-recycling, both for total estimated metal packaging in the respective countries, and for estimated aluminium packaging other than beverage packaging for Sweden since it has been indicated to be of particular interest.

The reported amounts of metal packaging waste recycled in 2011 were 46 kton for Sweden, and an estimated 176 kton for the Netherlands (Eurostat 2014c, Naturvårdsverket 2012a, p. 60, Nedvang 2012, pp. 77–79). Based on these figures, the Swedish reported per capita amount recycled was 4.9 kg/year and its estimated recycling rate 75%, while the corresponding figures for the Netherlands were 10.6 kg/year and 91%. Further, reported recycling in Sweden in 2011 was for steel packaging 29 kton, 3.1 kg/capita and an estimated 83%, and for aluminium 17 kton, 1.8 kg/capita and an estimated 66%. (Eurostat 2014b, c, d, e, Naturvårdsverket 2012a, p. 60, Nedvang 2012, pp. 77–79) For aluminium recycling in Sweden during 2010, it was for beverage packaging reported to be 15 kton, 1.6 kg/capita, and an estimated 87% (Eurostat 2014e, Naturvårdsverket 2012a, p. 69). Recycling of other aluminium packaging in Sweden during the same year has been reported to be 2 kton, 0.2 kg/capita, and an estimated 23% (Eurostat 2014e, Naturvårdsverket 2012a, p. 60, 65). For the Netherlands, recycling amounts for packaging of steel and of aluminium, respectively, have been calculated based on reported rates of metal packaging waste recycled in 2010 from the ashes after residual waste incineration for each of the two metal types. The figures have been modified for the 2011 figures available by changing the non-recycling shares of the two metal types with equal proportions and with a minor final adjustment equally of the residual metal packaging waste fractions from households and from organisations, respectively. The resulting recycling estimates becomes for 2011, for steel 162 kton, 9.7 kg/capita, and 95%, and for aluminium 14 kton, 0.8 kg/capita, and 63%.

Correspondingly, the estimated amounts of metal packaging waste not recycled in 2011 were 15 kton for Sweden, and 17 kton for the Netherlands (Eurostat 2014c, Naturvårdsverket 2012a, p. 60, Nedvang 2012, pp. 20, 77–79). Based on these figures, the estimated per capita amounts not recycled have been calculated to have been 1.6 kg/year for Sweden and 1.0 kg/year for the Netherlands, and the estimated non-recycling rates to be 25% for Sweden and 9% for the Netherlands. Further, non-recycling in Sweden in 2011 has been estimated to have been 6 kton, 0.6 kg/capita and 17% for steel packaging, and for aluminium packaging 9 kton, 0.95 kg/capita and 34%. (Eurostat 2014b, c, d, e, Naturvårdsverket 2012a, p. 60, Nedvang 2012, pp. 20, 77–79) For non-recycling of aluminium in Sweden during 2010, it has for beverage packaging been calculated to have been 2 kton, 0.2 kg/capita, and 13% (Eurostat 2014e, Naturvårdsverket 2012a, p. 69). Non-recycling of other aluminium packaging in Sweden during the same year has been estimated to have been 6 kton, 0.6 kg/capita, and 77% (Eurostat 2014e, Naturvårdsverket 2012a, p. 60, 65). For the Netherlands, estimated non-recycling has based on the figures in the previous paragraph been calculated to be, for steel 9 kton, 0.5 kg/capita, and 5%, and for aluminium 8 kton, 0.5 kg/capita, and 37%.

Regarding the annual amounts of metal packaging not recycled from a longer time perspective, the estimated Swedish amounts were on average 30 kton/year and 42% during 1997–1999, 17 kton/year and 24% during 2009–2011, and varied between 14 kton/year and 38 kton/year and 22% and 55% during 1997–2011. The estimated Swedish per capita amounts were on average 3.4 kg/year during 1997–1999, 1.6 kg/year during 2009–2011, and varied between 1.5 kg/year and 4.3 kg/year during 1997–2011. (Eurostat 2014c, e, Naturvårdsverket 2012a, p. 60) The estimated Dutch amounts were on average 56 kton/year and 25% during 1997–1999, 20 kton/year and 11% during 2009–2011, and varied between 20 kton/year and 70 kton/year and 9% and 33% during 1997–2011. The estimated Dutch per capita amounts were on average 3.6 kg/year during 1997–1999, 1.2 kg/year during 2009–2011, and varied between 1.0 kg/year and 4.6 kg/year during 1997–2011. (Eurostat 2014c, e, Nedvang 2012, pp. 20, 77–79)

Both additional overviews of and remarks on data accuracy for estimated amounts of total metal packaging waste recycled, and non-recycled are provided in Table 5, Figure 4, and Figure 5, earlier in this sub-chapter.

Finally, the trends of the estimates of non-recycled aluminium packaging other than beverage packaging in Sweden are here outlined, and are also illustrated in Figure 6 and Figure 7, further on in this sub-sub-chapter. It has been found to be of particular interest due to a combination of high estimated non-recycling rates, of estimated considerable amounts of non-recycled waste, and of the indicated large environmental impacts per kg produced non-recycled aluminium packaging described in the sub-chapter 3.3. The estimated non-recycled per capita amounts were on average 0.79 kg/year and 71% during 2004–2006, 0.56 kg/year and 66% during 2008–2010, and varied between 0.49 kg/year and 0.88 kg/year and 58% and 77% during 2004–2010 (Naturvårdsverket 2012a, pp. 60, 65). However, these trend indications ought to be interpreted with caution, not least since the 2004–2005 values include additional estimates of amounts from suppliers not registered in the national collection and reporting system (Naturvårdsverket 2012a, pp. 64–65). These and other data inaccuracy issues are further described in the footnotes of Figure 6, further on in this sub-sub-chapter, and of Figure 4, earlier in this sub-chapter. Despite these inaccuracy issues, considerable amounts and shares of aluminium packaging other than beverage packaging seem in Sweden not to be recycled throughout the period 2004–2010, based on this overview. In addition, these amounts seem according to the latest reported figures to be considerably larger than the non-recycled aluminium beverage packaging waste while the latter total waste stream of the latter one is significantly larger than the former one.

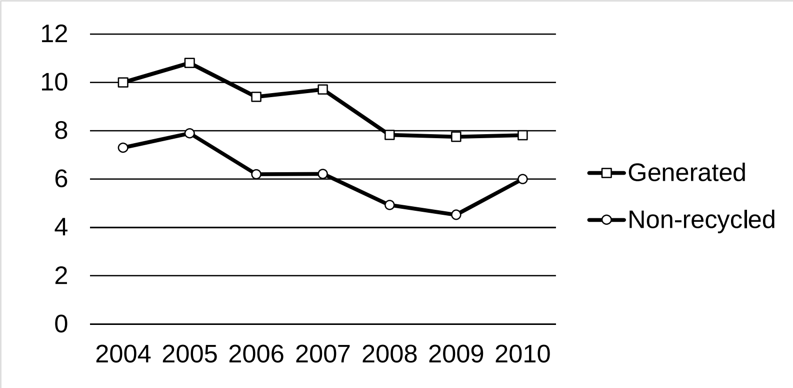

Figure 6. Estimated aluminium packaging excluding beverage packaging, in Sweden per year. Waste generated represents the sum of recycled and non-recycled amounts. Amounts in kton.[8][9][10][11][12][13]

[8] Based on data from Naturvårdsverket (2012a, p. 60, 65). Statistics for 2011 and onwards not yet published, as of 27 February 2014.

[9] The data is also presented in Appendix B, in Table A.5.

[10] The amounts of generated waste for 2004 and 2005 includes amounts of estimated packaging from producers that are not part of the national collection and reporting system. These amounts are 1 kton for 2004, and 1.4 kton for 2005. (Naturvårdsverket 2012a, pp. 64–65)

[11] Regarding the figures for 2010. TMR AB reported only one aggregated figure for steel and aluminium packaging taken together. In addition, the calculation of recycling by Svenska Metallkretsen has been performed using a new approach starting from the reporting on the 2010 results. In this new approach, aluminium cans that were not returned through the aluminium can refund system are accounted for as aluminium cans. (Naturvårdsverket 2012a, p. 64)

[12] Regarding the figures for 2010. The sum of the amounts of waste reported separately to be generated of steel and aluminium packaging excluding aluminium cans is 0.159 kton lower than the reported sum. The sum of the amounts reported separately to be recycled of steel and aluminium packaging excluding aluminium cans is 0.845 kton lower than the reported sum. (Naturvårdsverket 2012a, pp. 64–65)

[13] See also remarks about inaccuracy issues of the data in the footnotes of Figure 4 and in chapter 4.

