waste management

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Aims and Scope: *Waste Management* is an international journal devoted to the presentation and discussion of information on the generation, prevention, characterization, monitoring, treatment, handling, reuse and ultimate residual disposition of solid wastes, both in industrialized and in economically developing countries. The journal addresses various types of solid wastes including municipal (e.g., residential, institutional, commercial, light industrial), agricultural, and special (e.g., C&D, health care, household hazardous wastes, sewage sludge).

Waste Management is designed for scientists, engineers, and managers, regardless of their discipline, who are involved in scientific, technical and other issues related to solid waste management. Emphasis is placed on integrated approaches. These approaches require the blending of technical and non-technical factors. Although the dissemination and application of innovative technical information is extremely important, the implementation of sustainable waste management practices also requires a thorough understanding of the pertinent legal, social, economic, and regulatory issues involved.

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Editorial

WEEE: Booming for sustainable recycling

Recently, production of metals from secondary resources has gained vital importance due to the depletion of primary resources along with concurrent increase in production/consumption of waste materials which contain base/precious metals and rare earth elements (REE). Waste Electrical and Electronic Equipments (WEEE) or E-waste is of great interest due to its base (mainly Cu), ferrous (Co, Ni), precious (Au, Ag, Pd) metals and rare earth elements (REE). Many of these metals including REEs are currently classified as “critical” (e.g. indium, REE) with a high supply risk (EU, 2014). Notwithstanding this, there has been a growing environmental concern on disposal of WEEE in landfills due to its hazardous organic/inorganic material content. Amount of WEEE is rapidly growing due to the production of electrical and electronic equipments (EEE) as well as reduction in lifespan of EEE.

WEEE comprises high portion of municipal solid waste, which is around 5% (STEP, 2010). Historically, the WEEE in EU was observed to increase by 16–28% in every five years, which is three times faster than the average annual municipal solid waste generation. Specifically, 9 mt WEEE was generated in 2005 and it is expected to grow to more than 12 mt by 2020 which makes WEEE as one of the fastest growing waste stream in Europe (EU, 2016). Considering worldwide, the amount of WEEE increased from 33.8 to 41.8 Mt during the period 2014–2016 and it is expected to increase to 49.8 Mt in 2018 (Table 1) (UNU-IAS, 2014). Generation of WEEE in kg per inhabitant is also expected to rise from 5.0 (2015) to 6.7 (2018) (Table 1). Data of generation of WEEE (kg per inhabitant) in continents implied that Europe, Oceania and America are the dominant continents that generate WEEE (Table 1, Fig. 1).

Concurrent rise in amount of WEEE along with public/academic awareness has led to rapid increase in the number of articles/reports dealing with treatment of WEEE. Fig. 2 shows the article search results in the Scopus (Elsevier) during the period 2001–2016 with keywords comprising of WEEE, E-waste, E-scrap and Electronic waste. The number of search results have increased from 103 (2001) to 781 (2015). Totally number of the published papers is 87.967 with keyword as waste management and approx. 10% (approx. 8.000 papers) of these papers is relating to WEEE during the period 2001–2016.

Considering the economic/environmental facts, regulations have been issued in EU and worldwide for the management of WEEE through recycling/recovery of metals from this waste stream (EU, 2012). These regulations have forced producers as well as municipalities to properly manage WEEE by implementing technically and economically feasible processes for recycling/recovery activities. Various treatment options based on conventional physical, pyrometallurgical and bio/hydrometallurgical processes have been proposed for the recovery of metals from WEEE. There has been a successful worldwide industrial application based on pyrometallurgical methods. More recently, bio/hydrometallurgical processes have received attention due to their relatively low-cost, ecofriendly nature and suitability for both large scale as well as small scale applications. A mobile plant was also commissioned using hydrometallurgical processes which was built within an EU project called HydroWEEE (Tuncuk et al., 2012).

The content of this special issue also reflects the latest trends in WEEE management and treatment. This special issue includes 26 articles with over 200 pages. Depending on the toxicity of lead and mercury, some studies have focused on the removal/recycling of these metals from CRTs and fluorescent lamps. Several articles are based on the recovery of copper/gold from printed circuit

Table 1
Generation of WEEE worldwide (UNU-IAS, 2014).

Year	WEEE generated (Mt)	Population (billion)	WEEE generated (kg/inh.)
2010	33.8	6.8	5.0
2011	35.8	6.9	5.2
2012	37.8	6.9	5.4
2013	39.8	7.0	5.7
2014	41.8	7.1	5.9
2015[a]	43.8	7.2	6.1
2016[a]	45.7	7.3	6.3
2017[a]	47.8	7.4	6.5
2018[a]	49.8	7.4	6.7

[a] Data 2015 onwards are forecasts.

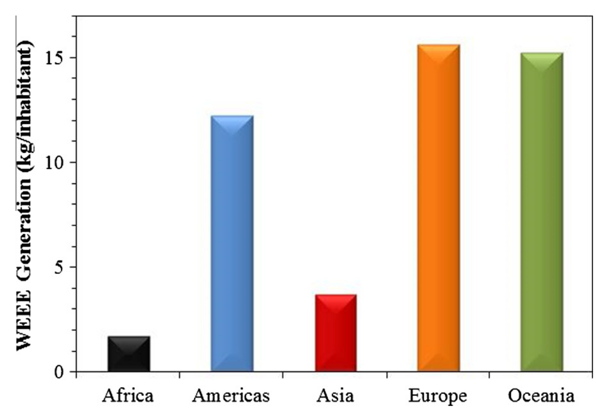

Fig. 1. WEEE generation (kg per inhabitant) in continents (UNU-IAS, 2014).

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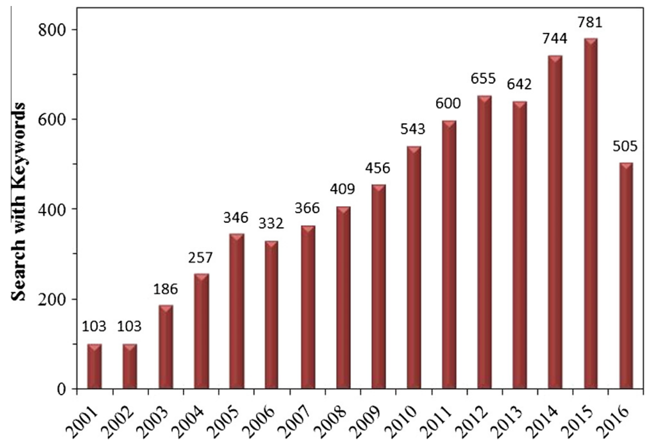

Fig. 2. Article search results in Waste Management between 2001 and 2016 using keywords of WEEE, E-waste, E-scrap and Electronic waste (2016 results is by 22 August 2016).

boards mainly using the bio/hydrometallurgical methods. There are also some interesting studies on the recovery of indium as a critical metal from liquid–crystal–display glass and on management strategies and the recycling of organic/plastic fraction from WEEE which also holds a lot of interest in the current scenario.

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Ata Akcil

Suleyman Demirel University, Isparta, Turkey

E-mail address: ataakcil@sdu.edu.tr

Ata Akcil, P.Eng., Ph.D, is full professor of base, precious and rare earth metal recovery techniques on hydrometallurgy and biohydrometallurgy, Engineering Faculty at the SDU, Isparta, Turkey. Group Leader, Mineral-Metal Recovery and Recycling (MMR&R) Research Group and Member of International Waste Management Group (IWWG Task Group on WEEE), Associate Editor of Waste Management. His research interests include: metal recovery techniques, critical raw materials, resource recovery, recycling technologies on primary and secondary resources including WEEE, Spent Batteries, Industrial Waste.

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Assessing the role of informal sector in WEEE management systems: A System Dynamics approach

Romadhani Ardi *, Rainer Leisten

Chair of Business Administration and Operations Management, Department of Mechanical and Process Engineering, Faculty of Engineering, University of Duisburg-Essen, Bismarckstr. 90, 47057 Duisburg, Germany

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ABSTRACT

Generally being ignored by academia and regulators, the informal sector plays important roles in Waste Electrical and Electronic Equipment (WEEE) management systems, especially in developing countries. This study aims: (1) to capture and model the variety of informal operations in WEEE management systems, (2) to capture the dynamics existing within the informal sector, and (3) to assess the role of the informal sector as the key player in the WEEE management systems, influencing both its future operations and its counterpart, the formal sector. By using System Dynamics as the methodology and India as the reference system, this study is able to explain the reasons behind, on the one hand, the superiority of the informal sector in WEEE management systems and, on the other hand, the failure of the formal systems. Additionally, this study reveals the important role of the second-hand market as the determinant of the rise and fall of the informal sector in the future.

1. Introduction

Waste Electrical and Electronic Equipment (WEEE) is an emerging global issue, especially in many developing economies. WEEE generation rate increases rapidly and continues to rise even higher in the foreseeable future. Several projections occur to support this idea. Yang et al. (2008) forecast the growth of Chinese obsolete personal computers (PC), TVs, refrigerators, washing machines, and air conditioners at the average level of 24.69%, 8.2%, 4.1%, 13.05%, and 40.01% per year, respectively. Dwivedy and Mittal (2010a,b) estimate the average growth of WEEE generation in India by 7% annually. In a study with a broader scope, Yu et al. (2010) present relatively large figures of 400–700 million units obsolete PCs in developing countries by 2030, as compared with 200–400 million units in developed ones. These growing numbers are influenced by several interrelated factors, i.e. (1) the growth of electronic industries, (2) the declining lifespans of electronic products, (3) the shift of customers' behavior, (4) the condition of the market that is far from being saturated, and (5) the consistency of technology innovations during last decades (Dwivedy and Mittal, 2010b; Jiménez-Parra et al., 2014; Wang et al., 2013a).

WEEE, by the nature of its components and treatments, implicates the three sustainability pillars: economic, environment, and social. Previous literature pays much attention in the economics of WEEE recovery activities such as in reuse, remanufacturing, recycling, and complete closed-loop supply chains (Bohr, 2007; Ferrer, 1997; Georgiadis and Besiou, 2009a; Geyer and Doctori Blass, 2009; Shinkuma and Managi, 2010; Toyasaki et al., 2011; Walther et al., 2009). This situation occurs, similarly, in the environmental aspect, in which various approaches have already taken places, e.g. Life Cycle Assessment, Material Flow Analysis, Multi-Criteria Analysis, System Dynamics (Georgiadis and Besiou, 2009b; Kiddee et al., 2013; Menikpura et al., 2014; Wäger et al., 2011). Remarkably, social issues of WEEE recovery operations have gained more authors' interest (Georgiadis and Besiou, 2009a; Manhart, 2007; Perez-Belis et al., 2014). Social aspects of sustainability become relevant to be addressed in the light of growing WEEE problems in developing countries, especially on informal recycling (Williams et al., 2013). Accordingly, Toxics Links (2003, 2004), Steiner (2004), Streicher-Porte et al. (2005), and Sinha-Khetriwal et al. (2005) provide several initiatives of this research field by capturing the operations of informal sectors in Indian cases.

In most developing countries, informal activities appear in many parts of the reverse supply chains, e.g. in collection, refurbishment, treatment, recycling, and secondary markets. Chi et al. (2011) and Manomaivibool (2009) comprehensively capture these

* Corresponding author.
E-mail addresses: romadhani.ardi@stud.uni-due.de (R. Ardi), rainer.leisten@uni-due.de (R. Leisten).

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existences in their Chinese and Indian cases, consecutively. These informal activities pose challenges to the whole WEEE management systems. Their improper treatment and recycling methods harm the environment (Sthiannopkao and Wong, 2013; UNEP, 2009), and their complex networks and process efficiency contribute to the failure of some formal initiatives (Chi et al., 2011; Raghupathy et al., 2011). Also, several authors list some difficulties in applying Extended Producer Responsibilities (EPR) approaches in developing regions (Kojima et al., 2009; Nnorom and Osibanjo, 2008).

Despite its importance, the informal sector is still majorly neglected by academia and legislators (Besiou et al., 2012; Chi et al., 2011). It is difficult to find the word "informal" in advanced directive and guidance (OECD, 2001; European Union, 2012, 2003). Therefore, there is an urgent need to understand the role and characteristics of the informal sector, so that the society can pursue the greater benefit.

Using System Dynamics (SD) approach, the objective of this study are threefold: (1) to capture and model the variety of informal operations in WEEE management systems, (2) to capture the dynamics existing within the informal sector, and (3) to assess the role of the informal sector as the key player in the WEEE management systems, influencing both the sector itself and its counterpart, the formal sector. By using India as the reference systems, this study explains the cause of the growth of the informal sector and the reason for the failure of the formal sector to compete with informal one. Furthermore, the scenario analysis in this paper reveals the key role of the second-hand market, determining the rise and fall of the informal sector.

The rest of this manuscript is organized as follows. Section 2 provides literature reviews regarding WEEE issues in developing countries, informal sector, and SD. Section 3 explains deeply about the SD methodology. Section 4 reports the model formulation and testing steps. Section 5 discusses the simulation processes and the results. Finally, Section 6 describes the important findings from this study.

2. Literature review

ILO defines informal waste worker as "individuals or small and micro-enterprises that intervene in waste management without being registered and without being formally charged with providing waste management services" (GIZ, 2011). Existing literature has characterized the impacts of the informal recycling sectors into society; many of them reveal harsh realities. Economically, the informal sector is described as the burden of the formal sector's profitability. Environmentally, informal operations are related to harmful treatments and processing methods, e.g. manual dismantling by bare hands, open acid leaching, burning components, chipping and melting plastics, sweeping toner, in unhealthy and unsafe working conditions (Wei and Liu, 2012). Socially, most of the informal workers are coming from marginalized social groups, with limited skills and low access to formal jobs (Wilson et al., 2006).

To be more specific, the informal sector is portrayed as the failure cause of several take-back projects and initiatives by formal sectors. For instance, Kojima et al. (2009) record the cease of a collection project conducted by Nanjing Jinze Metallic Material Co. Ltd. and Motorola in 2004 because of the shortage of input. In this environment, the formal actors are difficult to compete because: (1) they face lack of access to collect disposed products, as compared with effective door-to-door collection from scavengers, (2) they must comply with the environmental standards in treating the WEEE, while informal operations may operate with very efficient but unsafe methods; and consequently, (3) they pay higher recovery cost with limited recovered value. Chi et al. (2011) analyze the reason informal sectors dominate WEEE recycling sectors in China and point out that this situation might threaten the sustainability of the whole formal recycling systems.

It seems these conditions will be more problematic in the near future because of the steady growth of the informal sector in the developing countries (Pandey and Govind, 2014). This trend emerges in India, China, Pakistan, Nigeria, and possibly in more countries with similar situations (Chi et al., 2011; Wang et al., 2013b). The informal growth, exogenously, is affected by the unique trajectories of general solid waste management in developing countries that are situated by rapid urbanization, inequalities, and economic disparities (Marshall and Farahbakhsh, 2013). Furthermore, this growth could be observed from the increasing quantity of collected products, the vitality of reuse market, and the relatively huge number of the informal workers. Some factors endogenously influence this phenomenon, i.e. (1) adequate input of WEEE from illegal imports and households, (2) low level of treatments and recovery cost, (3) high rate of recovered value; (4) stable growing demand for recovered products, components, and materials; (5) absence of WEEE-specific regulations and law enforcement for a long period, and (6) limited capacity of initial formal systems (Chi et al., 2011; Manomaivibool, 2009; Wang et al., 2013b; Widmer et al., 2005).

Nevertheless, previous reviews have captured some potential benefits from the informal recycling sector. In their paper, Widmer et al. (2005) place a specific sub-section on informal WEEE management systems and assess the situations in India, China, and South Africa. These authors discuss the emergence of informal WEEE players and their benefit to increase job opportunities if the authorities provide them with adequate training and technology. In a more recent WEEE global review, Ongondo et al. (2011) record the presence of informal sectors in selected developing countries, including China, India, Kenya, Nigeria, and Argentina. They conclude that the potential impact of informal WEEE recycling to whole nation recycling systems might appear if the informal sectors could conform to the standards. These positive notions are also supported by an in-depth report from GIZ (2011). GIZ conducted several initial projects to integrate informal sector into solid waste management systems. The projects include different approaches such as integrated waste management planning, capacity building, and joint-partnership between formal and informal sectors. This GIZ report provides success stories of the initiatives using several cases from Brazil, Costa Rica, Philippines, and Mozambique.

It is worth to mention that the number of quantitative research that captured WEEE informal sectors is still limited. Streicher-Porte et al. (2007), as one of very few exceptions, introduce a dynamic stock-driven material flow model and an economic evaluation of gold and copper flows based on Indian PC recycling sector. They compare the behaviors of formal and informal recycling sectors using scenario analysis and conclude that the formal sector will not be able to push the informal sector out of the market. Li and Tee (2012), another exception, develop a mixed integer multi-objective linear programming reverse logistics model to assess the economic, environmental, and health benefits of integrating informal sectors into formal ones. Their model can determine which options/mix of options would be attractive to the informal sectors to leave their activities and integrate with formal sectors.