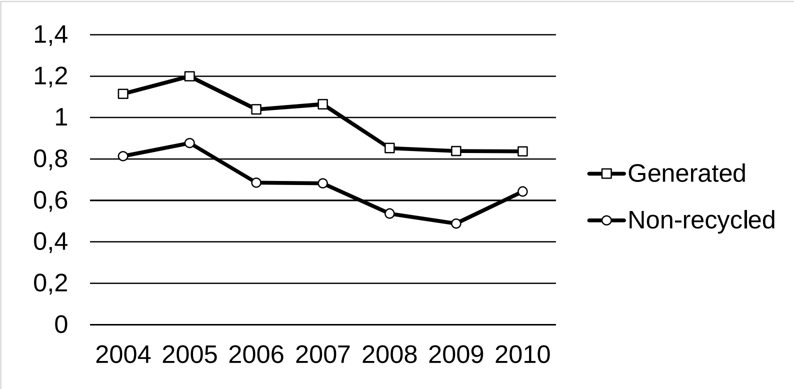

Figure 7. Estimated aluminium packaging excluding beverage packaging, in Sweden per capita and year. Waste generated represents the sum of recycled and non-recycled amounts. Amounts in kg.[14][15][16]

3.2. *From waste generation to recycling*

Material recycling seems to be the main focus in Sweden and the Netherlands, as well as at the EU level, for decreasing environmental impacts from packaging, and particularly for metal packaging (cf. Directive 94/62/EC 1994, Naturvårdsverket 2012a, Nedvang 2012, SOU 1991:76, Tillman et al. 1991). Therefore, the focus of this study largely lies on the details between waste generation and recycling. These are found in part three among the three product and management chain parts outlined in the beginning of this chapter. However, the life cycle environmental performance of these activities are highly dependent on particularly part one, as shown in the outline of environmental aspects in the subsequent sub-chapter.

Methods of separating metal packaging waste from other waste

Metal packaging collection and recycling systems vary both between the two countries and within each of them. The first distinction made here is between systems for sorting at the waste source, and *separation downstream of the waste source*. As introduced in chapter 1, sorting at the waste source is here used to denote the product user's disposal of metal packaging into a waste stream typically only containing metal packaging waste. Downstream separation is here used to represent separation of metal packaging waste from other waste by an actor located downstream of the product user. It includes separation either after or before incineration of solid waste that

[14] Based on data from Eurostat (2014e), and Naturvårdsverket (2012a, pp. 60, 65). Statistics for 2011 and onwards not yet published, as of 27 February 2014.
[15] The data is also presented in Appendix B, in Table A.5.
[16] See remarks about data inaccuracy in the footnotes of Figure 6 and of Figure 4.

contains metal packaging waste as well as other waste (cf. SFS 2006:1273, SKB 2007, Statens naturvårdsverk 1994). In Swedish national regulations, sorting at the waste source is the requested method. For aluminium beverage packaging, this has been separately regulated through an ordinance on recycling systems for plastic bottles and metal cans. A system with a refund fee for consumers is in place for this waste stream. (Returpack n.d., SFS 2005:220, see also SFS 2006:1273) Since the mid-1990s sorting at the source has also been the requested method for other metal packaging in Sweden (SFS 1994:1235, 1997:185, 2006:1273). Parts of the Swedish metal packaging waste used to be recycled via extraction from ashes after incineration (see Statens naturvårdsverk 1994) and Thord Görling at the Packaging and Newspaper Collection Service expressed that he was not sure whether such extraction in practice was ongoing, when interviewed in this study (see Görling per. comm. 2013a).

In the Netherlands, sorting at the waste source is currently (2011) reported to be performed for 100% of the 78 kton of metal packaging reported to be used for logistics purposes, for an estimated 14 kton of other metal packaging from non-households, for 2 kton of food containers, and for 6 kton of small scale chemical waste (Nedvang 2012). A small share of the approximately 400 Dutch municipalities currently seem to apply sorting at the waste source for tin cans and similar metal packaging (cf. Nedvang 2012, Verweij per. comm. 2013). The composition of the streams that these fractions are collected in vary, from covering only metal packaging to including other metal waste as well (Nedvang 2012). According to a report from Stichting Kringloop Blik in 2007, approximately 20% of metal food containers were collected in these collection systems where they were available, while the remaining 80% of the metal food containers in these areas were deposited as residual waste (SKB 2007). A larger number of systems for sorting food cans and other household metal packaging waste at the source used to be in place in the Netherlands. Whether sorting at the source is used for metal packaging waste is decided by each municipality in the Netherlands. (Verweij per. comm. 2013)

The other reported method of sorting currently used in the Netherlands is separation from bottom ashes after incineration of residual solid waste containing, among other, metal packaging waste (Nedvang 2012, p. 78). The method of separating metal packaging waste from residual solid waste before incineration was previously applied in the Netherlands, and in a report from 2007 it was stated to be performed at three out of the then thirteen waste treatment plants in use in the Netherlands (SKB 2007; cf. Ter Morsche per. comm. 2013, Verweij per. comm. 2013).

Further dividing into paths from waste generation to recycling

The complexity regarding the number of different paths from waste generation to re-melting for the production of new metal products is here addressed. In Sweden, a considerable number of such paths have been identified while only one of them is reported on separately and the other ones are reported on as one compound, and therefore they are in the following outlined. On the other hand, packaging of steel and of aluminium, respectively, are reported on separately for Sweden. In the Netherlands, two main paths between from waste generation and re-melting are identifiable from the official statistics, for metal packaging waste collected with residual waste and for transport packaging respectively. Three smaller fractions of metal packaging waste sorted at the source are also reported on separately. At the same time, packaging of steel and of aluminium, respectively, are not clearly reported on for the Netherlands, although the amounts of waste of them generated seem possible to with some effort deduct from the official reporting. Their respective recycling is, however, not possible to deduct only from the reporting for 2011. See sub-chapter 3.1 for more details on all the here outlined Dutch fractions.

In Sweden, eight main types of paths for metal packaging waste handling leading towards recycling have been distinguished in this study, whereof one for aluminium beverage packaging that also is the only one separately reported on, three additional for households, and four additional for organisations (companies, public bodies, etc.):

For aluminium beverage packaging:

- Via a refund collection system (cf. Returpack n.d., SFS 2005:220).

For other household metal packaging waste:

- Via unmanned *intermediate scale facilities* for sorting at the waste source (in Swedish: 'återvinningsstationer') (Naturvårdsverket 2009, p. 33, 2012a, p. 59, SOU 2012:56, p. 89).
- Via *small scale facilities* in close proximity of the households for sorting at the waste source (in Swedish: *'fastighetsnära insamling'*) (Naturvårdsverket 2009, p. 33, 2012a, p. 59, SOU 2012:56, p. 89).
- Via municipal *large scale facilities* for sorting at the waste source (in Swedish: *'återvinningscentraler'*) (SOU 2012:56, p. 89).

For other metal packaging waste from organisations:

- For barrels via a separate collection system (FTI n.d. b, Görling per. comm. 2013a, p. 6).
- Via different recycling companies to be selected on a free market basis (SOU 2012:56, p. 89).
- Via sorting facilities where organisations free of charge may deposit at a maximum one cubic metre per visit (Naturvårdsverket 2012a:59; cf. SOU 2012:56, p. 89).
- For small organisations via the households' intermediate scale facilities (SOU 2012:56, p. 89).

3.3. Environmental aspects

From a life cycle assessment perspective, the currently most emphasised and most obvious environmental issues related to metal packaging, and which were also mentioned in chapter 1 as environmental issues of general packaging, are (see, e.g., Directive 94/62/EC 1994, pp. 10–11, 2013, pp. 3–4, Rouw & Worrell 2011, p. 483, SKB 2007, pp. 12–18):

- Energy use.
- Global warming potential.
- Raw material scarcity.
- Landfill space scarcity.

Energy use

Energy use is particularly large during the processes of producing crude steel from iron ore (cf., SKB 2007, p. 14, Tillman et al. 1991), and during the Bayer and Hall-Heroult processes used for extracting aluminium from bauxite (SKB 2007, p. 16, Tillman et al. 1991). Proxies are here used for quantifying life cycle assessment based energy use, and an overview of the resulting figures is presented in Table 7, further on in this sub-sub-chapter. For steel packaging, proxies for life cycle energy use are here calculated based on almost cradle-to-gate figures, where mining is not included, for crude steel presented by the World Steel Association in 2008 and recalculated using the ratio between crude and finished steel production in 2011. One kg of steel packaging is assumed to require one kg of finished steel product, and average values of each of the four presented steel production routes are used. For the 75% of world crude steel stated to be produced from ore, 66 percentage points were stated to be produced via basic oxygen furnaces with cumulated energy supply of 19.8–31.2 GJ/ton, 6 percentage points via electric arc furnaces with cumulated energy supply of 28.3–30.9 GJ/ton, and 3 percentage points via open hearth furnaces with cumulated energy supply of 26.4–41.6 GJ/ton. The weighted and for finished steel products recalculated average of the averages of each of these three routes becomes 29 GJ/ton. For the recycled steel, it was stated to be produced via electric arc furnaces with cumulated energy supply of 9.1–12.5 GJ/ton, which yields the recalculated average of 12 GJ/ton. (World Steel Association 2008, p. 2, 2012, pp. 10, 16, see also Yellishetty et al. 2011, p. 657) This corresponds to 59% lower energy supply for the recycling route than for using ore. As points of reference, this lowering of energy supply has been stated to be 75% by the Swedish organisation Packaging and Newspaper Collection Service (Förpacknings- och tidningsinsamlingen AB, FTI), and to be 65% by a report in 2007 by the Dutch organisation Stichting Kringloop Blik (SKB) (FTI n.d. a, p. 3, SKB 2007, p. 14).

For aluminium packaging life cycle energy use, cradle-to-gate primary energy demand (net cal. value) for aluminium sheets used in Europe and reported by the European Aluminium Association is here used as a proxy. For aluminium sheet from ore, this has been calculated as the sum of 157 GJ/ton cradle-to-gate primary energy demand for aluminium ingots and of 10.7 GJ/ton gate-to-gate primary energy demand for aluminium sheet production (EAA 2013, pp. 38, 40, 42). This sum is 168 GJ/ton. For aluminium sheet from recycled aluminium, this energy value has been calculated as the sum of 8.5 GJ/ton gate-to-gate primary energy demand for scrap recycling and of the 10.7 GJ/ton gate-to-gate primary energy demand for aluminium sheet production (EAA 2013, pp. 40, 42, 59). This sum is 19.2 GJ/ton. This corresponds to 89% lower energy supply for the recycling route than for using ore. As points of reference, this lowering of energy supply has by the sources used as points of reference in the previous paragraph been stated to be 95% (FTI n.d. a, p. 3, SKB 2007).

These proxy figures for life cycle assessment based energy use in packaging of steel and of aluminium are in Table 7, further on in this sub-subchapter, used to calculate further proxies. These are calculated for metal packaging in 2011, which in turn is compared to national energy use in 2011 and to potential energy savings if metal packaging recycling rates based on 2011 figures are increased to 100%. For each of the two countries, these proxies of cradle-to-grave energy use in metal packaging amounts to around 0.1% of national energy use. Non-recycled aluminium packaging represents large shares of the respective countries metal packaging cradle-to-grave proxy energy use – 64% for Sweden and 36% for the Netherlands. This can be compared to these fractions shares of reported metal packaging waste generation in 2011 in these countries (based on the figures presented in subchapter 3.1) – 15% for Sweden and 4% for the Netherlands. The saving potentials for these fractions if 100% recycling is achieved, would, based on these proxy calculations, be considerably larger than if this was achieved for the steel packaging fractions – 13 times for Sweden and 8 times for the Netherlands. Reaching a 100% recycling level for these aluminium packaging fractions would, based on the proxies here calculated, lower cradle-to-grave energy use of the metal packaging systems with 56% in Sweden and 32% in the Netherlands. (EAA 2013, pp. 38, 40, 42, 59, Eurostat 2014a, World Steel Association 2008, p. 2, 2012, pp. 10, 16, and the figures on waste generation and recycling in 2011 presented in sub-chapter 3.1) Taken together, recycling the currently non-recycled aluminium fractions seems to have the potential to theoretically result in large percentage reductions of the two countries metal packaging systems' energy use, particularly for Sweden, while representing small shares of the annual waste generation in these metal packaging systems, and while these metal packaging systems account for around 0.1% of each nations energy use.

Table 7. Proxy life cycle assessment based energy use for metal packaging in 2011, including its share of national energy use and potential savings from increasing recycling to 100%. [1), 2), 3)]

	Per kg metal	Sweden			The Netherlands		
		Per capita	National share	Saving potential per capita	Per capita	National share	Saving potential per capita
	MJ	*MJ*	*%*	*MJ*	*MJ*	*%*	*MJ*
Steel, from ore	29	19	0.01	11	15	0.01	9
Steel, recycled	12	36	0.02		120	0.06	
Aluminium, from ore	170	160	0.07	140	81	0.04	72
Aluminium, recycled	20	35	0.02		16	0.01	
Total		250	0.11		220	0.11	

1) Data based on EAA (2013, p. 38, 40, 42, 59), Eurostat (2014a), World Steel Association (2008, p. 2, 2012, pp. 10, 16), and the figures on waste generation and recycling in 2011 presented in sub-chapter 3.1.
2) Proxies used are described in the body text earlier in this sub-sub-chapter.
3) National shares are based on gross inland energy consumption.

Global warming potential

Global warming potential for metal packaging in the two countries is here treated similarly to the life cycle assessment based energy proxies in the previous sub-sub-chapter. It is compared to national global warming potentials reported and potential impact decrease of reaching 100% metal packaging recycling is calculated.

Regarding the data used, for steel packaging life cycle assessment based global warming potential figures used in an environmental life-cycle comparisons of steel production and recycling published in 2011 are here applied (see Yellishetty et al. 2011, p. 651). Aluminium packaging life cycle assessment based global warming potential figures are calculated as proxies based on the life cycle inventory report used for the energy calculations used in the previous sub-sub-chapter. Similarly, cradle-to-gate carbon dioxide equivalents for aluminium sheets used in Europe and reported by the European Aluminium Association are used as the proxy. This has for aluminium packaging from ore been calculated as the taken together impacts from cradle-to-gate aluminium ingots production and from gate-to-gate production from aluminium sheet production. For recycled aluminium packaging it has been calculated as the sum of gate-to-gate production for scrap recycling and the mentioned sheet production. (See EAA 2013, pp. 38, 40, 42, 59)

From the subsequent calculations, and as outlined in Table 8, further on in this sub-sub-chapter, these proxies of cradle-to-grave global warming potential of metal packaging amounts to around 0.2% of national global warming potential in Sweden and to 0.1% for the corresponding share for the Netherlands. Non-recycled aluminium packaging represents large shares of the respective countries metal packaging estimated cradle-to-grave global warming potentials – 63% for Sweden and 37% for the Netherlands. This can be compared to these fractions shares of reported metal packaging waste generation in 2011 in these countries (based on the figures presented in sub-chapter 3.1) – 15% for Sweden and 4% for the Netherlands. The saving potentials for these fractions if 100% recycling is achieved, would, based on these proxy calculations, be considerably larger than if this was achieved for the steel packaging fractions – 8 times for Sweden and 5 times for the Netherlands. Reaching a 100% recycling level for these aluminium packaging fractions would, based on the proxies here calculated, lower cradle-to-grave energy use of the metal packaging systems with 56% in Sweden and 32% in the Netherlands. (EAA 2013, pp. 38, 40, 42, 59, UN 2013, p. 14, Yellishetty et al. 2011, p. 651) Taken together, recycling the currently non-recycled aluminium fractions seems to have the potential to theoretically result in large percentage reductions of the two countries metal packaging systems' global warming potential, particularly for Sweden, while representing small shares of the annual waste generation in these metal packaging systems, and while corresponding to around 0.1%–0.2% of the respective country's reported inland greenhouse gas emissions.

Table 8. Proxy life cycle assessment based global warming potential in carbon dioxide equivalents for metal packaging in 2011, including its share of national global warming potential and potential savings from increasing recycling to 100%. 1), 2), 3)

	Per kg metal	Sweden			The Netherlands		
		Per capita	National share	Saving potential per capita	Per capita	National share	Saving potential per capita
	kg	*kg*	*%*	*kg*	*kg*	*%*	*kg*
Steel, from ore	2.1	1.4	0.02	1.0	1.1	0.01	0.8
Steel, recycled	0.6	1.8	0.03		5.8	0.05	
Aluminium, from ore	9.3	8.9	0.14	7.8	4.5	0.04	4.0
Aluminium, recycled	1.1	2.0	0.03		0.9	0.01	
Total		14.0	0.21		12.4	0.11	

1) Data based on EAA (2013, pp. 38, 40, 42, 59), UN (2013, p. 14), Yellishetty et al. (2011, p. 651), and the figures on waste generation and recycling in 2011 presented in sub-chapter 3.1.
2) Proxies used are described in the body text earlier in this sub-sub-chapter.
3) National global warming potential figures represent statistics from the UN.

Raw material scarcity

Raw material scarcity for both packaging of steel and of aluminium is here estimated and is also presented in Table 9, further on in this sub-sub-chapter. For steel raw material scarcity applies prominently to iron ore, bituminous coal, and tin. For aluminium it applies to bauxite. Outlines are in the following presented for iron ore and bauxite, since these were the only of the four resources that sufficiently reliable and feasible data was found for.