Besiou et al. (2012) provide a further stepping-stone study for this research cluster by conducting a holistic analysis using System Dynamics (SD). Their work aims to assess the impact of informal scavenging into the overall systems of WEEE management systems, using sustainability aspects as the indicators. Particularly, the authors modeled the systems under three scenarios: (a) informal scavenging exists, but it is ignored by the regulation, (b) informal scavenging ceases to exist, and (c) informal scavenging is integrated by the regulation with the official collection. The authors concluded that the whole recovery systems might be more sustainable by the integration of the informal sector. While their work offers valuable insights, the model structures leave room for the future research. The inclusion of informal activities in their SD model as merely scavenging would be not enough to represent the complexity of the informal sectors in reality, especially in most developing countries.

The SD methodology, founded by Forrester (1961), aims to understand the interconnection among elements of the system to achieve a particular goal/set of goals (Meadows, 2008). SD models consist of the stocks and flows, feedback loops, and nonlinearities formed by interactions among physical and information structures and the decision-making process (Sterman, 2000). Altogether, it might reproduce a typical dynamic behavior over a particular period (Vlachos et al., 2007). In general, SD incorporates two main tools: causal-loop diagram and stock-flow diagram. Firstly, the causal-loop diagram visualizes the relationships among variables and the feedback structure within the system. Secondly, the stock-flow diagram depicts the mathematical formulation behind the model.

SD is suitable to model the real world problems that are identified by uncertainty, dynamics, time delays, and conflicting goals of multiple stakeholders (Besiou and Van Wassenhove, 2015; Van Wassenhove and Besiou, 2013). These authors study the preceding characteristics in several real-world problems, including also WEEE reverse logistics, and conclude that it would not be adequate to solve the real world problems with the mentioned characteristics, by relying only on optimization methods. Hence, our study proposes SD as proper modeling approach to capture the complexity of the informal sector.

3. Methodology

3.1. Generic conceptual model

Fig. 1 exhibits the simplified conceptual model of the system under study. It consists of three sub-models: the households, the reverse logistics and the dynamics within informal sectors. Households sub-model represents the behaviors of customers that buy and utilize the electronic products. For simplicity, this study restricts the user of the products as only coming from the households, excluding the businesses and the administrative offices. Later on, the customers dispose of the products, as WEEE, which then flows into the reverse logistics' channels. The reverse channels contain both formal and informal WEEE recovery systems. On the one hand, the structure of the formal sector is simplified in the model by making collection activity as its only representation. On the other hand, the informal sector includes different types of recovery operations, i.e. collection, reuse, refurbishment, recycling, secondary market and landfilling. This research further hypothesizes the endogenous dynamics that drive the growth of the informal sector as will be further discussed in the next sections.

3.2. Households sub-model

This sub-section provides households sub-model to illustrate the behavior of customers in buying and utilizing the electronic products and then disposing of these products in the end-of-life. To capture the preceding behaviors, this study adopts Input–Output Analysis (IOA) approach, the most common methodology to estimate WEEE generation in the literature (Wang et al., 2013a, b). IOA consists of three main variables: sales, stock, and lifespan.

Firstly, the "sales" variable is developed by adapting the structure of the Bass model (1969) taken from Sterman (2000). Secondly, the "stock" element is simply captured by utilizing the existing stock variable in SD modeling. Thirdly, the "lifespan", representing a specific time gap between the purchasing and disposal activities, is taken by combining the delay structure as Besiou et al. (2012) with the derived lifetime distribution from the Market Supply Model approach (Sinha, 2013).

Fig. 2 shows the households sub-model that combines the causal-loop diagram and the stock and flow diagram. The causal links, shown by the arrows, represent causal influence from one variable into another variable. As explained by Sterman (2000), one the one hand, the positive sign (+) means "if the cause increases (decreases), the effect increases (decreases) above (below) what it would otherwise have been". On the other hand, the negative sign (−) means the opposite direction from the previous definition. The stock variables, pictured as rectangular, and the flow variables, figured as valves and pipes, represent the accumulation and the flow equations, subsequently. The variable names are presented in italics in the remaining manuscript. The structures of the Bass Model and the detailed explanation for Fig. 2 are provided in Appendix 1 of the supplementary material.

3.3. Reverse logistics sub-model

Generated WEEE from households enters the reverse channels, either to formal channel or informal channel. The nature of collection competition, between these two sectors, determines the fate

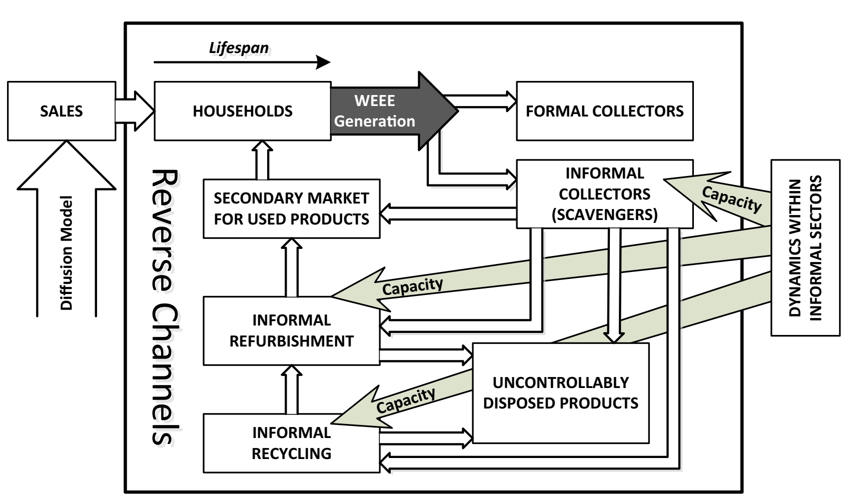

Fig. 1. Simplified conceptual model for the system under study.

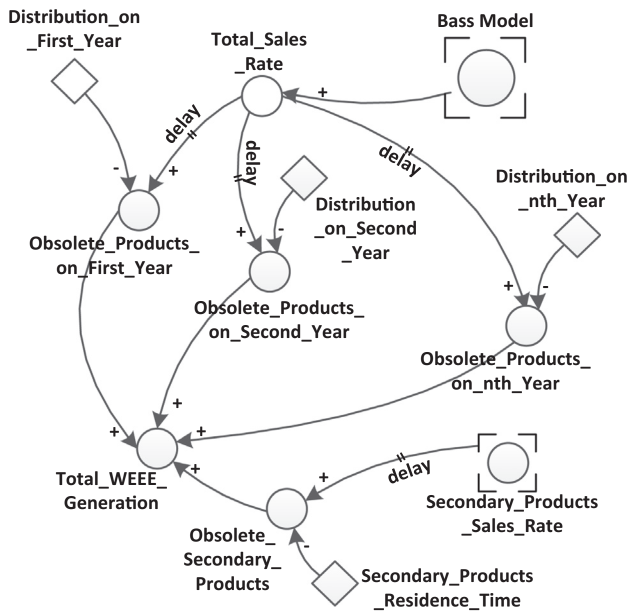

Fig. 2. The structure of households sub-model.

of the WEEE. This research proposes two conditions of collection competition. In the first situation, the formal system is assumed to have superior access to WEEE collection, gathering WEEE as much as the highest capacity and leave the rest of WEEE, if any, to the informal channel. Besiou et al. (2012) and Streicher-Porte et al. (2007) also use this idea in their works. In the second condition, the informal sector is captured as the superior actor in collection activities, representing the reality in many developing countries. Accordingly, the informal sector can collect WEEE at its maximum capacity because of the door-to-door operations. In any case, if both formal and informal sectors cannot collect all of the WEEE, the uncollected ones will flow directly to disposal. Fig. 3 depicts the generic version of the reverse logistics sub-model. The detailed description of Fig. 3 is provided in Appendix 2.

3.4. Sub-model: dynamics within the informal sector

This research hypothesizes the endogenous dynamics that occurs within the informal sector, causing the growth of informal sector. These dynamics further become the source of the capacity in running the collection, refurbishment, and recycling activities. In shorter periods, the dynamics ensure the continuity of the daily operations and on the longer run, it maintains the increasing state of the informal sectors.

3.4.1. Causal-loop diagram of the dynamics within the informal sector

Fig. 4 shows the simplified causal-loop diagram of this dynamics. The causal-loop diagram consists of five reinforcing (R) loops and 3 balancing (B) loops. Loop R1 characterizes the role of the secondary market to absorb the recovered products from informal channels and to satisfy the demand for second-hand products. In this loop, an increase in *Informal_Collection_Rate* increases *Secondary_Products_Inventory*, influencing the rise of *Secondary_Products_Sales_Rate*. After the time equal to *Secondary_Products_Residence_Time*, the second-hand products become obsolete, once again raising *Total_WEEE_Generation*. Hence, loop R1 causes *Informal_Collection_Rate* to increase even higher.

Loop R2 until R4, modified from Vlachos et al. (2007), represent the dynamics of the informal capacity. In these loops, the informal sector estimates the future capacity using smoothing factor to the level of current routines and then adjusts the current number of informal workers by hiring more workers. These loops have similar structures that include collection (R2), refurbishment (R3), and recycling (R4) activities. In loop R2, an increase in *Total_WEEE_Generation*, rises *Desired_Informal_Collection_Capacity*, triggering *Informal_Collection_Capacity_Discrepancy* to grow. Hence, *Desired_Additional_Informal_Workers* increases, further affecting the increase of *Desired_Employment_Rate*. This condition influences the increasing number of *Informal_Workers* through *Net_Employment_Rate*. The growing size of the workers causes the rise of *Informal_Collection_Capacity*, increasing *Informal_Collection_Rate*. This relationship then affects the increase of *Secondary_Product_Inventory*, increasing the number of *Secondary_Products_Sales_Rate*. After a specific usage period, the *Total_WEEE_Generation* increases again, closing the loop of R2.

Loop R5 signifies the influence of the profitability into the informal recovery operations. In loop R5, an increase of *Informal_Collection_Rate*, increases the availability of *Secondary_Products_Inventory*, influencing the rise of *Secondary_Products_Sales_Rate*. Hence, informal sector receives higher *Informal_Revenue*, growing the stock of *Informal_Cash_Availability*. The cash availability maintains the routines of informal collection, increasing *Informal_Collection_Rate* into even higher.

In loop R6, the causal links depict the influence of cash availability into the rise of informal capacity. Again, an increase in *Informal_Collection_Rate*, raises *Secondary_Products_Inventory* and further increases the stock of *Secondary_Products_Sales*. Thus, the level of *Informal_Revenue* raises, increasing the number of *Informal_Cash_Availability*. The cash availability further affects *Informal_Employment_Decision* to increase the number of *Net_Employment_Rate*, causing the rise of *Informal_Workers*. As consequence, *Informal_Collection_Capacity* increases, influencing the higher rise of *Informal_Collection_Rate* and closing the loop.

The balancing loops consist of loop B1 to B3 and depict the fulfillment of capacity discrepancy after the hiring process has been taken. In B1, an increase of *Informal_Capacity_Discrepancy* increases *Desired_Additional_Informal_Workers*. Hence, *Desired_Employment_Rate* increases, pushing informal actors to increase the number of *Informal_Workers* through *Net_Employment_Rate*. The raising level of *Informal_Workers* causes the rise of *Informal_Collection_Capacity*, closing the occurred gap in *Informal_Collection_Capacity_Discrepancy*.

3.4.2. Stock-flow diagram of the dynamics within the informal sector

The generic stock-flow diagram of the dynamics is presented in Fig. 5. In this figure, the stock of *Informal_Workers* is increased by *Employment_Rate* and decreased by *Unemployment_Rate*. *Employment_Rate* depends on *Desired_Employment_Rate* which is influenced by *Desired_Additional_Informal_Workers*, *Hiring_Approval_Decision*, and *Time_to_Adjust_Workers*. The equation of these relationship is as follows:

$$\begin{aligned}&Desired\_Employment\_Rate\\&\quad = (Desired\_Additional\_Informal\_Workers/Time\_to\_Adjust)\\&\qquad * (Hiring\_Approval\_Decision)\end{aligned} \tag{1}$$

$$\begin{aligned}&Desired\_Additional\_Informal\_Workers\\&\quad = PULSE((Desired\_Additional\_Informal\_Collectors\\&\qquad + Desired\_Additional\_Informal\_RecovWorkers), STARTTIME\\&\qquad + Pc\_A, Pc\_A)\end{aligned} \tag{2}$$

Furthermore, *Unemployment_Rate* comprises two types of layoff: (1) *Normal_Layoff_Rate*, which is influenced by *Time_to_Layoff_Workers*, and (2) *Acute_Layoff_Rate*, which is affected by *Time_to_Acute_Layoff_Workers* and *Acute_Layoff_Decision*. The two

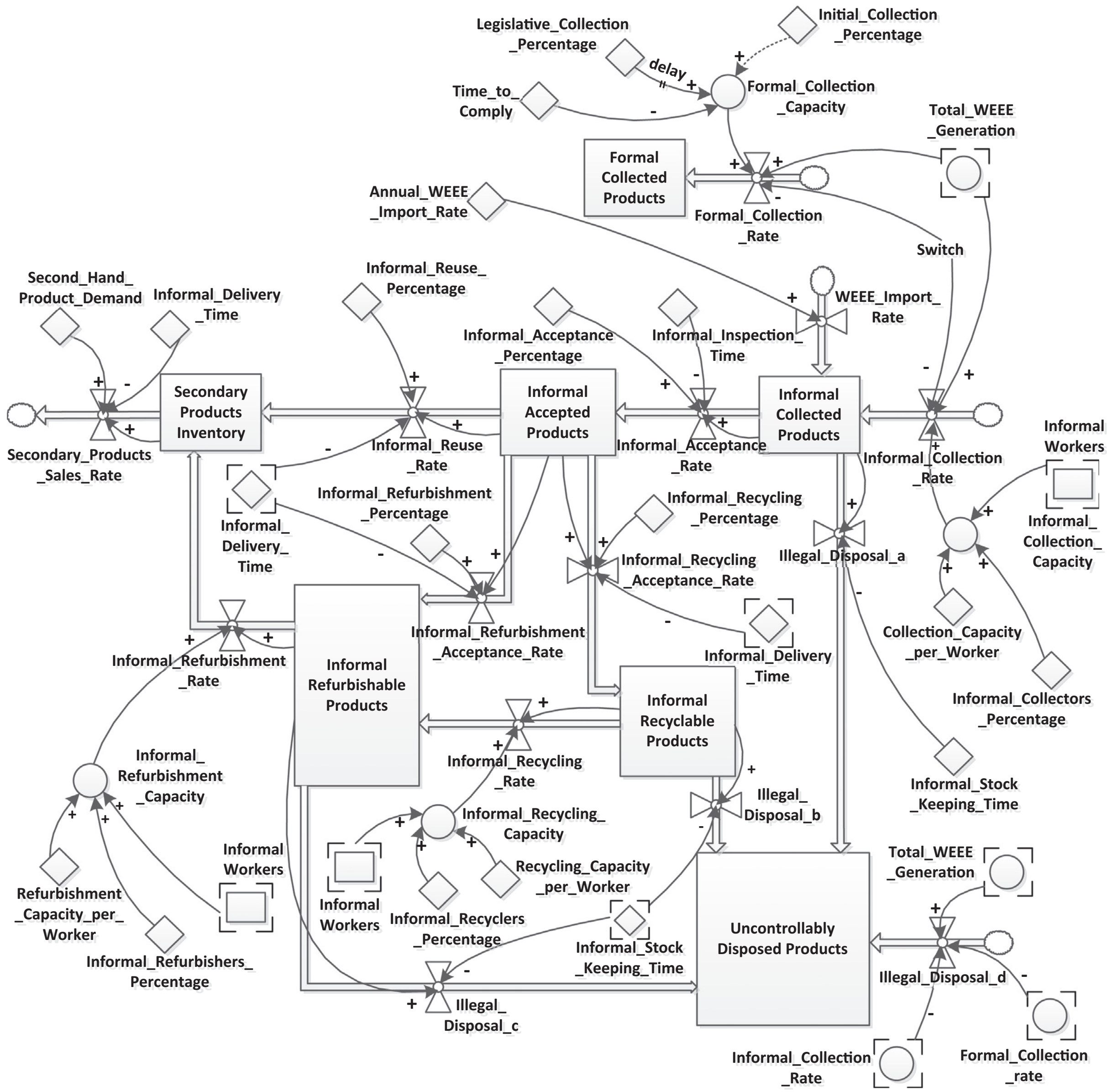

Fig. 3. The structure of reverse logistics sub-model.

decision-making processes of *Informal_Employment_Decision*, i.e. *Hiring_Approval_Decision* and *Acute_Layoff_Decision*, are further discussed in the following part of the paper.

Fig. 5 also illustrates the financial structure of the informal sector. *Informal_Cash_Availability* is influenced by *Informal_Revenue* and *Informal_Cost*. Moreover, *Informal_Cash_Availability* affects *Informal_Collection_Rate* through *Informal_Operations_Decision*. This particular decision is also further discussed in the next part.