Raw material scarcity is here approached by comparing world production of the two metals from virgin materials per capita to the amounts of these final substances used for the packaging consumed in each of the two countries per capita, as well as by presenting current reserves and resources reported for the raw materials used. Regarding iron ore, world production is here calculated as the 2011 consumption of finished steel products multiplied by the in 2008 reported share of 75% of crude steel reported to be produced as primary steel (i.e., form virgin resources) (see World Steel Association 2008, p. 2, 2012, p. 16). Years of iron ore reserves is here calculated as calculated iron content rate for 2011 (where the different reporting model for China is taken into account) applied to actual ore production weight in 2011 and compared to iron content in ore reserves reported in January 2012 (based on USGS 2012, p 85, and World Steel Association 2013, p. 20). A corresponding calculation is here used for iron ore resources based on iron content of resources reported in January 2012 (based on USGS 2012, p 85, and World Steel Association 2013, p. 20). Regarding bauxite, world production is here calculated as reported dry weight bauxite produced from mines in 2011 divided by the weight input ratios for the production of alumina, liquid aluminium, ingots, and ingot rolling (see EAA 2013, pp. 24, 27, 29, 40, USGS 2013, p. 27). Reserves and resources years for bauxite is here calculated using the dry weight figures presented for 2011 production and for reserves and resources in January 2012 (see USGS 2012, p. 27, 2013, p. 27). This ought, however, to be related to the statement that currently not economically viable resources other than bauxite for aluminium production are essentially inexhaustible in most major aluminium producing countries (USGS 2013, p. 27).

The results of these calculations, and which are outlined in Table 9, further on in this sub-sub-chapter, give some indications. The estimated per capita level of iron ore use for metal packaging in each of the two countries is around 0.4% of the total iron ore use per capita globally. The corresponding figures for bauxite are considerably higher – 15% for Sweden and 8% for the Netherlands. Regarding number of years that the here calculated reserves and resources of the two ore types will last is on the same order of magnitude. In addition, and already pointed out, abundant not currently economically feasible resources of aluminium in other materials are stated to exist (USGS 2013, p. 27).

Table 9. Estimated resource use of iron ore and bauxite for packaging compared to other products and to reserves and resources at current production levels based on reported figures. For 2011 if not stated. [1), 2)]

	Metal products from respective ore			Reserves	Resources
	Packaging per capita		All products per capita		
	Sweden	The Netherlands	World		
	kg	*kg*	*kg*	*years*	*years*
Iron ore	0.6	0.5	150	80	220
Bauxite	1.0	0.5	6	110	250

1) Data based on EAA (2013:24, 27, 29, 40), UN (2014:4), USGS (2012:27, 85, 2013:27), World Steel Association (2008:2, 2012:16).
2) Procedures for calculating the figures are described in the body text earlier in this sub-sub-chapter.
3) Other resources than bauxite for aluminium production have been stated to be essentially inexhaustible in most major aluminium producing countries (USGS 2013, p. 27).

Landfill space scarcity

Regarding landfill space scarcity, comparisons of non-recycled metal packaging amounts are here compared to reported amounts of waste landfilled or through other methods disposed. For metal packaging in 2010, 15 kton in Sweden, and 21 kton in the Netherlands, respectively, were not recycled (Eurostat 2014b). Regarding disposal of other waste, in Sweden in 2010 it was reported that 44 Mton of mining waste and 3.3 Mton of other waste was landfilled, while at least the last figure needs to be viewed with caution since for example 4.3 Mton of waste treated externally to its producers was not categorised (Naturvårdsverket 2012a, p. 7). Correspondingly for the Netherlands, 1.8 Mton was in 2010 reported to be either landfilled or released to the environment, but also these figures ought to be viewed with caution since the total waste amount that they are based on are 60 Mton while the amount reported to the EU for that year was 120 Mton and since imports and exports only amount to a small fraction (1 Mton and 3 Mton, respectively) of this difference (see Eurostat 2013a, IenM 2013, pp. 35, 44–47).

4. MANAGEMENT PRACTICES AND THEIR POTENTIAL ENVIRONMENTAL IMPLICATIONS

In this chapter, findings from combining the material and basic organisational properties of the metal packaging systems presented in the preceding chapter, with the interviews and study visits performed through this project, are presented.

Two main groups of management practices with potential environmental influence were identified, and these are the topics of sub-chapters 4.1 and 4.2. The first of them covers potential ineffectiveness of public policy and data. The second is on conflicting arguments and perceptions. The two groups encompass the following seven areas, which also form the sub-sub-chapters of these two sub-chapters:

For public policy and data:

- 5 issues on lack of reliability of data.
- 3 issues on lack of resolution in public policy and in data.
- 1 issue on lack of comparability internationally.
- 1 issue on lack of environmental accountability.

For arguments and perceptions:

- 1 issue on metals incineration.
- 1 issue on sorting in waste streams.
- 1 issue on public-private conflicts.

The findings are also presented in three tables that are discussed in sub-chapter 4.3. In Table 10 and Table 11, further on in this chapter, overviews of the findings are presented using the scope framework introduced in chapter 2. In Table 12, further on in this chapter, the findings are analysed using the characteristics framework also introduced in chapter 2.

Table 10. Environmental findings on metal packaging – grouped based on their scopes. 1)

		Country		Scopes clearly covered in findings					
		Sweden	The Netherlands	External	Policy: Contents	Policy: Processes	Product and management chains: Between chains	Product and management chains: Along chains	Product and management chains: Within nodes of
(j)	Potential accountability problems of focusing on different packaging materials separately and on packaging separated from other environmentally impacting activities	X	X	X			X	X	X
(f)	Aluminium not regulated separately despite being highly environmentally impacting, and despite being at least in Sweden considerably non-recycled	X	X		X			X	X
(i)	Country comparisons difficult since statistics calcu-lation methods differ between countries	X	X		X			X	X
(l)	Sorting in waste streams in relation to impurities	X			X			X	X
(h)	General packaging: Downgrading not accounted for	X	X		X	X		X	X
(a)	General packaging: *Free riders* might have caused statistics to look better than reality	X			X	X		X	X
(c)	General packaging: Not well defined and cross checked statistics	X			X	X		X	X
(m)	General packaging: Private-public conflicts effects, for stalling data quality improvements in Sweden, and for, for example, upsetting private actors in the Netherlands	X			X	X		X	X
(g)	Lack of statistics resolution on paths from waste generation to recycling	X	X			X		X	X
(b)	Statistics difficult to follow up due to change in calculation of misplaced aluminium cans	X				X		X	X
(k)	Whether complementary extraction from ashes is encouraged and feasible	X				X		X	X
(d)	Calculation of bottom ash extraction difficult since other metal sources are present		X			X		X	X
(e)	Assumptions based statistics calculations not scru-tinised by an otherwise rigorous national agency		X			X		X	X

1) Letters within parentheses refer to descriptions in the body text of sub-chapters 4.1 and 4.2.

Table 11. Comparison of scopes of socio-material environmental findings – metal packaging compared to the other socio-material environmental studies presented in chapter 2.

Scopes clearly covered in finding						Socio-material environmental findings	
External	Policy		Product and management chains			On metal packaging in this study	From other studies
	Contents	Processes	Between chains	Along chains	Within nodes of		
						n	*n*
X			X	X	X	1	1
X				X	X		3
	X			X	X	3	
	X	X		X	X	4	
		X		X	X	5	1
			X	X	X		1
				X	X		2
					X		1

Table 12. Comparison of characteristics of socio-material environmental findings – metal packaging compared to the other socio-material environmental studies presented in chapter 2. [1), 2)]

				Findings that each characteristic applies to			
				Metal packaging findings in this study		Difference compared to the other studies	
				Clearly	Partly	Clearly	Partly
				%	*%*	*percent-age points*	*percent-age points*
Entities		(1)	Socio-materiality	100		+11	-11
		(2)	Both social and material agency	100	8	+22	+8
		(3)	All levels from micro to macro	0	85		+7
Interac-tions	Quality	(4)	Imperfectness of interactions	77		-1	
		(5)	Open ended and close study useful	15	23	+4	-21
	Quantity	(6)	Boundary choices (incl. time)	77	15	-1	+15
		(9)	Number of interact-ion steps in a chain	54	8	-2	+8
		(10)	Frequency of inter-actions	54		-24	
		(11)	Many drivers	54	38	+43	+38
	Drivers	(7)	Mutually excluding practices	46	8	+2	-3
		(8)	Non-environmental drivers	77	15	-1	+15

1) Figures within parentheses refer to characteristics described in sub-chapters 2.1 (characteristics 1-7) and 2.2 (characteristics 8-11).
2) More detailed data used for calculating the percentage values are presented in Appendix A, in Table A.1 and in Table A.2.

4.1. Potential data and public policy ineffectiveness

In this sub-chapter, the ten, earlier in this chapter introduced, issues on potential data and public policy ineffectiveness are described.

Data lack of reliability

A few issues of potentially lacking reliability of data have been found for Sweden and the Netherlands. In Sweden, the statistics produced on metal packaging waste generation and recycling amounts are considered to be unreliable both for making informed and environmentally effective decisions. However, the reliability of packaging statistics is considered to be a problem for other countries as well. (Görling per. comm. 2013a, p. 3, Jonsson per. comm. 2013, p. 3) In Sweden, it is possible for so called free riders to enter the system (Jonsson per. comm. 2013, p. 3). Here, free riders refers to fillers, packaging producers, and packaging importers that supply packaging without it being accounted for as waste generated amounts. At the same the recycling amounts reported for Sweden consider all recycled metal packaging. The effect of these free riders is therefore that the statistics on both recycling rates and amounts of waste generated and recycled becomes incorrect in proportion to the potential amounts supplied by free riders. Further, the Swedish packaging system might therefore be seen as being less environmentally impacting than in reality. The municipal authorities are officially responsible for enforcing that free riders do not use the recycling system without officially taking part of it, including paying the packaging material fees that may result in them needing to charge higher prices for their products. However, in practice it is not considered to be feasible to require these authorities to carry out this task. Also, the latest survey on the size of free riding is considerably outdated, being from around the period 2002–2004. (Jonsson per. comm. 2013, p. 3). In the latest estimation of free riders included in the official reporting on the metal packaging system, regarding 2005, the free riders contributed to 7.0 kton of waste generation, or 9.6%, of the for that year total estimated 73 kton of metal packaging waste generated in Sweden (Naturvårdsverket 2012a, p. 65, 69). Free riders is less of a potential problem in the Netherlands from the point of view of some aspects of their calculations of recycling amounts and levels. This regards that their reported amounts of metal packaging waste generated first are subtracted with the amounts sorted at the source and that the recycling rates calculated for recovery from ashes is then applied to the remainder. (a)

Further, in Sweden, substantial amounts of aluminium beverage packaging are incorrectly disposed of in the collection systems for other metal packaging waste (Jonsson per. comm. 2013, p. 3). This is compensated for in the statistics starting with the reporting year 2010, but this will make the statistics less comparable to earlier years, and also necessarily needs to be calculated based on samples or estimated and thereby creates less reliability of the statistics. In relation to this, the figures for the different waste streams of aluminium packaging reported for 2011 do not align with the totals reported, as described in sub-chapter 3.1. (b)

Finally, for Sweden, the current statistics for recycling are not clearly defined or thoroughly cross-checked. The methods for producing the figures that the statistics are based on are not mentioned in the questionnaires used or provided through the replies given through them. The only cross-check performed considers that the data does not differ unexpectedly from the data of the preceding year. These comparisons are performed by the data gathering agency Statistics Sweden (Statistiska centralbyrån, SCB). (Jonsson per. comm. 2013, p. 3). (c)

For the Netherlands, two other aspects on lacking data reliability seem to be the most relevant ones clearly pointed out by this study. First, it is difficult to determine the recycled amounts of metal packaging waste extracted from bottom ashes after incineration of residual waste since it cannot easily be distinguished from other metal waste (d) (Verweij per. comm. 2013, p. 3). Second, one of the tasks of the Dutch agency Inspectie Leefomgeving en Transport (ILT) is to scrutinise and investigate the background on figures reported on Dutch packaging waste generation and recycling that the Dutch packaging co-ordinating organisation Nedvang provides. However, this has not been performed for these figures on metal packaging. (Verweij per. comm. 2013) These figures might need further study since they are based on several estimates and on what seems to be not thoroughly described calculations. These include that it is stated that the seemingly very high share of 100% of the metal packaging waste sorted at the waste source is recycled. For a point of reference, it can be compared to the by the Packaging and Newspaper Collection Service in Sweden reported around 93% of recycling of metal packaging sorted at the source (FTI n.d. c, p. 8). Further figures in potential need of scrutinising include are the calculations based on and using the estimate that 80% of the source metal packaging waste from organisations is collected at the waste source based on previous experience. The procedure for applying these 80% also is different in the reporting for 2011 than the method for applying them for the 2010 data leading to a seemingly not negligibly higher recycling rate for this fraction for 2011 than for 2010. (Nedvang 2011, p. 107, 2012, p. 78). (e)

Lack of resolution in public policy and in data

In both Sweden and the Netherlands, and following the EU directive, the recycling goal for metal packaging, besides aluminium beverage packaging in Sweden, is being focussed on the steel and aluminium packaging taken together (Directive 94/62/EC 1994, SFS 2006:1273, VROM 2005). The Swedish estimated recycling rates for this aluminium fraction was for 2010 reported to be 23%, compared to 83% for the steel packaging (Eurostat 2014c, Naturvårdsverket 2012a, p. 60), while no separate recycling rates for steel and aluminium packaging waste was reported for the Netherlands for 2011, as described in sub-chapter 3.2 (Nedvang 2012). Aluminium is considered to be most important of the two fractions to collect at a good rate, according to Thord Görling at the Packaging and Newspaper Collection Service, in Sweden (Görling per. comm. 2013a, p. 6). These two fractions were earlier separately regulated in Sweden, but following the EU directive this is no longer the case. If this issue is decided to be targeted, it may therefore need to go via the EU. From the environmental life cycle assessment proxies calculated in sub-chapter 3.3, increased aluminium packaging recycling seem definitely motivated from energy use and global warming potential perspectives, and can regarding raw material scarcity be both more or less of an issue compared to steel packaging depending on how the currently not economically viable aluminium resources are targeted. In addition, as described in sub-chapter 3.1, the major share of non-recycled aluminium packaging in Sweden seem to be other packaging than beverage packaging while their total amounts of non-beverage aluminium waste is considerably smaller than the corresponding amount for the beverage aluminium packaging fraction. (f) For Sweden, in addition, the policies and data does not differentiate between more than two of the eight different general paths that in the previous chapter were identified to exist between metal packaging waste generation and recycling, compared to a coverage in the Netherlands of the reported paths (g) (cf. Naturvårdsverket 2012a, Nedvang 2012, pp. 77–79). Finally, downgrading of material quality is generally not included for packaging recycling. Downgrading occurs for metal packaging through for example the introduction of and increased shares of tin (from tin cans) or copper (from electric and electronic appliances) into the metal recycling, and where copper is seen as a more problematic substance than tin due to several reasons. It is not being needed for alloys for new metal products (contrary to tin). Also, it forms alloys with both steel and aluminium. Downgrading can also be the result of corrosion caused if the generated metal packaging waste is not recycled quickly enough after the waste generation. (Görling per. comm. 2013a, cf. Nedvang 2012) (h)

Lack of comparability internationally

The possibilities of learning from best practices and comparing country statistics is limited due to variations in ways of determining packaging waste generation and recycling amounts (cf. Görling per. comm. 2013a, Verweij per. comm. 2013). However, even if regulations would become more clear on statistics production and also are given an increased level of resolution, the difficult task of enforcing such a structuring and harmonisation remains (cf. Naturvårdsverket 2009, Verweij per. comm. 2013) (i).

Lack of accountability

Taking the life cycle assessment perspective literally, the function of a system is at the core for comparisons to be useful, and this is not entirely straightforward for metal packaging, as well as for other packaging, over time and between countries. Thus the accountability, meaning the relevance of what the statistics on packaging cover, might be lacking. Over time, it is believed that both Swedish and Dutch packaging moves from metal to other materials, such as cardboard for food containers (Görling per. comm. 2013a) and plastics for paint buckets (Verweij per. comm. 2013). In addition, the scope of study, which would impact the choice of functional unit, has been suggested to be expanded to also account for the total environmental impact of the product that the packaging contains, since different packaging may influence for example the environmental impact from whether cooling of a food product during storage is needed or not (see SKB 2007, p. 13). In another functional unit extension, the metal packaging use might depend on the share of meals eaten at restaurants with potentially less use of metal packaging per amount of ingredients but with potential other environmental implications as well share, or on a country's share of providing global transporting services. (j)

4.2. Conflicting arguments and perceptions

In this sub-chapter, issues on conflicting arguments and perceptions are described.