Total revenue and total cost faced by informal sector are simply calculated from:

$$\begin{aligned} &Informal\_Revenue \\ &\quad = (Value\_per\_Recovered\_Product * Secondary\_Sales\_Rate) \\ &\qquad + (Value\_per\_Recycled\_Product * Informal\_Recycling\_Rate) \end{aligned} \quad (3)$$

$$\begin{aligned} &Informal\_Cost \\ &\quad = (Recovery\_Cost\_per\_Product * Informal\_Refurbishment\_ \\ &\qquad Acceptance\_Rate) + (Recycling\_Cost\_per\_Product \\ &\qquad * Informal\_Recycling\_Acceptance\_Rate) \end{aligned} \quad (4)$$

3.4.3. Decision-making processes within the informal sector

The model under study incorporates three decision-making processes, i.e. *Informal_Operation_Decision, Hiring_Approval_Decision*, and *Acute_Layoff_Decision*. These processes are influenced by two situations of cash availability: a declining state and a limited stock of cash availability. Firstly, if the amount of the cash declines, the managers would adjust their collection operations and approval of the regular employment through *Informal_Operation_Decision* and *Hiring_Approval_Decision*, subsequently. Secondly, if the limited state of the cash appears, the informal decision makers would lay-off all of their workers through *Acute_Layoff_Decision*. Fig. 6 captures the detailed stock-flow diagram of these decision-making structures.

In Fig. 6, the increasing/declining state of *Informal_Cash_Availability* is tracked by *Cash_Ratio*, which represent the comparison between the current and the expected future value of cash. This decision structure is represented as follows:

$$\begin{aligned} Expected\_Informal\_Cash = DELAYINF(&Informal\_Cash\_Availability, \\ &a\_EIC, 3, Informal\_Cash\_Availability) \end{aligned} \quad (5)$$

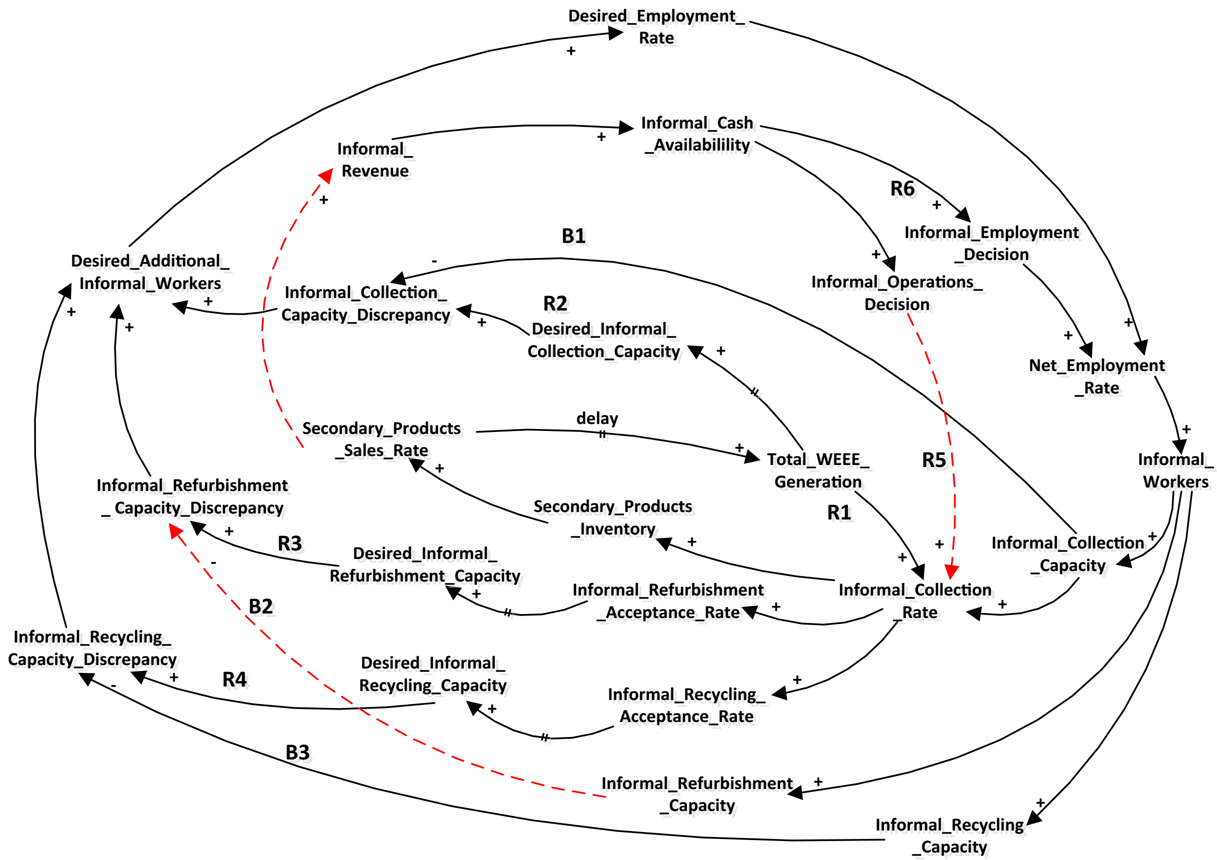

Fig. 4. The simplified causal-loop diagram of endogenous dynamics within the informal sector.

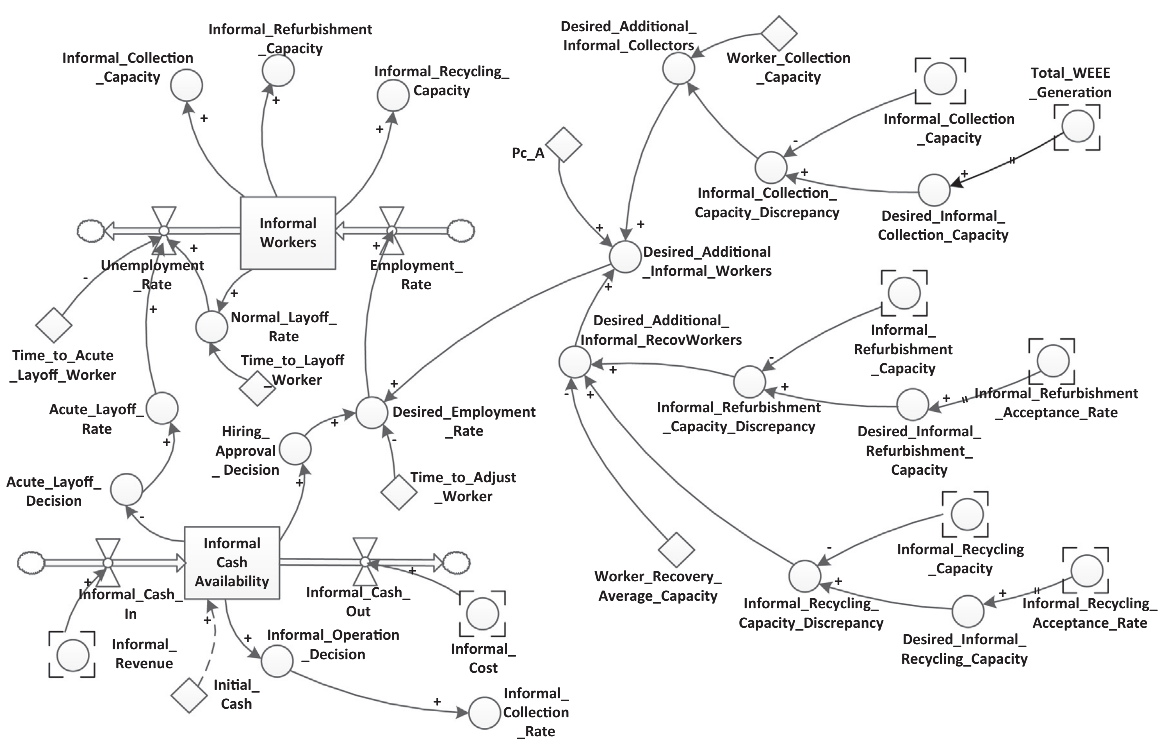

Fig. 5. The generic stock-flow diagram of endogenous dynamics within the informal sector.

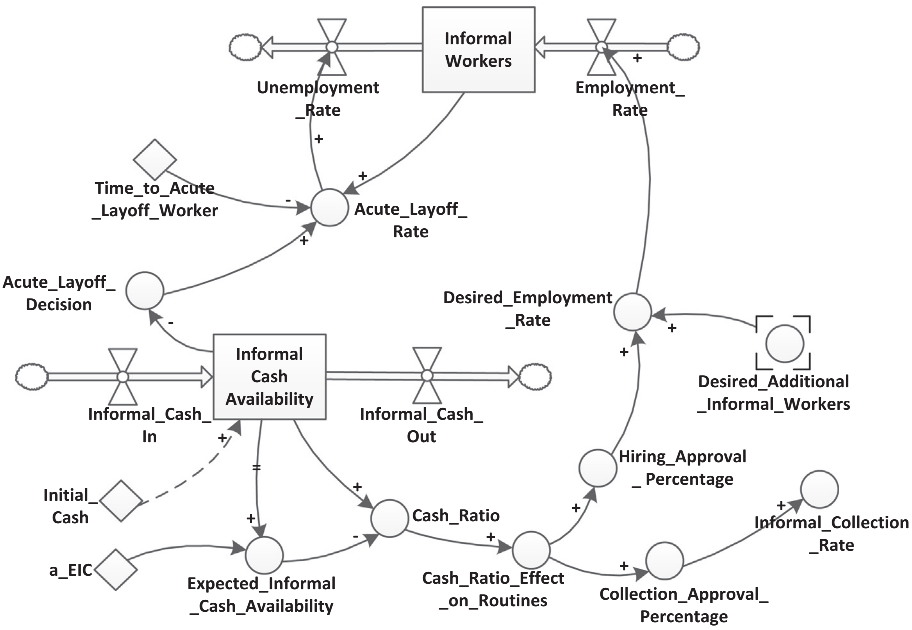

Fig. 6. The decision – making structures based on the level of informal cash availability.

$$Cash\_Ratio = Informal\_Cash\_Availability/Expected\_Informal\_Cash \tag{6}$$

$$Cash\_Ratio\_Effect\_on\_Routines = IF(Cash\_Ratio > 1, 100 << \% >>, GRAPH) \tag{7}$$

$$Hiring\_Approval\_Percentage = Cash\_Ratio\_Effect\_on\_Routines \tag{8}$$

$$Collection\_Approval\_Percentage = Cash\_Ratio\_Effect\_on\_Routines \tag{9}$$

GRAPH function in Eq. (7) represents the three plausible adjustment behaviors by informal managers, i.e. proportional, highly sensitive, and insensitive behaviors (Fig. 7).

Lastly, the decision-making structures are complemented by *Acute_Layoff_Decision*. In this structure, if the declining state passed a certain low level of *Informal_Cash_Availability*, the informal manager would activate *Acute_Layoff_Decision*. The equation of these relationships is as the following:

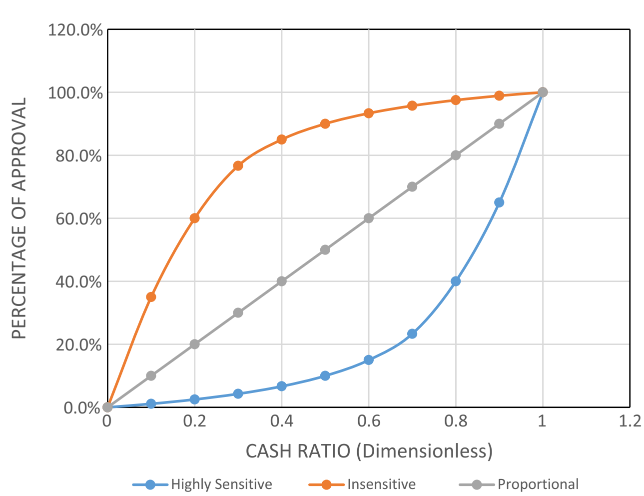

Fig. 7. Three plausible adjustment behaviors of informal managers.

$$Acute\_Layoff\_Decision = IF\ (Informal\_Cash\_Availability < 10000 << USD >>, 1, 0) \tag{10}$$

$$Acute\_Layoff\_Rate = (MAX\ ((Informal\_Workers/Time\_to\_Acute\_Layoff\_Worker), 0)) * Acute\_Layoff\_Decision \tag{11}$$

4. Formal model formulation and testing

This study employs the data from India to assess the behavior of the model under consideration. India is chosen as the reference system because of three reasons. Firstly, the informal sector exists significantly in the systems. It is estimated that 1% of the Indian population is involved in the informal waste sector (Chikarmane et al., 2008). These informal workers are recognized as *Kabadiwalas* (waste collectors/dealers), *Thailawalas* (collectors), small *Kabaris* (small scrap collectors), and big *Kabaris* (large scrap collectors) (Pandey and Govind, 2014; Wath et al., 2010). Secondly, previous literature provides rich studies on Indian WEEE recycling sectors and their characteristics. These studies include empirical studies, comparative analyses between Indian and developed systems, estimations of WEEE generation, and the status of the informal WEEE recycling systems (Chaturvedi et al., 2008; Dwivedy and Mittal, 2010a, 2010b; Khetriwal et al., 2009; Raghupathy and Chaturvedi, 2013; Raghupathy et al., 2011; Sinha-Khetriwal et al., 2005; Streicher-Porte et al., 2007, 2005). Lastly, India will emerge as a major WEEE producer in the next decade without yet having effective regulatory approaches (Pandey and Govind, 2014; Premalatha et al., 2014). India started to face the WEEE problems on the early 1990s after the first period of its market liberation (Wath et al., 2010). This country is experiencing, not only the rise of WEEE generation rate but also the problems when dealing with informal sectors. Not until 2011 did India finally introduce a specific regulation addressing the WEEE problems, by issuing "E-Waste (Management and Handling) Rules" (MoEF, 2011). This approach applies distinct responsibilities to Original Equipment Manufacturers (OEMs) of electronic products, consumers, bulk consumers, recyclers, collection centers, and dismantlers of WEEE. Remarkably, it also provides a door for the informal sector to become a member of formal WEEE management, as collection centers or dismantlers, through formalization (CERAG, 2013). The effectiveness of this WEEE regulation, however, is still questioned as the weak enforcement of the regulation and inadequate infrastructures persist (Pandey and Govind, 2014).

4.1. Data gathering and parameter setting

On account no reliable historical data on formal and informal recycling sectors in India, this research utilizes and synthesizes data from various sources to assess the behaviors of the model subjected to the purpose of the study, including published scientific papers, published reports, census data (Census of India, 2011, 2001), and regulation text. The data is treated, adapted and modified when necessary. To estimate the parameters of the Bass' Model, the authors conduct GRG non-linear method in Excel Solver (Frontline System Inc., 2013) using historical sales data of Indian desktop personal computer (PC) from 1994 to 2012 (Dwivedy and Mittal, 2010b; MAIT, 2013). Other further important parameters are provided in Table 1.

4.2. Model testing

This sub-section incorporates the model testing steps taken from Sterman (2000). Firstly, a model boundary adequacy and structure assessment tests were conducted through literature reviews and a set of colloquiums. These tests clarify the importance of incorporating the informal sector as an endogenous element in the model. Secondly, the study inspected directly the mathematical equations behind the model to assess the dimensional consistency and found no suspect variables. Thirdly, to reveal flaws in the model and to assess its robustness, the extreme condition test was performed by putting an extreme value to several selected variables. For instance, if there were no innovative adopters at the beginning of life-cycle, i.e. *Innovation_Fraction* is "0", there would be no adopters of the products in all of the life-cycles; thus sales rate would remain on zero level through entire simulation period. Additionally, if the informal sector collects no obsolete WEEE from the households and in the same time there is no imported WEEE from developed countries, the number of informal workers would never grow up.

Fourthly, a numerical integration test was carried out to assess the acceptability of the selected integration method, i.e. Euler integration. The test was done by choosing a time step one-fourth of the smallest time constant and running the model. After that, the time step was cut in half, and the model was run again. The result showed no significant difference between the observed behaviors. Lastly, behavior reproduction test was done to assess the ability of the model to reproduce the historical time series or reference modes. This research selects *Total_Sales_Rate* and *Total_WEEE_Generation* as the main indicators for the assessment. For *Total_Sales_Rate*, the test compared the historical and simulated data from 1994 to 2007 (Fig. 8). The series from 2008 and afterward are omitted in this comparison because the 2008 global crisis has arguably affected the Indian economy. With Mean Absolute Percentage Error (MAPE) of 11.66%, the model showed fairly good predictive ability in this particular variable. Our study further compared the behavior of *Total_WEEE_Generation* (see Appendix 3) with the results from other studies dealing with Indian PC waste generation (Dwivedy and Mittal, 2010a,b). This assessment test found that the SD model produced similar modes with the reference studies as partly shown in Table 2.

5. Simulation analysis: base case and scenario analysis

Simulation analysis consists of base case and scenario analysis. Different assumptions were put in each of them. Afterward, the SD

Table 1
Parameter values for model testing.