About metals incineration

The obvious difference between Swedish and Dutch metal packaging recycling regarding the use or not of basing a main strategy or not on collecting metal packaging remains from residual waste incineration ashes seems to be a question not entirely settled in Sweden. One actor who has been involved in this discussion is Thord Görling. He currently works at the Packaging and Newspaper Collection Service (Förpackning- och tidningsinsamlingen AB, FTI) in Sweden, and was previously the CEO for the now within the Packaging and Newspaper Collection Service incorporated materials recycling organisation Metallkretsen. He has earlier suggested to complement the sorting at the source of metal packaging waste with recovery from the incinerated residual waste. When interviewed for this study he described that this suggestion at that time been declined by the Swedish Environmental Protection Agency (Naturvårdsverket) since it would violate the key principle to only incinerate combustible materials. (Görling per. comm. 2013a) Catarina Östlund, at the Swedish Environmental Protection Agency – who since a little more than one year, as of December 2013, provides guidance on the public policies on packaging and packaging recycling and who is also to a considerable degree recruited as a consultant expert for waste studies – did on the other hand see a possible potential in such a complement, when the issue was brought up as part of this study. The risk of demotivating consumers was at the same time discussed as a potential threat to the feasibility of such an approach. (Östlund per. comm. 2013) Also, it would result in introducing an additional waste stream to the already many not separately reported on streams for Swedish metal packaging waste collection. And the as low as 20% stated sorting rate, previously in this report mentioned, for such a combined system in the Netherlands may be worth studying closer if this option is considered, even if the Dutch system may not be comparable to the Swedish one since the incineration path in the Netherlands is the promoted option for this waste stream. Also, the effect of such an initiative depends on whether in practice currently extraction of metal packaging residues is performed from incineration ashes as described not to be fully settled in sub-chapter 3.2. If such extraction does occur, this also implies that current recycling amounts and rates would might need to be adjusted accordingly. (k)

On sorting in waste streams

Sorting in material specific waste streams has been a suggested and in several reports investigated approach to replace the division between packaging and other household waste containing the same type of main material in Sweden (Naturvårdsverket 2007, 2009, SOU 2012:56). It has been promoted in order for sorting at the waste source to be more logical for citizens (Naturvårdsverket 2007, p. 5). However, the presence of tin among the packaging (Görling per. comm. 2013a, SOU 2012:56) and copper among the other metal waste (Görling per. comm. 2013b, p. 6) is seen as problematic since the other waste fraction thereby could be contaminated. Copper is, as described in sub-chapter 4.1, a particularly not useful pollutant, and it is in this waste stream context mainly currently part of the, per weight, smaller of the two fractions that have been discussed to be merged to the metal material stream. (Görling per. comm. 2013a, b) However, tin was presented as the reason for not suggesting these metals waste fractions to be merged in a recent, large study on the Swedish waste management system (SOU 2012:56), and at the same time tin might be on its way to be phased out from food metal cans and is considered to possibly last for 50 years at current consumption and recycling rates (SKB 2007). Large efforts were thus directed to non-successful initiatives which might have been possible to avoid by carefully considering relatively basic properties of the waste streams considered. (l)

Public-private conflicts

Conflicts seem to be present between public and private actors in relation to the packaging recycling systems in both Sweden and the Netherlands. In Sweden, the whole waste management sector has been seen as a conflict area due to it being divided between municipal and private actors (Görling per. comm. 2013a). Residual waste management and waste incineration are municipal, but also partly the packaging collection through the ownership of the land where the intermediate scale facilities for household packaging collection are located. Private companies, on the other hand, manage the transports, storage, and recycling of the packaging waste collected at the waste source. Large disagreements are visible related to a large, recent investigation where the collection part of the sorting at the waste source system was suggested to be managed by municipalities instead of the current private actors (Görling per. comm. 2013a). One problematic starting point might be, but may not need to remain, a feeling that packaging producers were already in the 1980s pointed out as the actors mainly responsible for the environmental and other problems related to packaging (cf. Görling per. comm. 2013a). Less drastic changes, at least organisationally, seem to follow by the Dutch approach, where goals are set but the internal organising of activities are less centrally determined. There, for example, and as already mentioned, each municipality decides on whether collection should be performed at the source of packaging waste or after incineration of residual waste or by other means (Verweij per. comm. 2013).

The effects of this tension can be seen in the practices of packaging policy in Sweden. Due the uncertainty about the future, since large changes might occur or might not occur, the development of the questionnaires for collecting information about packaging recycling amounts has been stalled. This questionnaire development is intended to come closer to measuring actual recycling than the current practices for which it is not even clear which processes between collection and recycling that the reported amount represent (described further in finding c, in the previous sub-chapter). (Jonsson per. comm. 2013) As a point of reference, the rate between collection and final recycling of the Packaging and Newspaper Collection Service in Sweden was, as earlier stated, presented to be approximately 93% in their 2012 annual report (FTI n.d. c).

However, the Netherlands are not free from drastic and unpopular measures related to the packaging system. In the Netherlands, conflicts are or have been visible around packaging material fees. These fees were changed in order to help covering national budget deficits, at levels several times higher than the costs within the packaging system required to be covered (Ter Morsche per. comm. 2013, Verweij per. comm. 2013). This upset the packaging producers (Ter Morsche per. comm. 2013), while at the same time it was seen as useful by the enforcement agency Inspectie Leefomgeving en

Transport since it made packaging producers aware of and motivated to decrease packaging amounts (Verweij per. comm. 2013). (m)

4.3. Analysis

The results are in the following analysed further from a few perspectives.

First, the findings have been related to the product chain processes and the sizes of material flows and environmental impacts in order to look for overarching patterns. The outcome of this approach is outlined in Figure 8, further on in this sub-chapter. The findings were divided into two groups. The first group consists of six findings that were mainly related to organisational differences between the two countries. The other seven findings were found mainly to be related to the complexity of the metal packaging systems.

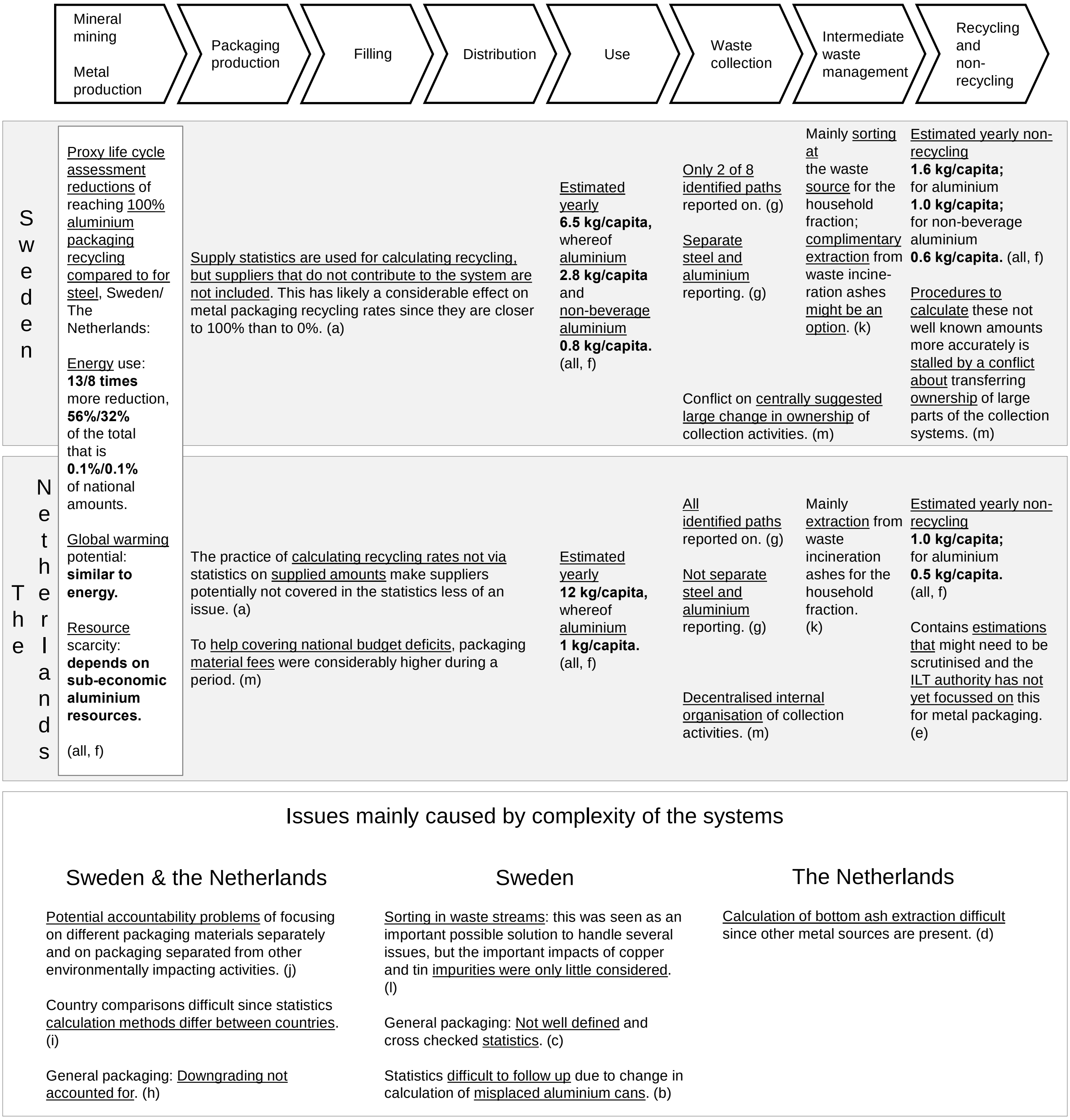

Figure 8. The complex landscape and characteristics of the identified socio-material environmentally significant issues of the metal packaging systems in the two countries. Letters in parentheses refer to the labelling used for the findings in sub-chapters 4.1 and 4.2. Figure 3 is used as the basis for this figure.