Variable	Value	Description	Data source
Innovation_Fraction	0.001522	Coefficient of innovation in bass model	Dwivedy and Mittal (2010a,b) and MAIT (2013)
Adoption_Fraction	0.04752	Coefficient of imitation in bass model	
Contact_Rate	5.1447	Frequency of contact between potential adopters and adopters	
Average_Consumption_per_Adopter (Units/house/year)	0.2	Value of PC replacement rate	World Bank (2010)
Distribution_on_First_Year (%)	0	Percentage of products that obsolete in the first year of usage period	Yu et al. (2010)
Distribution_on_Second_Year (%)	0	Percentage of products that obsolete in the second year of usage period	
Distribution_on_Third_Year (%)	0	Percentage of products that obsolete in the third year of usage period	
Distribution_on_Fourth_Year (%)	20	Percentage of products that obsolete in the fourth year of usage period	
Distribution_on_Fifth_Year (%)	70	Percentage of products that obsolete in the fifth year of usage period	
Distribution_on_Sixth_Year (%)	10	Percentage of products that obsolete in the sixth year of usage period	
Initial_Collection_Percentage (%)	5	Collection percentage at the beginning of simulation period	Authors' own assumption due to no available data
Legislative_Collection_Percentage (%)	60	Collection percentage imposed by regulation	
Time_to Comply (Years)	7.5	The gap time required by the systems to comply with regulation	
Secondary_Residence_Time (Years)	3	Average time of second-hand product to become obsolete	Dwivedy and Mittal (2010b)
WEEE_Import_Rate (Unit/year)	1,650,000	Number of imported WEEE from developed country	Manomaivibool (2009)
Informal_Acceptance_Percentage (%)	95	Percentage of WEEE accepted as recoverable products	CERAG (2013)
Informal_Reuse_Percentage (%)	1.8	Percentage of WEEE accepted as reusable products	Streicher-Porte et al. (2005)
Informal_Refurbishment_Percentage (%)	23.8	Percentage of WEEE accepted as refurbish-able products	Streicher-Porte et al. (2005)
Second_Hand_Product_Demand (Unit/day)	786	Daily demand for second-hand PC in reuse market	Streicher-Porte et al. (2005)
Value_per_Recovered_Product (Dollar/unit)	366	Revenue per second-hand product sold in reuse market	Streicher-Porte et al. (2005)
Recovery_Cost_per_Product (Dollar/unit)	183	Cost per product for recovery activities	Streicher-Porte et al. (2005)
Value_per_Recycled_Product (Dollar/unit)	4.05	Revenue of recycled material per product	Streicher-Porte et al. (2007)
Recycling_Cost_per_Product (Dollar/unit)	2.5	Cost per product for recycling activities	Streicher-Porte et al. (2007)

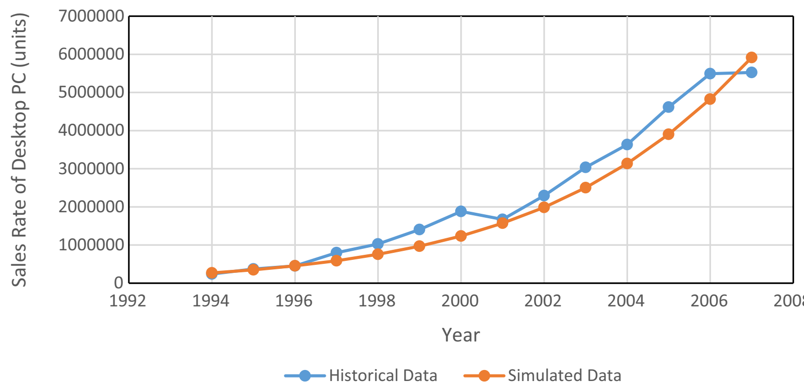

Fig. 8. Comparison between historical and simulated data of *Total_Sales_Rate*.

Table 2
Comparison between parameter values from the model and the references.

Parameter	*Total_WEEE_Generation*[a] – in units	Estimated obsolete PC generation (Dwivedy and Mittal, 2010b)[b] – in units	Estimated obsolete PC generation (Dwivedy and Mittal, 2010a)[c] – in units
Estimated value for 2010	7.45 million	10.66 million	5.52 million
Estimated value for 2015	20.02 million	52.58 million	–
Estimated value for 2020	41.46 million	79.98 million	–
Estimated value for 2025	64.45 million	92.14 million	–

[a] Desktop PC only, considering no store phase.
[b] Desktop and notebook PC, considering store phase.
[c] Desktop PC only, considering no store phase. The store phase is showed separately.

model was simulated using *Powersim* 10 ® for 20 years of simulation period in two conditions of collection competition: (1) the informal sector has superior access to obsolete products, or (2) the formal sector has the superior access. Then, the results were analyzed using several selected indicators.

5.1. Base case analysis

In the base case analysis, this study runs the model using the basic parameters from Table 1. Note should be taken that one important parameter is relaxed in this particular analysis: *Second_Hand_Product_Demand* is assumed to have constant value during the entire simulation horizon. This assumption aims to assess what would the system behave if the secondary market for used products is stagnant and unprofitable. Additionally, the proportional adjustment behavior is selected from Fig. 7 as input to *Cash_Effect_on_Routines*. Figs. 9–11 illustrate the results of the annual WEEE collection rate, the number of informal workers, and the availability of informal cash in the base case analysis. An additional result of the base case analysis is provided in Appendix 4.

In general, Fig. 9 depicts the increasing state behaviors from both formal and informal collection. It is further observed that the informal sector dominates the collection activities, in more than three-quarters of the simulation period. The formal sector is unable to gather adequate WEEE in the early period. It is even worse when the informal sector has superior access to obsolete products; the formal sector would not even operate normally until the 8th year. Not until the fourth quarter did the formal collection finally surpass the informal one. On a practical level, it implies that the formal sector in developing country requires a relatively long period to establish itself and finally finds its way to becoming a dominant player in collection activities. Additionally, this result is remarkable because the structure of formal sector in this study is simplified to focus on the informal sector. If this study disaggregated the formal sectors in detail and incorporated the financial availability as the driver for formal recovery activities; most likely, the formal system would face complete failure because of the lack of source and limited revenue. Lastly, the results shown in Fig. 9 confirm the realities in which the formal WEEE recycling faces the shortage of collected products (Kojima et al., 2009).

The results shown in Fig. 10 illustrate the growing state of the number of informal workers. The number grows, in average, by 11.26% and 10.08% per year for the case of superior informal sector and superior formal sector, consecutively. In the absence of any elements to divert the net flow of informal workforce, except the generic *Unemployment_Rate*, it seems that the size of informal worker might raise continuously in the near future.

The expected continuous growth, however, would not happen in the base case analysis as Fig. 11 shows clearly that there is a limit in the informal growth. After the rapid growth in the first half of simulation period, the level of informal cash started to decline continuously in the 11th and the 15th year for the case of informal superiority and formal superiority, subsequently. A joint examination of the sub-models, especially on "Dynamics within the Informal Sector", shows that the constant level of *Second_Hand_Product_Demand* appears to be the limit of growth. While the cost continued to rise because of the increasing state of the recovery operations and employment activities, the constant demand restricted the revenue. Hence, the loop dominance shifted from the reinforcing to the balancing state and, inevitably, the informal sector would run out of cash.

Lastly, Fig. 11 also depicts one interesting behavior: the profitability of the informal sector would be higher when the informal sector has no superior access for the collection. Instead, the informal cash would reach a higher level and remain profitable longer if the formal sector is preferable for disposal activities. The most plausible explanation for this behavior is as explained by Donella H. Meadows in her book, "Thinking in System" (2008): "the higher and faster you grow, the farther and faster you fall, when you're building up a capital stock dependent on a nonrenewable resource". In the environment with stagnant secondary market – similar with nonrenewable resources, the inferior position of informal collection

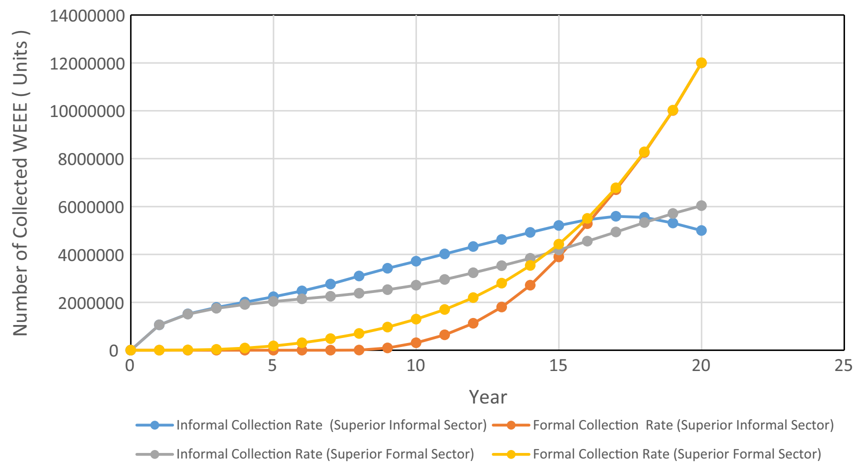

Fig. 9. Comparison of annual collection rate between formal and informal sector in base case analysis.

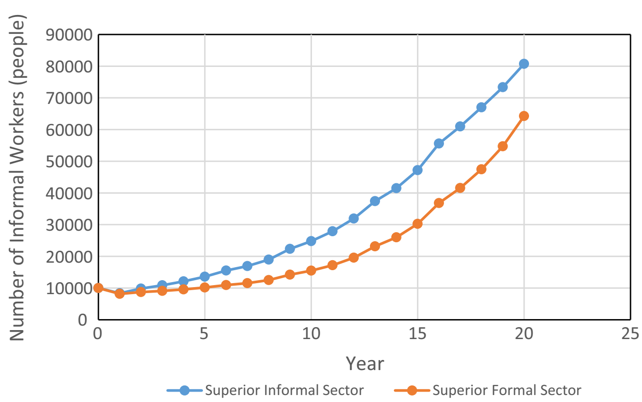

Fig. 10. Comparison of the number of informal workers between two types of collection in base case analysis.

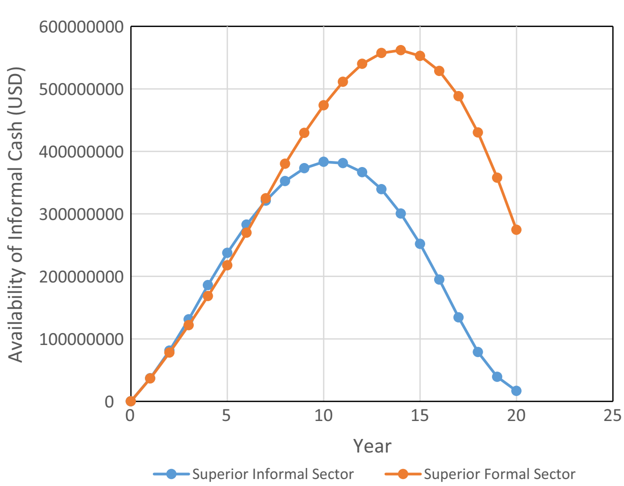

Fig. 11. Comparison of the availability of informal cash between two types of collection in base case analysis.

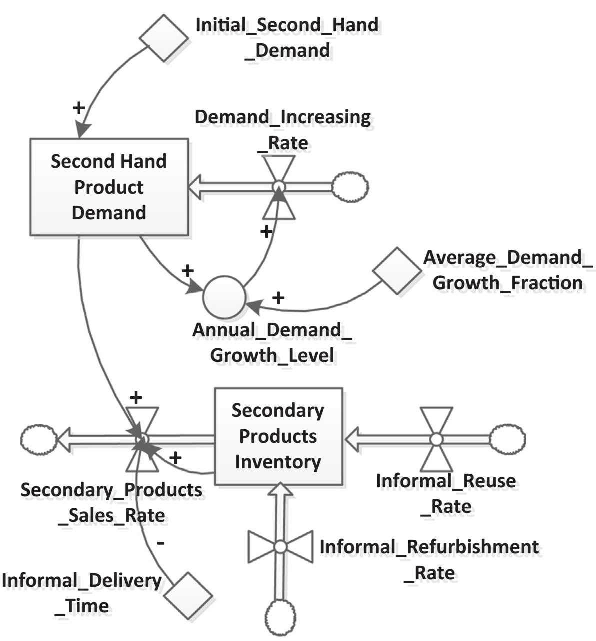

Fig. 12. The structure of *Second_Hand_Product_Demand* in scenario analysis.

slows the informal growth, delaying the informal cash level to reach its peak and holding this sector to exist in longer period.

5.2. Scenario analysis

Scenario analysis is developed to investigate the effect of changes in parameter value and model structure. Particularly, this analysis aims to assess the influence of growing second-hand market to the dynamics within the informal sector. It is carried out by giving a minor modification to the structure of *Second_Hand_Product_Demand* so its value will grow every year (Fig. 12).

Instead of being constant, *Second_Hand_Product_Demand* is treated as stock and increased by *Demand_Increasing_Rate*. This rate depends on *Annual_Demand_Growth_Level*, calculated from the current level of *Second_Hand_Product_Demand* and annual growth rate of the secondary market (*Average_Demand_Growth_Fraction*). This study follows Suryani et al. (2010) that added random exponential distribution to the average demand growth in their case. The equation of the growth rate is as following:

$$Demand\_Increasing\_Rate = Annual\_Demand\_Growth\_Level \tag{12}$$

$$\begin{aligned} &Annual\_Demand\_Growth\_Level \\ &\quad = Second\_Hand\_Product\_Demand * (Average\_Demand\_Growth\_ \\ &\qquad Fraction + EXPRND(1 << \%/year >>)) \end{aligned} \tag{13}$$

With "*EXPRND (1 <<%/year>>)*" is used as a command in *Powersim* ® to generate random numbers that are exponentially distributed with 1% of the mean value.

To implement this scenario, this study employs a growth of 15% per annum for *Average_Demand_Growth_Fraction*. This number was taken from a report published by the Associated Chamber of Commerce and Industry of India about the market for second-hand and recycled products (ASSOCHAM, 2014). The value is generic in nature since the specific number for second-hand PC market was not found. To analyze the longer horizon, the model was run for 30 years of simulation period. By also considering the types of collection competition (formal or informal superiority), this study compares: (1) the results from the growing market with the stagnant case using selected indicators i.e. *Informal_Collection_Rate*, *Informal_Workers*, and *Informal_Cash_Availability* (Figs. 13–15), (2) the results from the annual collection rate between the two sectors in the growing used market case (Fig. 16). An additional result of the scenario analysis is provided in Appendix 5.

Fig. 13 illustrates two different behaviors: the steady growth of informal collection in the environment of growing used market and the shifting dominance in the stagnant market. The growth in the former case accelerated significantly after the 14th year of simulation period; while in the same period, the latter nearly shifted the loop dominance from the increasing into the declining state. The growth acceleration occurred even sooner in the case of informal superiority of collection access, which happened in 11th year. This finding suggests two things: (1) the significant influence of second-hand market condition into the level of informal collection, and (2) the influence, but less significant, of the superiority in accessing WEEE from households into the same indicator.

Figs. 14 and 15 show the continuous increasing state of informal sectors in the growing used market. The number of informal workers grows on the average level of 14.39% and 12.33% for the case of informal and formal superiority access in collection, subsequently. For the level of informal cash availability, the growing rate appears slightly higher around 24% of average growth for both collection conditions. In contrast to the growing case, this study observes the collapsing state of the informal sector in a stagnant market environment as the informal cash was approaching the zero level in the last half of simulation horizon.

Fig. 16 reveals the tight competition between the formal and the informal collection for more than 15 years of the simulation period. After that, the state of both collections started to be different. Nevertheless, the collection rate of the informal sector remains in the steady increasing state in the entire horizon. Even the superiority of formal sector could not push the informal collection to cease. Moreover, if the informal sector has the superiority in the collection, it is clearly seen that the formal sector almost reaches its peak. Most likely, if the horizon of the simulation period is extended, the cease of the formal sector might be seen.

The phenomena mentioned in the previous paragraphs indicates that the conditions in the second-hand market significantly affect the existence of the informal sector, while at the same time potentially influence the end of formal sectors. The results of the

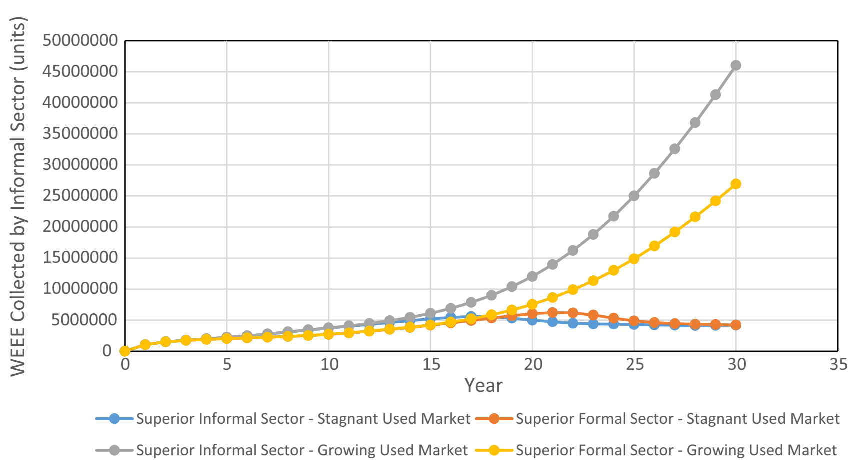

Fig. 13. Comparison of informal collection rate between stagnant and growing used market.

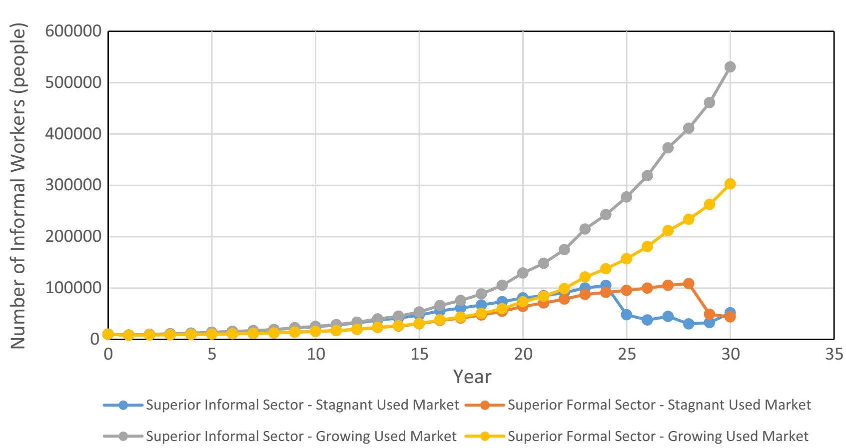

Fig. 14. Comparison of the number of informal workers between stagnant and growing used market.