Through this overview, the findings in this study seem to form a rich and complex landscape. Based on this observation, a few points are in the following reasoned about. First, the combination of considerable potential opportunities for lowering environmental impacts and the difficulty to assess the effects of such measures are discussed. Second, the issue of how the complexity adds to the risk of easily missing vital aspects when managing the systems are addressed. Third, and finally, the potential improvement areas of the systems, besides being aware of the complexity, are discussed.

Regarding systems opportunities and their effects, Figure 8 was designed to create only a basic overview of the systems. Nevertheless, a multitude of environmentally significant practices were found to be necessary to include in order for the figure to be representative. This also lies in line with the considerable number of significant findings, (a)-(m), described in subchapters 4.1 and 4.2. Further, this seems to be the case although the management of these metal packaging systems through the interviews in this study seemed to be seen as successful compared to the management of other waste streams. The system was seen as partly driven by itself due to the comparable intrinsic worth of scrap steel and scrap aluminium. (Cf. Görling per. comm. 2013a, b, Jonsson per. comm. 2013, Östlund per. comm. 2013, Ter Morsche per. comm. 2013, and Verweij per. comm. 2013) Actually, these views on the management of the systems as successfully governed might be part of the reason for the limited focus on them this far. Regarding the potential effects of implementing the opportunities identified in this study, these effects seem difficult to assess both due to aspects of each of them and due to their complex interdependences.

Second, it seems easy to overlook factors of considerable importance. This does likely apply not only to the metal packaging systems, but to other waste management systems as well. The complex landscape of environmentally significant practises seems to have resulted from including both material and management aspects in this study. The practises seem in many cases to be interrelated, such as monitoring data quality, relations between public and private actors, and the aspect of downgrading of material quality through recycling.

Third, Figure 8 assists in pointing out five seemingly particularly relevant potential opportunities to address for lowering the systems' environmental impacts. These are in Sweden (re-)introducing complementary extraction of metal packaging waste from incineration ashes, an increased focus on aluminium packaging, and on downgrading of material quality, improving relations between public and private actors, as well as scrutinising of the production of official statistics.

It should be noted, however, that these five potential opportunities, as well as the results of this study in general, cover concrete identified potential changes to current practices. Thus, other practices not identified in this study might be of importance in addition. These other practices include the actual possibility to use more metal packaging waste in the metal re-melting processes (cf. EAA 2007), potential environmental and other effects of conducting the re-melting in remote locations (cf. Görling per. comm. 2013a, b), the accounting of non-recycled metal packaging litter (cf. SKB 2007, p. 6), and work on minimising packaging material per unit of packaging. (cf. Verweij per. comm. 2013)

Fourth, and finally, regarding more structural aspects of this study, it has been compared to the scopes and characteristics of earlier socio-material environmental studies. This is also presented in Table 11 and Table 12, in the introduction of this chapter. Regarding scopes, this study has a larger coverage of issues related to policy design and policy processes. This should not be surprising since this is a topic where policy is and has been used to a comparably large extent. Nevertheless, the issues of this study, like earlier studies, are in all cases related to socio-material relations both along product and management chains as well as within the nodes of these chains. Interactions between different product and management chains and with external factors and between product and management chains are also present in one of the findings of this study.

Regarding socio-material characteristics that are prominent in the findings of this study, a few remarks can also be made. This study differs only clearly in one of these characteristics from earlier socio-material environmental studies. A multitude of drivers is in this study found to be a more common feature. In common with the previous studies, frequent characteristics include: that both social and material agency is seen, imperfectness of interactions, the importance of where the boundary of activities considered is set, as well as non-environmental drivers.

5. CONCLUSIONS AND RECOMMENDATIONS

A few conclusions and recommendations for lowering environmental impacts from metal packaging systems are made based on the analysis of the results in this study, and they are summarised in Table 13, further on in this chapter. Conclusions are presented for both countries. The conclusions might be of relevance both specifically for the metal packaging systems, but also for waste management more generally. The conclusions are targeted for all actors in the respective systems, including producers, consumers, waste management actors, and public authorities. For an overview, see also Figure 8 on page 52.

Table 13. Summary of conclusions from this study.

<table>
<tr><th>Sweden</th><th>The Netherlands</th><th>Details in chapter 4</th></tr>
<tr><td>Use complementary extraction from ashes:
This can be a viable option besides the current sorting of household metal packaging waste at the waste source. However, the feasibility seems to depend on, for example, the effect this will have on the consumers' waste handling behaviour.</td><td></td><td>(k), p. 48–9</td></tr>
<tr><td colspan="2">Increase the focus on aluminium:
Aluminium packaging seems to be recycled at low rates and considerable environmental improvements can be reached if its recycling is increased.</td><td>(f), p. 46–7;
(b), p. 45–6</td></tr>
<tr><td colspan="2">Scrutinise official statistics:</td><td rowspan="2">(a)-(j),
p. 45–8</td></tr>
<tr><td>For example, recycling rates are calculated on consumption data that exclude a likely considerable amounts from not registered fillers and importers.</td><td>For example, assumptions of 100% recycling is used for one of the largest metal packaging waste streams.</td></tr>
<tr><td colspan="2">Better avoidance of downgrading, in for example re-melting:
Downgrading of material quality is related both to re-melting and to other waste handling processes. Downgrading is currently not monitored.</td><td>(h), p. 47;
(l), p. 49</td></tr>
<tr><td colspan="2">Improving relations between public and private actors could help:</td><td rowspan="2">(m),
p. 50–1</td></tr>
<tr><td>The presence of public-private conflicts have stalled improvements in monitoring data quality.</td><td>Material packaging fees were used for helping to cover national budget deficits, and the effects were, for example, that private actors were highly upset.</td></tr>
<tr><td colspan="2">Opportunities for considerable improvements exist, particularly in Sweden, but are difficult to assess:
Opportunities exist although the management of these metal packaging systems is viewed as successful compared to the management of other waste streams. Actually, this view might be part of the reason for the limited focus on them this far. The effects of changes are difficult to assess both due to aspects of each of them and due to their complex interdependences.</td><td></td></tr>
<tr><td colspan="2">It is easy to overlook factors of considerable importance:
The combined material and management aspects of these systems seem to form a complex landscapes with many interrelations, such as between monitoring data quality, relations between public and private actors, and the aspect of downgrading of material quality through recycling.</td><td>(f), p. 46–7;
(l), p. 49</td></tr>
<tr><td colspan="2">Other practices not identified in this study can be of importance in addition:
These practices include the actual possibility to use more metal packaging waste in the metal re-melting processes, potential environmental and other effects of conducting the re-melting in remote locations, the accounting of non-recycled metal packaging litter, and work on minimising packaging material per unit of packaging.</td><td></td></tr>
</table>

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APPENDIX A – DETAILED RELATIONS BETWEEN SOCIO-MATERIAL CHARACTERISTICS AND ENVIRONMENTAL FINDINGS

Table A.1. Socio-material characteristics that are indicated to influence environmental performance in findings from previous studies. [1), 2), 3), 4)]

		Socio-material characteristics indicated to influence environmental performance										
		Entities			Interactions							
					Quality		Quantity				Drivers	
		(1)	(2)	(3)	(4)	(5)	(6)	(9)	(10)	(11)	(7)	(8)
		Socio-materiality	Both social and material agency	All levels from micro to macro	Imperfectness of interactions	Open-ended and close study useful	Boundary choices (incl. time)	Number of interaction steps in a chain	Frequency of interactions	Many drivers	Mutually excluding practices	Non-environmental drivers
(D)	The environment setting the frame	X	X	(X)		(X)	X				X	
(A)	Caring or emergency-driven acting	X	X	(X)	X	X			X			X
(C)	Need for bottom-up indicators	(X)		(X)	X	(X)	X	X	X		(X)	X
(I)	Company pressured by currency instability	X	X	(X)	X		X	X	X	X	X	X
(G)	Contact with authority repeatedly	X	X	(X)		(X)			X			
(E)	Organisationally long action chain	X			X	(X)	X	X	X			X
(B)	Other impacts from new type of building	X	X	(X)	X		X					X
(H)	Organisational stability during growth	X	X	(X)	X		X	X	X		X	X
(F)	Emergency situation leading to change	X	X		X		X	X	X		X	X

[1)] Letters within parentheses refer to descriptions in sub-chapter 2.2.
[2)] X = characteristic applying clearly to the respective finding.
[3)] (X) = characteristic applying partly to the respective finding.
[4)] Figures within parentheses refer to characteristics described in sub-chapter 2.1 (characteristics 1-7) and in sub-chapter 2.2 (characteristics 8-11).