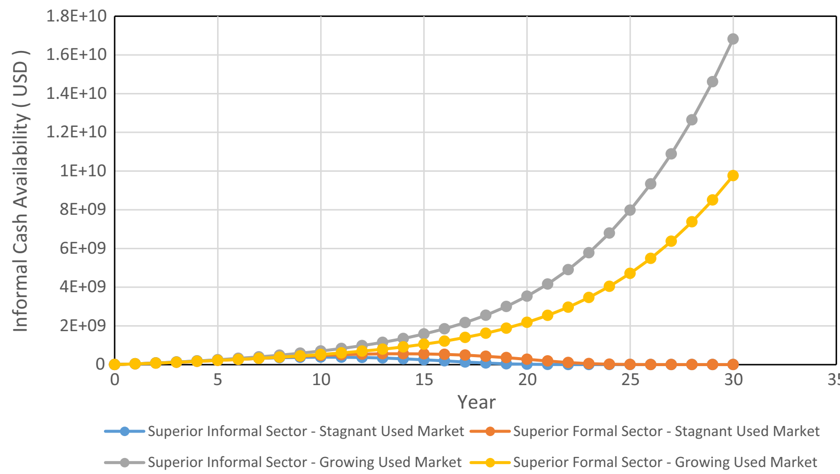

Fig. 15. Comparison of the availability of informal cash between stagnant and growing used market.

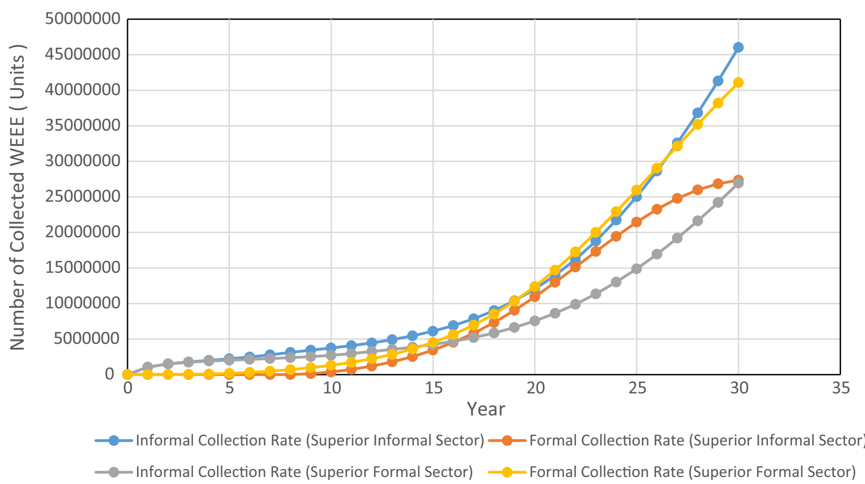

Fig. 16. Comparison between informal and formal collection rate in the growing used market.

scenario analysis complement the base case results: the second-hand market appears as both the limit (when it is constant) and the leverage of the informal growth (when it is growing). As consequence, it is concluded that *Second_Hand_Products_Demand* appears as an important parameter in the model under study. On the practical level, the results shown in Fig. 14 until 16, confirm the influential position of the second-hand market as the determinant for the informal WEEE recycling, as can be seen in the cases of India and China (Manomaivibool, 2009 and Chi et al., 2011). In the absence of any integrated approach to slow or even to divert the drivers of informal growth, the profitability of the second-hand market will still highly influence the growth of the informal sector in the foreseeable future.

6. Conclusion

This paper aims to understand the complexity and the dynamics within informal sector dealing with WEEE recovery operations. Specifically, this study assesses the effect of the dynamics to the nation-wide WEEE management systems using SD methodology.

The main important finding appears from the analysis: the SD model in this paper can explain the dynamics occurring within the informal sector. The dynamics cause, on the one hand, the failure of formal collection and, on the other hand, the growth of the informal sector. The dynamics are structured and determined by several interrelated factors: (1) the availability of the WEEE coming from households and illegal import, (2) the number of informal workers, (3) the degree of informal superiority for accessing WEEE, (4) the efficiency of informal recovery operations, and (5) the profitability of the second-hand market. From the base case and the scenario analysis, this study further reveals the important role of the second-hand market in the model under consideration. The second-hand market may play as the leverage of the continuing growth if it has growing demand, or as the limit of the informal growth, when the demand is limited.

This study offers some practical insights for policy makers. Since numerous studies have pointed out the harmful nature of the informal recycling sector (Streicher-Porte et al., 2007), the ways to slow down or even to cease the informal sector are still preferred by policy makers. If only the pragmatic approaches are considered, then the findings provide insight on how to solve the informal sectors' problems: weakening the reinforcing loop and strengthening the balancing loop in the informal dynamics. Practically, it can be achieved through: (1) limiting the access of scavengers to the obsolete products by declaring scavenging as illegal, (2) strengthening the law enforcement to prevent the trans-boundaries movement of WEEE, (3) creating additional outflows from the informal workforces by restriction, (4) slowing the informal recovery processes by forcing standardization, and (5) intervening the second-hand market. Nevertheless, the promoted solutions should be relevant to the real conditions and not create additional problems. The solutions should be kept away from two extreme sides: on the one hand from cracking down the entire informal recycling sector without considering the side effects such as higher unemployment, and on the other hand, leaving this sector to run business-as-usual, thus, e.g. exposing the informal workers to the more acute health situation. Therefore, the solutions that conform to sustainability pillars may be encouraged. Firstly, the way to enhance the informal sector should be developed because it is conceptually better for the sustainability (Besiou et al., 2012) and practically achievable (GIZ, 2011). It may be accomplished through the integration of the informal sector into the formal one and building its capacity and environmental awareness in recovering the WEEE. Secondly, the policy makers should give concern to the input shortage faced by the formal sectors. They may give the formal sector higher access to WEEE, i.e. by collaborating with another governmental bodies to obtain easy access to obsolete products and collaborating with the retailers to increase collection rate. Lastly, the capacity building for the formal sector should be endorsed, concerning high losses of the valuable materials (Streicher-Porte et al., 2007). The limited capacity in the formal collection and recovery process contributes to this loss. Additionally, the results may indicate at some indirect aspects outside the boundaries of waste management systems that help creating the landscapes for the emergence of informal growth. These aspects include economic, education, agricultural, and urban planning. Hence, the policy makers should explore more holistic approaches to enhance the solutions, e.g. by building cross-sector collaborations with more relevant stakeholders.

This study acknowledges several limitations, which offer directions for the future research. To prioritize the informal sector, this study simplifies the SD structure of the formal sector, represented only by the formal collection. In reality, the formal sector has its own complexity and dynamics. It is interesting to know which results might appear if the SD model disaggregates the structure of the official systems into detail. This study also has a limitation with its assumption that the government does nothing with the emerged behaviors. Useful suggestion includes incorporating endogenous environmental policies (Georgiadis and Besiou, 2008), simultaneously in the SD model, which response to the behavior of the informal sector. The future studies may also develop a further SD model that assesses the formalization of the informal sector and its impacts on the society. Lastly, the results of this study are also subjected to the synthesized parameters, with its limitation. Hence, the issue of replicability of the model may rise. Therefore, additional empirical studies accompanied by data enhancement are necessary to give a deeper understanding of the realities in the informal sector.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2015.11.038.

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High-quality collection and disposal of WEEE: Environmental impacts and resultant issues

John Baxter *, Kari-Anne Lyng, Cecilia Askham, Ole Jørgen Hanssen

Ostfold Research, Stadion 4, 1671 Kråkerøy, Norway

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ABSTRACT

Life cycle assessment of the collection, transport and recycling of various types of waste electrical and electronic equipment (WEEE) in Norway shows that small amounts of critical materials (refrigerants, precious/trace metals) are vital for the overall environmental accounts of the value chains. High-quality recycling ensures that materials and energy are effectively recovered from WEEE. This recovery means that responsible waste handling confers net environmental benefits in terms of global warming potential (GWP), for all types of WEEE analysed. For refrigeration equipment, the potential reduction of GWP by high-quality recycling is so large as to be of national significance. For all waste types, the magnitude of the net benefit from recovering materials and energy exceeds the negative consequences of irresponsible disposal. One outcome of this may be widespread misunderstanding of the need for recycling. Furthermore, framing public communication on recycling in terms of avoiding negative consequences, as is essentially universal, may not convey an appropriate message. The issue is particularly important where the consumer regards products as relatively disposable and environmentally benign, and/or where the "null option" of retaining the product at end-of-life is especially prevalent. The paper highlights the implications of all these issues for policy-makers, waste collectors and recyclers, and consumers.

1. Introduction

The collection, recycling and treatment of waste electrical and electronic equipment (WEEE) has in recent years come into increasing focus as an important element of national and international waste and environmental management strategies. WEEE is recognised as a rapidly growing waste stream, in terms of its overall volume but also in terms of its environmental significance. Scandinavian territories have been at the forefront of developments, with high collection rates and well-developed systems for waste handling and treatment (see Ylä-Mella et al., 2014). Widespread collection and recycling of WEEE offers considerable environmental advantage compared to other disposal options. Two main factors are identified:

- WEEE contains many elements that result in direct environmental impacts if disposed of improperly – they contribute to global warming, and some are toxic/hazardous.
- Recycling of WEEE leads to the recovery of valuable metals, plastics and other components. This brings obvious economic advantages, but also environmental benefits where recovered materials obviate the need for production of virgin materials. Even where material recovery is not possible or practical, energy recovery as part of a well-managed incineration process recovers some of the environmental burden of treatment.

Both policy-makers and consumers generally focus much more on the first of these than the second. *Avoiding negative consequences* is arguably the bedrock of mainstream discourse on the environment. This is wholly unsurprising, not least since legislation and regulation of activity in the environmental perspective is framed almost entirely in terms of avoiding negative consequences. The two principal European directives relating to WEEE, namely the revised WEEE Directive (2012) and the RoHS or Restriction on the Use of Hazardous Substances in Electrical and Electronic Equipment Directive (2011), are both framed in this fashion. Furthermore, at the macro (global political) scale, avoiding negatives *is* the ostensible purpose of environmental activity – the Kyoto Protocol (1997) for limiting the negative effects of greenhouse gases being a widely recognised example. The paper shows how avoiding negatives (direct environmental impacts) is most important for some, but not all, WEEE product groups and end-of-life

* Corresponding author.
E-mail address: john@ostfoldforskning.no (J. Baxter).

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value chains. The policy and practical implications of this finding are explored. Both recyclers and consumers have significant roles to play, and hence both policy and public communication instruments are vital.

As in many other European countries, Norwegian WEEE collection and recycling is almost entirely driven by the concept of Extended Producer Responsibility (EPR) which requires those putting items on the market to be ultimately responsible for their end-of-life treatment (Sander et al., 2007). EPR is implemented via a number of governmentally approved companies for the take-back, treatment and processing of WEEE. These companies are membership organisations, funded by subscriptions from technology producers and importers. Elretur AS is one of the biggest such organisations in Norway. It is responsible for tens of thousands of tonnes per year of WEEE from a network of several thousand collection points nationwide. Norway is a large country, with a sparse, widely separated population outside the major cities. National WEEE regulations (Milkødirektoratet, 2013) include a responsibility on approved companies to collect from all parts of the country. This poses considerable challenges in terms of costs, logistical efficiency and consequent emissions. Optimising these parts of the value chain is a distinct field of research in itself, which is not considered in detail here.

Examination of the environmental burdens and benefits of WEEE recycling via the Life Cycle Assessment (LCA) approach has become fairly prevalent in the past decade or so. Studies have included overall examinations of WEEE recycling value chains in particular countries or regions, such as Japan (Menikpura et al., 2014), Switzerland (Hischier et al., 2005; Wäger et al., 2011) and Lombardia in Italy (Biganzoli et al., 2015). Others include examinations of particular product groups in detail, such as refrigerators (Xiao et al., 2015) and fluorescent lamps (Tan et al., 2015). Studies vary considerably in range and scope – the part of the value chain that is studied in detail, the range of environmental indicators considered, the level of detail in the description of treatment and recycling, the level of data in inventory data, and so on.

1.1. Waste management

Many studies focus on the WEEE value chain as a whole – incorporating raw materials extraction, manufacturing, transport, use and disposal. Studies focusing on end-of-life are relatively rare, moreover they often focus on waste management as it is intended to happen. For example, products are assumed to be subjected to high-quality recycling that yields a net environmental credit to the overall value chain via avoided materials or energy production (Xiao et al., 2015 provides a typical example). Such a perspective is captured in our "best practice" recycling scenarios described below.

However, the reality is that recycling may not proceed exactly as intended in best practice. Here, we provide a novel focus on waste management activities by presenting disposal scenarios that are realistic yet non-optimal. Crucially, we highlight the relative responsibility of different actors in the value chain, eventually showing that the primary focus should be on different actors for different product groups. Specifically, optimal treatment of mobile phones depends primarily on consumer action, whereas for the other product groups, the actions of recyclers and processors are likely to be most important for the overall environmental account of the value chain.

The work presented in this paper concerns environmentally responsible collection, treatment and disposal of WEEE, with specific reference to the activities of Elretur, whose responsibilities extend along the value chain. From Elretur's standpoint, the work serves the following purposes:

- Analysing the take-back and recycling value chain to identify, document and communicate the environmental burdens and benefits therein.
- Identifying those parts of the value chain which impact most on the environment.
- Highlighting activities which need particular attention to ensure good overall environmental stewardship of the waste.

2. Material and methods

The study uses Life Cycle Assessment (LCA) – see for example, EU-European Commission (2010) – a standardised approach to systematically assessing the life cycle environmental impacts of products. As per the relevant ISO standards 14040 and 14044, there are distinct phases in the assessment – goal and scope definition, inventory analysis and impact assessment, each coupled with an interpretation stage. LCA is applied specifically here to the waste treatment parts of the value chain, following relevant European Commission guidelines (Simone and Rana, 2011).

This study focuses on three specific types of electronic waste – refrigerators, LCD screens and mobile telephones, considered separately. The goal and scope of the study was to calculate selected life cycle environmental impacts of the specific products along the parts of the value chain highlighted in Fig. 1, with a view to addressing the issues listed above with respect to Elretur's operations and public activity/communication. The functional unit for the study was the waste treatment of one typically-sized device in each of the product groups. More precisely, it was treatment of a mass of waste equal in mass to a typical device in each group. These masses were taken from Elretur's own data as 51 kg for refrigerators, 20 kg for LCD-TVs and 140 g for mobile phones, with the latter consisting of a 115 g phone plus a 25 g battery.

The collection, distribution and processing system for Norwegian WEEE operated by Elretur is highly complicated – owing to geographic factors, coupled with quite different treatment pathways for different types of WEEE. The system is broadly hub-and-spoke in nature. Elretur's responsibility for the waste begins at the collection sites across the country. These include municipal waste sites, electronics dealerships and others. Earlier parts of the extended value chain relate directly to consumer behaviour and hence the factors that influence if, when and how end-of-life WEEE reaches the initial collection site. This represents a substantial research area in its own right, and is not considered here.

Waste is classified in one of six product groups at the collection points. It is collected, with greatly varying frequency depending on location, and shipped to one of about 12 regional reception centres. Some types of waste demand a pre-treatment step, for example the removal of batteries from mobile phones. In some other cases, depending on waste type and location, pre-treatment consists of an intermediate reception/holding step, where waste from a number of locations is collated before onward transport. Then, depending on waste type, it is transported to treatment or recycling locations – most in Scandinavia, with some elsewhere in Europe and further afield.

Fig. 1 shows the system boundaries of the study and highlights the parts of the extended value chain in specific focus here. There are three transport stages from collection to treatment. Very detailed information was available on transport – over 8000 lines of data captured the collection and onward transport arrangements from every site in the country. This was combined with GIS data for distances between collection/treatment locations to compute accurate national weighted-average transport distances for every waste type at every stage of the chain. Further data enabled the modes of transport (road, rail or sea) to be identified and hence accurate computations made of the emissions and environmental impacts arising from transport.

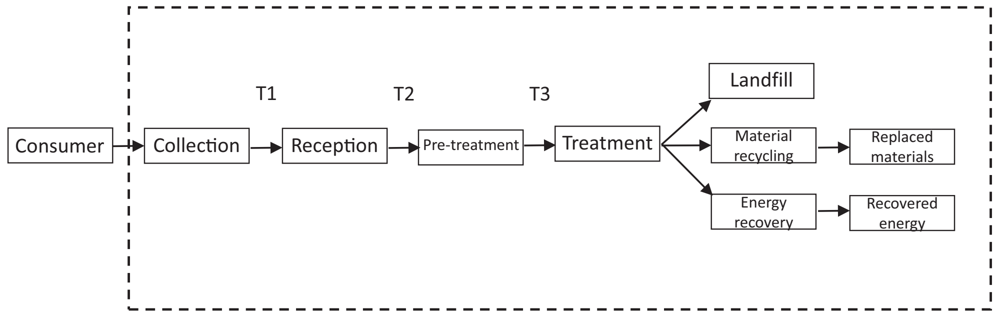

Fig. 1. Schematic of WEEE treatment value chain and system boundaries for the study.

Treatment consists of a combination of material recycling, energy recovery and landfill, depending on the waste type. Recovered materials and/or energy are assumed to replace equivalents on the European market. As a result, the value chain is credited with the environmental impact of producing such materials from existing sources – hence, for example, using an existing production mix for a specific material. However the actual delivery of recovered materials or energy back into the market lies outside the system boundary of this study. Furthermore, it is assumed that recovered materials and energy are readily sold back into existing markets. This is something of an idealisation, yet in practice the revenues from the sale of recovered resources provide an important financing stream for the collection and treatment operations. As such, the scenarios presented in the study are reasonably realistic. The treatment value chain itself gives rise to environmental impacts – for example CO2 emissions arising from waste transport, and the emissions and energy use in storing, treating and processing waste. This is combined with the derived benefit from recovered materials to give the net impact for the treatment activity.

The effect of the treatment and processing value chain in each case is assessed by considering two scenarios for each waste type (see Table 1). The two scenarios were based on extensive experience of waste treatment operations within Elretur. Specifically, efforts were made to ensure the less attractive scenario was as realistic as possible, rather than an idealised worst-case which would maximise the apparent effect of good product stewardship in collection and treatment. Implicitly there is also a third (null, do-nothing) scenario whereby the end consumer retains the product at end-of-life, which gives zero environmental impact for the treatment and disposal value chain. As discussed below, this is both more realistic and provides more useful insight for some waste types than for others.

The recycling scenario attempts to capture what happens to waste if it enters the official take-back and recycling value chain, and the elements of the recycling process occur essentially as intended. It represents a feasible best-practice scenario for WEEE collection and treatment. The waste scenario attempts to realistically capture sub-optimal collection, treatment and recycling of waste. It is not an absolute worst-case scenario for all waste types, particularly refrigerators.

Models of the six value chains (three waste types with two scenarios each) were constructed using the SimaPro LCA software (v7) and the ecoinvent databases (v2). A range of environmental impact indicators were calculated, but only global warming potentials (GWP) are presented in detail in this paper, calculated using the IPCC 2007 GWP 100a method version 1.02.

The composition of refrigerator waste was taken to be as shown in Table 2 (see Elretur, 2012), with the fate of the various fractions depending on the scenario. The calculations were based on an average mass of refrigerator being 51 kg.

Table 1
WEEE types and treatment pathways in the modelled scenarios.

Waste type	Recycling scenario	Waste scenario
Refrigerators	Waste is pre-shredded and separated at one of two treatment sites in Norway, with metals sent to material recycling and plastics either to material recycling or energy recovery. All metals/plastics are recovered, subject to appropriate reject/recovery rates for the various treatment processes. The different recyclable fractions (steel, aluminium, copper, plastics) are sent to various locations for reprocessing and assumed to replace the equivalent materials on the European market. All refrigerants are captured and treated	The recyclable materials are recovered and treated as in the recycling scenario. The refrigerants are not captured and instead are allowed to leak into the atmosphere. This scenario – to varying degrees – reflects one of two plausible outcomes. Waste may be diverted outside the official take-back scheme, perhaps by sale to scrap metal dealers or as the result of theft (see Baxter et al., 2015). Otherwise, waste may remain within official channels but become damaged in the handling or transport stages. In either case, the recyclables may well eventually be recovered but the refrigerants probably not
Mobile phones	The waste is pre-sorted at three reception sites, where the batteries are removed and sent for separate treatment. The phone waste is sent to a smelting plant in Sweden where the precious/trace metals (in particular) are recovered. A small fraction of the waste is landfilled	It is assumed that consumers do not recycle mobile phones, instead placing them in normal domestic waste, which is treated in municipal waste incineration. However, as discussed below, for this product group the null option/scenario is probably more significant than the waste scenario
LCD TVs	All of this waste is collated at a single reception point then shipped to a specialist treatment plant in Germany. It is shredded, the hazardous waste fractions (including fluorescent lamps) are securely removed in a controlled environment, and the recyclable fractions (metals, plastics some glass) are recovered and sent for recycling elsewhere in Germany. Other fractions are thermally treated or landfilled	The waste products are disposed of in a specialist industrial waste incineration process in Sweden. There is no attempt at energy recovery. The scenario reflects a typical disposal route prior to the specialist treatment route in the recycling scenario becoming available

Standard LCA processes were used for the recycling/landfill/incineration routes for the various fractions; where a specific process was not available (such as landfill of residual waste) the fraction was assumed to be inert waste. The report from which the data were derived (Elretur, 2012) gives an average of around

Table 2
Average composition of refrigerator waste and fate of different fractions.

Waste fraction	Mass (kg)	Mass (%)	Fate
Steel	36.06	70.7	Recycling or landfill
Other plastics	5.92	11.6	Recycling or landfill
Polyurethane	6.12	12.0	Incineration (energy recovery) or landfill
Residual waste	1.84	3.6	Landfill
Refrigerant (taken as R134a)	0.41	0.8	Thermal destruction or emission to atmosphere
Aluminium	0.36	0.7	Recycling or landfill
Copper	0.31	0.6	Recycling or landfill

400–450 g of refrigerant in an average refrigerator, split between the cooling circuit and the insulation. For simplicity, this refrigerant was assumed to be R134a and was taken to be emitted to atmosphere either when the cooling circuit contents were not specifically isolated and recovered, or when the insulation was not specifically treated. It was noted that end-of-life refrigerators of different ages may well contain different refrigerants (more or less environmentally damaging substances such as R12 and R600a respectively) but these factors were not taken into account.

Table 3 shows the equivalent data for LCD-TVs, with the weight of a single device taken as 20 kg.

The waste scenario represents a simplification for LCA of a complex state-of-the-art treatment and disposal process, particularly focused on secure separation and elimination of hazardous waste components such as the liquid crystals themselves (in the "other waste" fraction).

For mobile phones, the composition and treatment is as shown in Table 4, with the phone mass taken as 115 g:

In addition, the phone is assumed to contain a 25 g Li-ion battery which is assumed in the recycling scenario to be treated using a market-average mix of processes as given in the ecoinvent database. In the waste scenario it is subject to municipal incineration as is the phone itself. The "other metals" fraction is a mixture of lead, tin, zinc and precious/rare metals. Of the latter, gold and silver are modelled explicitly; literature data on the other trace metals was too scarce and uncertain to be of use in the study. The analysis below, and the principal conclusions drawn from it, acknowledges the uncertainty in this data.

Transport distances were calculated on a mass-averaged basis as described above from the detailed logistical and GIS data: (see Table 5).

There was no T3 stage for refrigerators since they are shipped directly from the reception point to the treatment location. The rather high T3 values for the other product groups reflect that the final treatments take place outside Norway (in Germany and northern Sweden). Most of the transport in all three stages was modelled by road transport, with a small proportion of the overall waste known to be transported by rail. Also note that these transport distances were those for the integral waste products – for example, essentially whole refrigerators. Where products are disassembled or shredded before onward transport for further recycling or processing, such transport was included within the treatment life-cycle stage. In some cases such as material recycling in China, this transport can be quite considerable (20,000 km by sea).

Table 3
Average composition of LCD waste and fate of different fractions.

Waste fraction	Mass (kg)	Mass (%)	Fate
Steel	6.0	30	Recycling/landfill/incineration
Aluminium	1.4	7	Recycling/landfill/incineration
Copper	0.3	1.5	Recycling/landfill/incineration
Plastics	7.7	38.5	Recycling/landfill/incineration
Residual waste	2.7	13.5	Landfill or incineration
Glass	1.0	5	Incineration
Other waste	0.9	4.5	Incineration

Table 4
Average composition of mobile phone waste and fate of different fractions.

Waste fraction	Mass (g)	Mass (%)	Fate
Plastic	68	59	Recycling/incineration
Ceramics	18	16	Recycling/incineration
Copper	17	15	Recycling/incineration
Steel	3	3	Recycling/incineration
Aluminium	2	2	Recycling/incineration
Other metals	7	5	Recycling/incineration

Table 5
National mass-average transport distances for WEEE product groups.

Waste fraction	T1 (km)	T2 (km)	T3 (km)
Refrigerators	113	588	0
LCD-TVs	116	380	959
Mobile Phones	116	155	879

Transport was generally modelled using standard modules in the ecoinvent database: a heavy lorry meeting the EURO 5 standard for road, European average rail freight, and a transoceanic freight ship where applicable. The LCA models for treatment and recycling were mostly constructed from standard ecoinvent modules which are adapted as appropriate, for example to reflect specific intermediate transport steps or local electricity mixes. In some cases, notably the treatment by smelting of mobile phones, the LCA data – on emissions and energy use – is derived directly from a mix of literature and process-specific data.

3. Results

The LCA results are first presented per functional unit (a single device at end-of-life as described above) for the different scenarios. Then the material flows (i.e., the number of devices encountering the various scenarios) are estimated. These findings are combined to give total environmental impacts and differences therein resulting from the different treatment paths, and there is also a brief summary of results for indicators other than global warming potential.

3.1. Impact assessment for individual devices

Global warming potentials are shown in Fig. 2 for each waste type and for each scenario. Net potentials for the value chains, in kilograms of CO2 equivalents per functional unit, are given by the numbers shown. The GWPs clearly vary by orders of magnitude depending on the waste type; this is perhaps unsurprising given the quite different masses and types of waste per functional unit in the different cases. Since in practice different product groups are essentially managed separately, it is sensible to maintain the study in effectively three different parts.

For all three product groups, there is an overall GWP advantage associated with the recycling scenario; the net potentials are negative and hence the value chain is better than the null, do-nothing option. This means that the benefit derived from the avoided production of materials and the avoided generation of energy exceeds the environmental cost of the treatment value chain. Of course, in practice the drivers for the value chain are economic, political and/or regulatory as well as environmental. Nonetheless it is noteworthy that the treatment value chains deliver net environmental benefits.

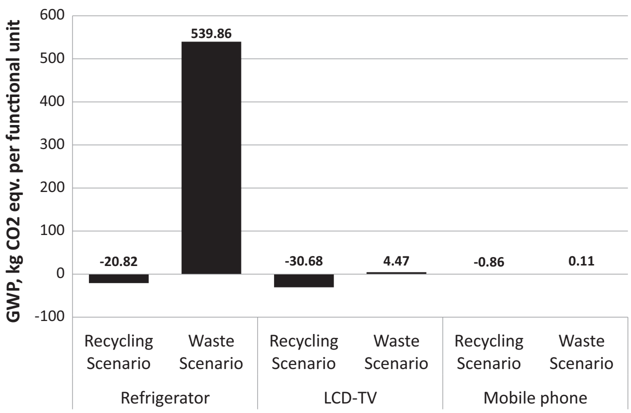

Fig. 2. Global warming potential for waste treatment scenarios.

Next we assessed the treatment value chains in more detail, illustrating the environmental benefits or burdens of individual stages. Fig. 3 shows the breakdown for refrigerators. Life cycle stages that give rise to environmental burdens (positive GWP) are stacked above the origin on the vertical axis, with those giving benefits (negative GWP) below it. It is apparent that some stages are much more significant from others. Most of the environmental burdens arise from the various aspects of treatment, and most of the benefits accrue from recovered materials. Pre-treatment, transport and recovered energy seem relatively insignificant. A more detailed breakdown for each of the scenarios is shown in Tables 6 and 7.

The treatment burdens mostly arise from steel recycling and the emissions associated with energy recovery from polyurethane foam. The "other treatment" burden consists of numerous small contributions from other parts of the treatment process. Benefits arise mostly from avoided production of steel, with a significant contribution from avoided plastics. We can see that the scenario as a whole is environmentally beneficial; the environmental benefit of recovering materials and energy exceeds the environmental cost of treatment.

The waste scenario carries a high environmental burden as a whole, and this is dominated by a single process. The emissions of refrigerant gases, modelled as HFCs, results in a very large GWP – just over 400 g of gas results in nearly 600 kg of CO2 equivalent. This accounts for 96% of the total GWP burden, which in turn overwhelms the benefit from recovered materials and energy. The result shows that proper handling of refrigerator waste to prevent leakage of refrigerants, and effective recovery/destruction of refrigerants in treatment, is by far the most important factor for the

Table 6
Detailed breakdown of GWP (kg CO2 eq) for refrigerators, recycling scenario.

Burdens	kg CO2 eq	Benefits	kg CO2 eq
Steel recycling	20.6	Recovered steel (28.8 kg)	−48.0
Energy recovery (PUR)	12.3	Recovered plastics (4.73 kg)	−16.1
Other treatment	14.4	Other recovered materials	−3.4
Transport	3.0	Recovered energy	−3.6
Total	50.3	Total	−71.1

Net burden = −20.8 kg CO2 eq.

Table 7
Detailed breakdown of GWP (kg CO2 eq) for refrigerators, waste scenario.

Burdens	kg CO2 eq	Benefits	kg CO2 eq
Emission of refrigerant (0.4 kg)	583.0	Recovered steel (28.8 kg)	−48.0
Recycling of steel	10.3	Recovered plastics (4.73 kg)	−16.1
Other treatment	14.7	Other recovered materials	−3.4
Transport	3.0	Recovered energy	−3.6
Total	611.0	Total	−71.1

Net burden = 539.9 kg CO2 eq.

GWP of the whole value chain. In the recycling scenario, over 30 kg of steel and plastics is recovered. The entire GWP benefit of this material recovery is negated if only 50 grams of refrigerant is allowed to leak to atmosphere.

The value chain breakdown for GWP in LCD-TV treatment is shown in Fig. 4. Unlike for refrigerators, some of the transport steps have a small but tangible effect. Replaced energy carriers are marginally important in the waste scenario, but replaced materials in the recycling scenario are most significant overall. In the recycling scenario, most of the benefit arises from replaced plastics (mostly acrylonitrile butadiene styrene – ABS) with some steel, copper and aluminium recovered as well. Treatment and transport in the recycling scenario is a little more burdensome overall than in the waste scenario – unsurprisingly since the waste is transported further and then treated more carefully. Nonetheless the enhanced benefit of material recovery more than compensates.

For mobile phones, the overall GWP breakdown is shown in Fig. 5. This shows small, but tangible, contributions from transport and pre-treatment – the latter referring to removal, separate transport and treatment of the batteries. Once again, however, replaced materials in the recycling scenario are the most important

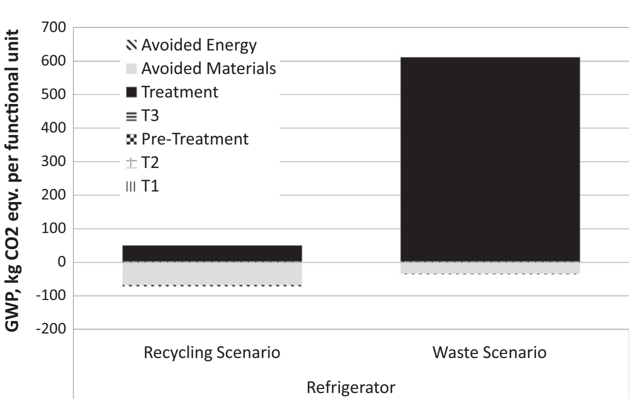

Fig. 3. GWP benefits and burdens of different value chain stages for refrigerators.

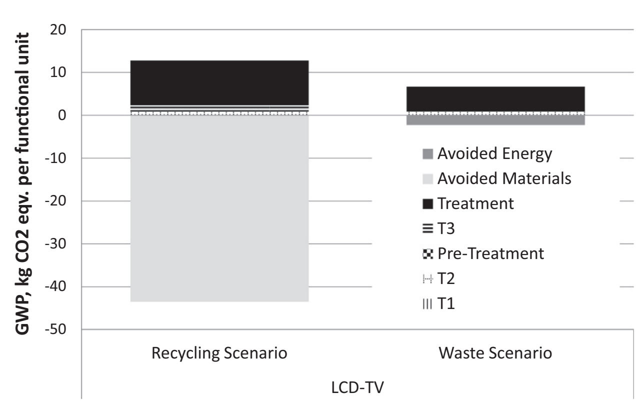

Fig. 4. GWP benefits and burdens of different value chain stages for LCD-TVs.

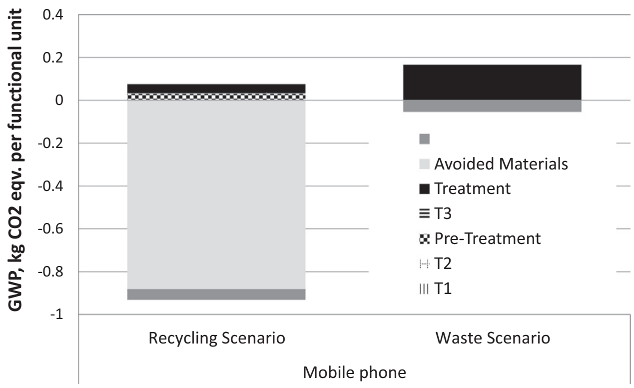

Fig. 5. GWP benefits and burdens of different value chain stages for mobile phones.

contributor, dominating both the total benefit and the overall impact of the value chain as a whole. The waste scenario shows a relatively large burden, with a little GWP retrieved through energy recovery for the plastics. The breakdown for the recycling scenario is shown in Table 8.