Table A.2. Environmentally related findings on the metal packaging systems, and characteristics of these findings. 1), 2), 3), 4), 5)

		Socio-material characteristics indicated to influence environmental performance										
		Entities			Interactions							
					Quality		Quantity				Drivers	
		(1)	(2)	(3)	(4)	(5)	(6)	(9)	(10)	(11)	(7)	(8)
		Socio-materiality	Both social and material agency	All levels from micro to macro	Imperfectness of interactions	Open ended and close study useful	Boundary choices (incl. time)	Number of interaction steps in a chain	Frequency of interactions	Many drivers	Mutually excluding practices	Non-environmental drivers
(j)	Potential accountability problems of focusing on different packaging materials separately and on packaging separated from other environmentally impacting activities	X	X	(X)			X	X		X	(X)	(X)
(f)	Aluminium not regulated separately despite being highly environmentally impacting, and despite being at least in Sweden considerably non-recycled	X	X		X		(X)					X
(i)	Country comparisons difficult since statistics calculation methods differ between countries	X	X	(X)	X		(X)	X	X	X	X	X
(l)	Sorting in waste streams in relation to impurities	X	X	(X)		X	X	X	X	(X)	X	X
(h)	General packaging: Downgrading not accounted for	X	X	(X)			X	X		X		X
(a)	General packaging: Free riders might have caused statistics to look better than reality	X	X		X	(X)	X	X	X	X		X
(c)	General packaging: Not well defined and cross checked statistics	X	X	(X)	X		X	X		(X)		X
(m)	General packaging: Private-public conflicts effects, for stalling data quality improvements in Sweden, and for, for example, upsetting private actors in the Netherlands	X	X	(X)	X	X	X	X	X	X	X	X
(g)	Lack of statistics resolution on paths from waste generation to recycling	X	X	(X)	X		X	(X)		X	X	X
(b)	Statistics difficult to follow up due to change in calculation of misplaced aluminium cans	X	X	(X)	X	(X)	X		X	(X)		X
(k)	Whether complementary extraction from ashes is encouraged and feasible	X	X	(X)	X	X	X		X	X	X	(X)
(d)	Calculation of bottom ash extraction difficult since other metal sources are present	X	X	(X)	X		X			(X)	X	X
(e)	Assumptions based statistics calculations not scrutinised by an otherwise rigorous national agency	X	X	(X)	X	(X)			X	(X)		

1) Letters within parentheses refer to descriptions in sub-chapters 4.1 and 4.2.

[2] X = characteristic applying clearly to the respective finding.
[3] (X) = characteristic applying partly to the respective finding.
[4] Figures within parentheses refer to characteristics described in the body text of the previous section of the report (1-7) and of this section (8-11).
[5] Lightly shadowed columns indicate characteristics found in addition to the ones derived from the four socio-material approaches referred to in the previous section of the report.

APPENDIX B – DETAILS ON METAL PACKAGING AMOUNTS

Table A.3. Estimated metal packaging amounts per year. Totals represent sums of recycled and non-recycled amounts. 1), 2), 3)

	Total generated		Non-recycled		Recycled	
	Sweden	The Nether-lands	Sweden	The Nether-lands	Sweden	The Nether-lands
	kton	*kton*	*kton*	*kton*	*kton*	*kton*
1997	70	216	38	71	32	145
1998	74	236	17	48	58	188
1999	69	217	34	48	35	169
2000	67	220	38	48	29	172
2001	68	211	21	47	47	164
2002	69	222	22	45	47	177
2003	66	219	20	33	46	186
2004	70	213	25	30	46	183
2005	73	211	27	34	47	177
2006	68	187	20	35	48	152
2007	69	180	18	30	51	150
2008	65	182	19	25	47	157
2009	63	172	14	23	50	149
2010	60	178	15	21	46	157
2011	61	193	15	17	46	176

1) Data from Eurostat (2014c), Naturvårdsverket (2012a, p. 60), and Nedvang (2012, p. 20, 77–79). The data presented in the table has in many cases been rounded off, in order to provide a clearer overview. However, the original data is used for the corresponding graphs in Figure 4.
2) Packaging statistics for 2012 and onwards not yet published, as of 27 February 2014.
3) See remarks about inaccuracy issues of the data in the footnotes of Figure 4.

Table A.4. Estimated metal packaging amounts per year and capita. Totals represent sums of recycled and non-recycled amounts. 1), 2), 3)

	Population		Total generated		Non-recycled		Recycled	
	Sweden	The Nether-lands	Sweden	The Nether-lands	Sweden	The Nether-lands	Sweden	The Nether-lands
	millions	*millions*	*kg*	*kg*	*kg*	*kg*	*kg*	*kg*
1997	8.8	15.6	7,9	13,9	4.3	4,6	3.6	9.3
1998	8.8	15.7	8,4	15,1	2.0	3,1	6.5	12.0
1999	8.9	15.8	7,8	13,8	3.9	3,0	3.9	10.7
2000	8.9	15.9	7,6	13,9	4.3	3,0	3.3	10.8
2001	8.9	16.0	7,7	13,2	2.4	2,9	5.3	10.3
2002	8.9	16.1	7,8	13,8	2.5	2,8	5.3	11.0
2003	8.9	16.2	7,4	13,5	2.2	2,0	5.2	11.5
2004	9.0	16.3	7,9	13,1	2.7	2,0	5.2	11.3
2005	9.0	16.3	8,1	12,9	3.0	2,1	5.2	10.9
2006	9.0	16.3	7,5	11,4	2.2	2,1	5.3	9.3
2007	9.1	16.4	7,6	11,0	2.0	2,0	5.6	9.2
2008	9.1	16.4	7,2	11,1	2.1	1,5	5.1	9.6
2009	9.3	16.5	6,9	10,4	1.5	1,4	5.4	9.0
2010	9.3	16.6	6,5	10,7	1.6	1,3	4.9	9.5
2011	9.4	16.7	6.5	11,6	1.6	1,0	4.9	10.6

1) Data from Eurostat (2014c, e), Naturvårdsverket (2012a, p. 60), and Nedvang (2012, p. 20, 77–79). The data presented in the table has been rounded off, in order to provide a clearer overview. However, no rounding off is performed for the data used for the corresponding graphs in Figure 5.
2) Packaging statistics for 2012 and onwards not yet published, as of 27 February 2014.
3) See remarks about inaccuracy issues of the data in the footnotes of Figure 4.

Table A.5. Estimated aluminium packaging excluding beverage packaging, in Sweden per year. Totals represent sums of recycled and non-recycled amounts. [1), 2)]

	Population	Total generated		Not recycled			Recycled		
		Total	Per capita	Total	Per capita		Total	Per capita	
	millions	*kton*	*kg*	*kton*	*kg*	*%*	*kton*	*kg*	*%*
2004	9.0	10.0	1.11	7.3	0.81	73	2.7	0.30	27
2005	9.0	10.8	1.20	7.9	0.88	73	2.9	0.32	27
2006	9.0	9.4	1.04	6.2	0.69	66	3.2	0.35	34
2007	9.1	9.7	1.06	6.2	0.68	64	3.5	0.38	36
2008	9.1	7.8	0.85	4.9	0.54	63	2.9	0.32	37
2009	9.3	7.8	0.84	4.5	0.49	58	3.2	0.35	42
2010	9.3	7.8	0.84	6.0	0.64	77	1.8	0.19	23

[1)] Data from Eurostat (2012e), and Naturvårdsverket (2012a, pp. 60, 65). Packaging statistics for 2012 and onwards not yet published, as of 27 February 2014. The data presented in the table has been rounded off, in order to provide a clearer overview. However, no rounding off is performed for the data used for the corresponding graphs in Figure 6 and Figure 7.
[2)] See remarks about inaccuracy issues of the data in the footnotes of Figure 6 and Figure 4.