This shows that the scenario is dominated by replaced materials, specifically gold and to some degree silver, even though they are only present at trace levels. Precious metals are expensive to mine in environmental as well as in economic terms, because the industry is extremely energy and resource-intensive. As mentioned above, determining the precise composition of mobile phone waste with respect to trace and precious materials is a difficult exercise. Chancerel and Rotter (2009), Chancerel et al. (2015) describe in detail the difficulty and uncertainty in characterising WEEE in terms of precious/trace metal content. The key point is that trace materials present in the milligram range can have very significant environmental effects, and that recovery of such materials is paramount in developing a good overall environmental account for the value chain.

We also know that traces of rare earth metals are present in this waste. The degree to which they are recovered at all is much less clear than for gold and silver. Analysis indicates that worldwide recycling of precious metals is considerable but that of lanthanides is almost nil (Reck and Graedel, 2012). Specific LCA data on such materials is relatively scant. However, recent analysis (Bakas et al., 2016) based on a comprehensive, wide-ranging LCA study of metals production (Nuss and Eckelman, 2014) and on other assessments (Vats and Singh, 2015) suggests that – at least in the perspective of GWP and other common indicators – rare earths may be of secondary importance to precious metals. The analysis reinforces our conclusions here, that the overall environmental picture is very sensitive to precious metals even in very small quantities. However, based on what we know about rare earths and their production in general, they would likely be significantly more important were other indicators (such as those relating to resource use, toxicity and occupational health) included in the analysis.

Table 8
Detailed breakdown of GWP (kg CO2 eq) for mobile phones, recycling scenario.

Burdens	kg CO2 eq	Benefits	kg CO2 eq
Treatment (smelting of scrap)	0.035	Recovery of gold (0.05 g)	−0.689
Pre-treatment (removal and treatment of battery)	0.028	Recovery of silver (1.1 g)	−0.104
Transport	0.012	Other recovered materials	−0.089
		Recovered energy	−0.049
Total	0.075	Total	−0.931

Net burden = −0.856 kg CO2 eq.

3.2. Likely disposal pathways of different devices and overall impacts/savings

In this section we provide estimates for the numbers of devices subject to each of our scenario pathways, to give an annual national assessment of the environmental impacts and savings of WEEE collection and treatment.

3.2.1. Refrigerators

In 2013, according to the Norwegian national register (EE-registretet, 2015) there were 16,147 tonnes of *kuldemøbler* (refrigerators and freezers) collected by official take-back companies. Assuming our model product here to be representative of that product group, this corresponds to over 300,000 refrigerator units. With responsible recycling yielding a net benefit of almost 600 kg of CO2 per unit, this means that nearly 200,000 tonnes of CO2 equivalent is saved if all refrigerators are responsibly treated compared to if none of them are. Norway's total CO2 emissions in 2013 were 53.9 million tonnes of CO2 equivalent (Statistics Norway, 2015) and hence refrigerator treatment represents a saving of up to 0.4% of national global warming impact.

Naturally, the figure of 200,000 tonnes of CO2 equivalent is an upper limit for the environmental savings. In practice, only a fraction of devices are responsibly treated. As already mentioned, two factors interfere with responsible treatment – diversion of waste outside the official pathway, and mishandling of waste within the official pathway. An estimate for the former can be derived from analysis within the EU CWIT (Countering WEEE Illegal Trade) project. This shows that Norway performs relatively well by European standards with almost 80% of WEEE (all categories) being reported and documented within the official pathway (Huisman et al., 2015). About 15% of all WEEE is subject to "non-compliant recycling" outside official pathways, and this is "... mainly steel dominated consumer appliances" – hence one would expect refrigerators to feature heavily in this group. From this, we suggest that a noticeable fraction – perhaps 20% or more – of refrigeration equipment follows this path. For this path it seems reasonable to assume that all the gases in the cooling circuit and the refrigerants in the oil/insulation are lost.

Mishandling of waste within official channels is harder to assess. Most historical analyses focus on operations at refrigerator recycling plants (for example RAL, 2002; Dehoust and Schüler, 2010). A specific Norwegian audit of the broader collection and treatment system (RAL, 2002) highlighted shortcomings in the collection and transport steps, including the knocking of equipment, laying of equipment on its cooling coils, and haphazard storage at intermediate sites. More recent field observations (for example Karlsen and Hannestad, 2015) indicate that such problems are still evident to some degree, and that the pressure on transporters to pack waste in vehicles as densely as possible remains an important factor. We suppose that refrigerants in cooling circuits are particularly vulnerable here.

Taking these factors together, we estimate that perhaps 25% of all refrigerants in the value chain as a whole are not recovered satisfactorily. We therefore conclude from the above total figures that responsible refrigerator recycling saves perhaps 150,000 tonnes of CO2 equivalent per annum, with further improvements of 50,000 tonnes being possible in principle.

3.2.2. Mobile phones

There is little reliable data on the number of mobile phone handsets actually collected and recycled; unlike for refrigerators, official collection data aggregates mobile phones with many other types of products. Elretur (private communication) suggest that of the order of 150,000 handsets per annum are collected in Norway against annual sales in excess of 2 million. Other analyses (Three out of four Norwegians, 2015) indicate that perhaps 10 million end-of-life handsets remain in circulation.

Our analysis suggests that approximately 1 kg of CO2 equivalent is saved by responsible recycling of a single handset. Hence, the current net effect of recycling is relatively modest (150 tonnes of CO2 equivalent or so). However, recovering the handsets currently in circulation could yield net savings of perhaps 10,000 tonnes, with an ongoing saving of perhaps 2000 tonnes per annum by fostering recycling and hence limiting further accumulation of mobile devices within the system.

3.2.3. LCD TVs

Direct data on LCDs is also scarce. Recent analyses (Bakas et al., 2016) based on (somewhat reliable) historical sales figures and (somewhat uncertain) assumed device lifetimes indicates that Norwegian annual waste generation for LCDs is of the order of 10,000 tonnes per annum (hence, based on our assumption, 500,000 units). Assuming, as per our analysis, a net difference of 35 kg CO2 equivalent between the two disposal scenarios, this gives an upper bound for the net saving via responsible recycling as around 17,500 tonnes CO2 equivalent per annum. We have no clear estimate for the fraction of LCDs that are currently responsibly recycled, but we can suppose that both the current annual saving and the potential for further improvement are each of the order of several thousand tonnes of CO2. Analyses of TV product use and lifetimes, conducted for Sweden but probably broadly applicable to Norway as well, suggest that WEEE generation in this area over the next five years will continue to rise (Kalmykova et al., 2015).

3.3. Results for different environmental indicators

Global Warming Potential is by far the most widely considered environmental indicator and many LCA studies focus closely upon it, as we have done here. Nonetheless, environmental impacts were calculated for a number of other indicators, and the broad overall findings are briefly summarised here. The indicators presented are the four other most prevalent in LCA analyses (ozone depletion, photochemical oxidation, eutrophication and acidification potentials).

- For refrigerators, the scenarios for other indicators re-emphasise the importance of refrigerants for GWP and highlight that refrigerants carry limited environmental burdens for other indicators. The waste scenarios for other indicators resemble the recycling scenario for GWP; there is a small net environmental benefit of the value chain, and steel recycling burdens and benefits are most significant. The recycling and waste scenarios give very similar results for the other indicators.
- For mobile phones, the scenarios for other indicators follow the same broad trends as for GWP. For all indicators, the recycling scenario is substantially better than the waste scenario, and this can be attributed to the recovery of trace materials (particularly gold and silver). In turn this can be attributed to the high environmental costs – across all categories – of mining these rare/precious materials.
- For LCDs, once again the scenarios follow similar overall trends for other indicators as they do for GWP. The recycling scenario carries a significant net environmental benefit and is much better than the waste scenario across all indicators. Disposal of the liquid crystals themselves carries a substantial burden but – particularly in the recycling scenario – the recovery of metals and plastics more than compensates.

Naturally, ever more comprehensive analyses involving yet further environmental indicators (such as those relating to toxicity, resource or land use) are possible in principle, as is the weighting of different indicators. Nonetheless, on the basis of the evidence from the indicators shown here, both the basic trends across different pathways and those for specific indicators seem reasonably consistent. Recycling pathways deliver considerable advantages over waste pathways, usually related to the environmental cost of virgin material production.

4. Discussion

Life cycle assessments relating to electronic equipment are commonplace in the literature. However, there are so many variables that few, if any, studies permit more or less direct comparison with the present work. Studies may be based around an entire life cycle of the product. Focus on specific life-cycle stages much more commonly encompasses product manufacture – for example assessing the effect of different materials and technologies – and/or the use phase – for example assessing the effect of energy efficiency. Most studies show that in a total life cycle perspective, the use phase dominates the total emissions for refrigerators (for example Xiao et al., 2015) and the production and use phases dominate for LCDs and mobile phones (Andrae and Andersen, 2010). However, these studies are predicated on optimal recycling which typically yields a small net environmental credit. Particularly for refrigerators, the present study shows that sub-optimal recycling can mean a substantial environmental burden from end-of-life, meaning that use phase no longer totally dominates entire life-cycle impact.

The treatment and disposal phase is rarely, if ever, considered in isolation as we do here. Even where there are very detailed studies of particular products that focus on disposal – for example, the study of refrigerators by Dehoust and Schüler (2007) – the overall scope, assumptions, impact assessment approaches and other factors are often quite different and direct comparison is difficult. A few studies do yield some broadly comparable data on the impacts of waste treatment. The study of refrigerators by Johnson (2004) suggests life cycle global warming potential of around 500–600 kg CO2 equivalent that is attributable to HFC refrigerants. A study of mobile phones by Andrae and Vaija (2014) suggests an impact for "end-of-life treatment" of older mobile phone models of 0.1–0.2 kg CO2 equivalent. Bhakar et al. (2015) give an end-of-life impact of about 7 kg CO2 equivalent for LCDs. All of these examples seem broadly in line with our findings.

Here, we are solely concerned with the end-of-life phase, and specifically the net environmental benefit of "responsible" waste treatment of various products. One way of quantifying the benefit is as the simple difference between the two waste scenarios, as shown in Table 9.

The findings have wide-ranging implications: for policy-makers, for those responsible for waste collection and recycling, and for end-consumers. High-quality electronics waste recycling is a Norwegian national imperative of some significance, which should concern policy makers, recyclers and the public.

Table 9
Overall effect of treatment value chain on GWP, per device.

Waste fraction	GWP, kilograms of CO2 equivalent per functional unit		
	Impact of waste scenario	Impact of recycling scenario	Net effect of value chain
Refrigerators	539.90	−20.82	560.72
LCD-TVs	4.47	−30.67	35.14
Mobile phones	0.11	−0.86	0.97

For refrigerators, the principal function of the treatment value chain is to avoid the downside of the conventional waste scenario (575 kg of CO2) – the net benefit of the recycling scenario (20 kg of CO2) is small by comparison. The other two product groups provide a complete contrast – most of the overall advantage derives from the benefits in the recycling scenario, rather than avoiding the burdens of the waste scenario. It is useful to illustrate the relative burdens and benefits of responsible and irresponsible waste treatment in a normalised form for the different product groups, as shown in Fig. 6.

As noted above, avoiding direct impacts is the primary basis both for most public communication and for most policy drivers with respect to WEEE. Yet these results show that, for two of the three product groups, the environmental benefits of WEEE recycling are not solely about – or even *mostly* about – avoiding direct impacts. This mismatch between perception (or message) and reality matters in two specific ways for consumer decision-making:

- The moderately well-informed domestic or commercial consumer may well have some broad awareness of direct impacts, and strive to dispose responsibly of waste they consider to be harmful. However, they may not take such care with waste they consider to be relatively benign.
- The consumer may, implicitly or otherwise, consider the "null" option (that of doing nothing) to be acceptable, environmentally neutral or at least not particularly harmful. Our results show that the null option would be relatively neutral for the refrigerator (although in practice consumers are unlikely to retain large WEEE items indefinitely), but for the other two products the null option is almost as harmful as the irresponsible disposal route.

Studies show both of these factors to be a considerable concern in WEEE and consumer behaviour.

- The "benign WEEE" issue is reflected in the analysis of Darby and Obara (2005) who found an increasing prevalence in smaller WEEE items of essentially 'disposable' products containing electronics. These are not produced with recycling or reuse in mind, and it seems certain that consumer consideration of them as electronics items is loose at best. They include elements of clothing, toys and novelty items such as pens, badges and cards. Halvorsen (2012) examined consumer motivation for household recycling and the effect of policy instruments, concluding that the most important motivating factor was that consumers were convinced that recycling makes a tangible environmental difference. It follows that consumers are relatively unmotivated where they believe the waste to be relatively harmless.

Fig. 6. Burdens and benefits (GWP) of WEEE treatment options for different waste types.

- The "null option" issue (i.e. waste stockpiling) is known to be a particular issue for mobile phones. Ongondo and Williams (2011) estimated that end-of-life retention of mobiles in the UK could be as high as 60%. Norwegian studies (Three out of four Norwegians, 2015) suggest an average of around two mobile handsets per head of population in circulation but not in use. Stockpiling is an obviously easy option for the relatively well-informed consumer who would hesitate to put electronics in municipal waste, yet may not be sufficiently informed or motivated to recycle the item, and may also be concerned with data security.

In summary, whilst it may be reasonable to accept that consumers mostly dispose of large and/or obviously electronic items (particularly refrigerators and LCD-TVs in this study) in an environmentally responsible fashion at end-of-life, small electronic waste presents considerable risks. Part of the reason for such risks is that the balance between burdens and benefits of responsible and irresponsible waste treatment is not widely understood. Melissen (2006) highlights the particular issues for small electronics, focusing on the collection rather than the treatment angle for these types of products. More specifically with respect to mobile phones, literature reviews show that research has focused mostly on the technical aspects of material recovery in recycling, despite all the evidence showing that the collection part of the chain is vital (Sarath et al., 2015).

5. Conclusions

Life cycle assessment of the WEEE treatment value chains for three specific product groups in Norway has revealed a number of specific pointers for those involved in WEEE take-back, collection, transport and treatment.

Firstly, whilst transport aspects are normally very significant in terms of cost, from the strict environmental impact standpoint their effects are relatively insignificant. This is in spite of the country's geography, with collecting stations being relatively dispersed and hence transport distances being long (hundreds of kilometres on a mass-averaged basis). Transport amounts for 10% or less of the total environmental burden of the value chain.

The overall environmental impact of WEEE collection and treatment may be highly sensitive to a single issue. For refrigerators, failure to recover just a few grams of refrigerant negates the entire GWP benefit of recovering kilograms of metals and plastics. Impacts relating to mobile phones are highly sensitive to recovery of tiny amounts of precious metals. The key step in the value chain is not always obvious nor taken into account by policy or common practice. Policy instruments driving electronic waste collection and recycling, such as the EU WEEE Directive (2012) and the Norwegian national waste regulations (Milkødirektoratet, 2013) generally focus on quantitative rather than qualitative issues. Where quality is introduced at all, it is in the form of prohibiting or limiting certain (dangerous or hazardous) materials in the value chain. Such regulations and drivers can, at least in principle, be monitored and policed via sampling and measurement.

However, our results show that many quality aspects of WEEE recycling cannot easily be driven and regulated by policies like these. Refrigerator recycling is particularly vulnerable to careless or irresponsible practice, but it results in the offending material (the refrigerant) leaving, rather than remaining in, the value chain. Policing is clearly problematic in practice; field observations suggest that, even within official take-back schemes, elements of waste handling practice are less than optimal. Some actors responsible for waste collection and transport are largely ignorant of, or oblivious to, the environmental impacts of careless handling of WEEE. Their incentives and targets are essentially related to the mass of metals and plastics recovered, irrespective of the fact – for refrigerators in particular – that the environmental impact of collection is nil or worse if the waste is not handled appropriately. We conclude that policy drivers to incentivise good practice as well as providing clear disincentives to poor practice should be strongly considered. Refrigerator treatment is probably most important overall, and it may be that specific accounting for refrigerant materials may be required in addition to a blanket prohibition on their uncontrolled release.

Mobile phone recycling is sensitive to precious metals recovery. In principle this should be driven by economic factors, but these are clearly insufficient at present given that recycling rates remain so low. The responsible recycling pathway is primarily in competition with the null option, and hence enhancements to mobile recycling will largely arise from consumers rather than from recyclers. Many consumers may be encouraged to recycle more if they become more aware of the implications beyond the simple direct impacts of "irresponsible" disposal, specifically that the null option is almost as harmful as irresponsible disposal (Baxter and Gram-Hanssen, 2016).

GWP alone does not represent a comprehensive environmental footprint of the treatment value chains for the product groups under scrutiny here – in general a wide range of indicators should be examined, and possibly weighted. Nonetheless, our analysis suggests that trends generally hold across the range of indicators, with the notable exception of the huge GWP burden from refrigerant materials.

The net impact of a WEEE treatment value chain – and hence the net benefits of effective take-back, logistically efficient transport, careful treatment and so on – often derives from environmental savings in avoided materials or energy production as much, if not more, than from the directly harmful impact of irresponsible disposal. Whilst the modelling of avoided materials and energy is somewhat simplified here (for example, product quality is not really taken into account and it is implicitly assumed that the relevant markets can, and do, absorb the recycled materials), the conclusions are clear. Recovering trace components such as precious and rare earth metals – even in microscopic quantities – brings a considerable environmental, as well as financial, premium. Particularly for rare earths, virgin material extraction is technically difficult. It is also known to carry considerable – if not presently well-quantified – impacts both on the environment and on occupational health.

The notion of avoided production, and the positive environmental consequences thereof, could form a part of more sophisticated public communication regarding waste disposal and recycling than is currently typical. Communication and understanding on recycling issues is largely focused upon avoiding or limiting negative consequences rather than fostering positive ones. In some cases such perception is probably damaging to overall behavioural patterns. An increased focus on the positives of "responsible" behaviour could also conceivably influence the practice of recyclers and producers, leading to fuller implementation of the principles of Extended Producer Responsibility. These findings could extend well beyond WEEE to all forms of waste.

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Measuring treatment costs of typical waste electrical and electronic equipment: A pre-research for Chinese policy making

Jinhui Li [a,c,*], Qingyin Dong [b,c], Lili Liu [b,c], Qingbin Song [a,c]

[a] School of Environment, Tsinghua University, Beijing 100084, China
[b] Basel Convention Regional Centre for Asia and the Pacific, Beijing 100084, China
[c] State Key Joint Laboratory of Environment Simulation and Pollution Control, School of Environment, Tsinghua University, Beijing 100084, China

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ABSTRACT

Waste Electrical and Electronic Equipment (WEEE) volume is increasing, worldwide. In 2011, the Chinese government issued new regulations on WEEE recycling and disposal, establishing a WEEE treatment subsidy funded by a levy on producers of electrical and electronic equipment. In order to evaluate WEEE recycling treatment costs and revenue possibilities under the new regulations, and to propose suggestions for cost-effective WEEE management, a comprehensive revenue-expenditure model (REM), were established for this study, including 7 types of costs, 4 types of fees, and one type of revenue. Since TV sets dominated the volume of WEEE treated from 2013 to 2014, with a contribution rate of 87.3%, TV sets were taken as a representative case. Results showed that the treatment cost varied from 46.4 RMB/unit to 82.5 RMB/unit, with a treatment quantity of 130,000 units to 1,200,000 units per year in China. Collection cost accounted for the largest portion (about 70.0%), while taxes and fees (about 11.0 %) and labor cost (about 7.0 %) contributed less. The average costs for disposal, sales, and taxes had no influence on treatment quantity (TQ). TQ might have an adverse effect on average labor and management costs; while average collection and purchase fees, and financing costs, would vary with purchase price, and the average sales fees and taxes would vary with the sales of dismantled materials and other recycled products. Recycling enterprises could reduce their costs by setting up online and offline collection platforms, cooperating with individual collectors, creating door-to-door collection channels, improving production efficiency and reducing administrative expenditures. The government could provide economic incentives—such as subsidies, low-cost loans, tax cuts and credits—and could also raise public awareness of waste management and environmental protection, in order to capture some of the WEEE currently discarded into the general waste stream. Foreign companies with advanced WEEE utilization technology could invest or participate in this area, producing profits for themselves while helping to develop and implement environmentally friendly and energy-saving technologies applicable to the Chinese market.

1. Introduction

Waste Electrical and Electronic Equipment (WEEE) is one of the fastest-growing sustainability problems throughout the world, as the variety of toxic substances contained therein can contaminate the environment and threaten human health, if disposal protocols are not meticulously managed (Kiddee et al., 2013). Intensive uncontrolled processing of WEEE in China has resulted in the release of large amounts of heavy metals into the local environment, and created high concentrations of metals in the surrounding air, dust, soil, sediment and plants (Song and Li, 2014). As a result of the increased consumption and continual turnover of EEE, not only nationally but worldwide, China is now facing serious WEEE problems from both growing domestic generation and foreign imports. Because of the environmental and social concerns surrounding WEEE recycling, the Chinese government has established domestic collection and recycling systems in order to promote environmentally sound WEEE treatment (Li et al., 2011). Implemented on 1 January 2011, the Chinese Management Regulation for WEEE Recycling and Disposal can be regarded as the counterpart of the EU's WEEE Directive (2012/19/EU), and is a pivotal piece of national WEEE management legislation in China. To implement the regulation, the Chinese government established a "specialized fund" to subsidize the formal collection and recycling activities of WEEE in China. Since September 2010, 5 types of home appliances (TV sets, refrigerators, washing machines, air

* Corresponding author at: Sino-Italian Environment and Energy Building, Tsinghua University, Haidian District, Beijing 100084, China.

E-mail address: jinhui@tsinghua.edu.cn (J. Li).

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conditioners and personal computers) have been regulated, and producers must pay fees for their products, to the WEEE treatment fund. Based on their volume of processed WEEE, WEEE recycling enterprises can then apply for a subsidy, to ensure the safe and responsible processing of WEEE. Tang and Wan (2014) found that 5 types of WEEE, with a total amount of 42.35 million units, were treated in an environmentally sound way by permitted enterprises in 2013. Beginning in March 2016, another 9 types of WEEE will be included in the subsidy lists, according to the "WEEE Treatment Catalogue (2014)." The exact subsidy amounts for these 9 types of products will be specified in the near future.

Several different models have been reported in the literature, for estimating WEEE treatment and recycling costs. Achillasa et al. (2013) used a cost–benefit approach to analyze the appropriate manual dismantling extent required in order to maximize profitability and minimize the end-of-life management costs. Moussiopoulos et al. (2012) provided guidelines for facility locations for WEEE collection, and calculated the transportation costs for WEEE management, and Ahluwalia and Nema (2009) presented an approach to achieving the multiple objectives of economy, perceived risk, and health and environmental risk, over the entire life cycle of waste computers. However, it is still difficult to gather specific details involved in WEEE treatment research.

This study focused on the cost of the entire WEEE recycling process, to develop a cost structure and model that could be used as a reference for WEEE treatment in China. Since about 90% of WEEEs are TV sets, this study took TV sets as an example, to calculate the treatment cost and determine the contributions of the multiple factors influencing those costs.

Based on literature review, a questionnaire survey, and face-to-face interviews, this study analyzed the WEEE treatment cost structure systematically, embedded all relevant cost elements, and interpreted each constituent explicitly. The results could prove helpful for other types of waste treatment cost calculations.

The remainder of this study is organized as follows: Section 2 is a review of the literature of WEEE treatment cost calculation methods; Section 3 describes the model used here for WEEE treatment cost calculations; Section 4 is a framework to help address the WEEE treatment cost in China (especially TV sets) and give suggestions for WEEE management; and Section 5 presents the main conclusions.

2. Literature review

Kang and Schoenung (2006) identified the costs and revenue drivers by using technical-cost modeling (TCM) for the various techniques used for WEEE processing at material recovery facilities (MRFs) in the state of California, United States. The United Nations University (UNU, 2008) launched an evaluation of the implementation of the WEEE Directive in the EU Member States, focusing on the total environmental, economic and social impacts, and technical, operational and additional costs were defined in an economic and eco-environmental benefits analysis. The United Nations Environment Programme (UNEP, 2007) released their "E-Waste Volume II: E-waste Management Manual", and selected India as a case study, to analyze the technical and economic feasibility of establishing a WEEE treatment facility, with financial analysis carried out by calculating the capital, operational and maintenance costs.

Gregory and Kirchain (2008) proposed a framework for evaluating the economic performance of a recycling system, and used data on the collection, processing, and management costs as a preliminary test of the framework, creating a matrix of several net costs for stakeholders within each system. Dahmus et al. (2008) developed a general model for evaluating the economic and environmental performance of electronics recycling systems, including collection, processing, and system management costs.

The UNEP (2011) also published "E-Waste Volume III: WEEE/E-waste 'Take-Back System'", presenting 5 financing models and funding for supply chains, and, in order to cover every aspect of WEEE management, individual costs, including collection, transportation and treatment costs, were summed mathematically.

The Association for Electrical Home Appliances (AEHA, unpublished data 2009) of Japan analyzed the operational cost of WEEE treatment enterprises, mainly focusing on the costs of management, facilities investment, plant construction, waste disposal, sales revenue of valuable materials, and ongoing maintenance charges. The China Household Electrical Appliances Association (CHEAA), the China Association of Circular Economy (CACE), and the China Resources Recycling Association (CRRA) also carried out research on WEEE recycling and treatment costs in 2010, and their treatment cost evaluation included collection fees, treatment expenditures, and sales revenue. In the Taiwan region of China, the fees for the collection and treatment were estimated according to the ratified rates, the calculation of which included sales revenue and the costs of collection (including transportation), equipment, marketing, construction, land, operations, and administration. The specified WEEE treatment enterprises used their own financial methods to calculate the costs and profits, calculating the net profits as revenues minus expenses, which included corporate income taxes.

The Stiftung Elektro-Altgeräteregister (Foundation EAR, German clearing house, 2011) compiled the logistics, storage and treatment costs of WEEE, starting with the pickup of the containers at the municipal collection points, to evaluate the WEEE treatment cost; their analysis did not therefore include the public waste management authority (PuWaMA)'s cost for the WEEE collection. In 2012, the Regional Environmental Center (REC) in Turkey developed cost-benefit models based on several scenarios using different projection tools, to estimate WEEE (cooling & freezing appliances) logistics and treatment costs.

Blaser and Schluep (2012) conducted research on the economic feasibility of building a WEEE treatment facility in Dar es Salaam, Tanzania, and their model considered three main processes: collection, recycling and refurbishment. Other researchers have investigated costs in other fields, using cost-effectiveness analyses of chemical risk control policies (Oka, 2003), and input–output energy analysis in Pistachio production (Külekci and Aksoy, 2013); both of these researches were based on cost-benefit analysis (CBA). Song et al. (2013) used life cycle assessment (LCA) to discuss a typical e-waste treatment enterprise in China.

A literature review shows that the WEEE treatment cost comes mainly from collection, transportation, dismantling, recycling, and final disposal. Each of these can be calculated separately and independently, and summed to produce the total cost.

Comparing WEEE collection prices and dismantled material values (Xu and Qiu, 2011), the bigger the gap between price and value, the lower the quantity of a particular type of WEEE treated by permitted enterprises (Table 1), leading to a suspicion that some unregulated WEEE treatment enterprises may still exist in China. Countermeasures, such as raising the standard subsidy for high-value WEEE, could be implemented, in order to regulate and optimize the recycling and treatment activities.

The WEEE recycling and disposal process consists of three major activities: collection, treatment and management. Table 2 lists the detailed components, divided into these three categories, and indicates which ones have been found to be concerns. According to reports in the literature, the logistics cost is of the most concern, followed by dismantling and pretreatment fees, disposal fees, sales revenue, and labor costs; fund audit and collection costs, and labor and temporary storage costs in the collection stage, are of

Table 1
Relationship between collection quantity and dismantled material values.

Item	TV sets (CRT and flat panel)	Refrigerator	Washing machine	Air conditioner	Personal computer
Collection price/(RMB/unit)	80	74	65	167	65
Value of dismantled materials/(RMB/unit)	66	158	106	265	89
Collection quantity in 2013/(unit)	38,947,000	603,000	1,687,000	5,000	1,109,000
Collection percent	91.96%	1.42%	3.98%	0.01%	2.62%

less concern. However, the UNU, the EAR, Dahmus et al. (2008), UNIDO, CACE and CRRA considered the sales revenue of dismantled materials as a type of cost. Kang and Schoenung (2006) added consumer payments in the collection stage, and the UNU and UNEP considered the registration fee paid by recyclers, while other institutions ignored these.

In this study, considering the operation of domestic recycling enterprises in China, WEEE treatment refers to the entire end-of-life stage of WEEE in the enterprise, starting from the WEEE's arrival, and ending with the dismantled material sales and waste disposal. It does not include WEEE collection from consumers, recyclers, or public collection points (Fig. 1). Based on the cost-benefit analysis in the literature and the present situation in China, a revenue-expenditure model (REM) was formulated to estimate WEEE treatment cost.

An REM has several advantages. First, it is comprehensive, including all aspects in the WEEE treatment process; secondly, it is simple and easy to comprehend, with no sophisticated or professional calculation formulas; thirdly, each aspect or constituent is clear, distinct from all the others, and convenient to modify. WEEE treatment cost equals expenditures minus revenues; the expenditures include previous construction costs, operational costs, etc., and revenues mainly come from the sale of materials.

3. Methods

3.1. Cost-benefit model for treatment cost calculation

As mentioned above, 7 types of costs (C*dac*, C*fc*, C*lc*, C*mc*, C*moc*, C*pc*, and C*tsc*), 4 types of fees (C*df*, C*pf*, C*sf*, and C*tf*) and one revenue source were used in the treatment calculation; these are described in detail in Table 3.

The formula for WEEE treatment cost can be stated as follows:

$$\mathrm{f(cost)} = \mathrm{f(expenditures)} - \mathrm{f(revenues)} + \mathrm{f(unpredictable\ costs)} \quad (1)$$

$$\mathrm{f(expenditure)} = Cdac + Cfc + Clc + Cmc + Cmoc + Cpc + Ctsc + Cdf + Cpf + Csf + Ctf \quad (2)$$

$$\mathrm{f(revenues)} = Bma \quad (3)$$

3.2. Input elements

Many different inputs, such as WEEE purchase fees, capital investment in facilities, employee wages, treatment quantity, complexity and extent of WEEE dismantling, level of treatment technologies, and sales prices of materials, can influence the treatment cost, making cost assessment and calculation difficult. In order to calculate the cost, six elements were input to the REM, as follows:

a. The treatment cost calculation begins with the arrival of WEEE at the dismantling enterprise site, and ends with the sale of dismantled materials and valuable products to downstream companies for utilization or disposal. The sales fee was calculated as a percentage λ of sales revenue.

$$Csf = Bma \times \lambda \quad (4)$$

b. Because of the variations in WEEE composition, the sales revenues for dismantled materials and other valuable products do not remain constant. An estimated "unpredictable" cost was therefore proposed to make this model more realistic. In this research, the estimated unpredictable cost was calculated using a percentage φ of expenditures.

$$\mathrm{f(unpredictable\ cost)} = \mathrm{f(expenditure)} \times \varphi \quad (5)$$

c. According to the CRRA (2010), management cost can be simplified as a percentage α of labor costs.

$$Cmc = Clc \times \alpha \quad (6)$$

d. Depreciation was based on the expected useful life. For a processing plant, this was set to 20 years; for equipment, 15 years, with a residual rate of 5%. The cost for equipment maintenance and overhaul was calculated as a percentage β of permanent capacity value.

$$Cmoc = Cdac \times \beta \quad (7)$$

e. Enterprises take out loans from banks at an interest rate of 6.55% at the beginning of every quarter, to pay collection costs, and repay the loans at the end of every quarter.

f. Presently, treatment cost is based on the assumption that all types of WEEE have equal weight, and contain identical components in identical proportions. This research assumed, however, that the total amount, *M*, of WEEE *n* treated in fiscal year by a given enterprise, is as stated in the formula given below, whose parameters are listed in Table 4. Six costs (C*tsc* can be incorporated into C*dac*)—C*dac*, C*fc*, C*lc*, C*mc*, C*moc*, and C*pc*—are difficult to calculate for a single unit, and were therefore evaluated based on annual quantities (*M*). However, the remaining input elements—C*df*, C*pf*, C*sf*, C*tf*, and B*ma*—are usually stable, and can be calculated for single units.

$$\begin{aligned}\mathrm{F(cost)} &= Cdf + Cpf + Csf + Ctf \\ &+ \frac{Cdac + Cfc + Clc + Cmc + Cmoc + Cpc}{M} \\ &- Bma + \mathrm{f(expenditure)} \times \varphi = \{(Cpf + Ctc \times \mathrm{Dis} \times \mathrm{W}n) \\ &+ \left(\sum_{k=1}^{p} Cdfp \times \mathrm{R}p \times \mathrm{W}n + \sum_{k=1}^{q} Cdfq \times \mathrm{R}q \times \mathrm{W}n\right) \\ &+ Ctf + \left\{\sum_{k=1}^{\mu} Clc\mu \times 1 + \alpha + \left[\frac{Clan + Cco}{\mathrm{N}} + \sum_{k=1}^{j} \frac{Cp\eta + Ce\eta}{\mathrm{N}\eta}\right.\right. \\ &\left.\left. \times 1 - \mathrm{R}r\eta\right] \times (1+\beta) + \sum_{k=1}^{\eta} (Ce\eta + Cw\eta + Cf\eta) + Cfc\}/M\right\} \\ &\times (1+\varphi) - (1-\lambda) \times \mathrm{W}n \times \sum_{k=1}^{i} \mathrm{b}o \times \mathrm{R}o \end{aligned} \quad (8)$$

4. Results and discussion

4.1. Treatment cost of TV sets in China

From 2013 to 2014, 106 WEEE treatment enterprises dismantled about 1101 million units of WEEE, of which TV sets made up

Table 2
Comparison of WEEE treatment concerns, as determined by various researchers and institutions.

Item		UNU	UNEP (2007)	UNEP (2011)	EAR	REC	AEHA, Japan	Kang and Schoenung	Gregory and Kirchain	Dahmus and Fredholm	UNIDO	CHEAA, China	CRRA, China	Taiwai, China	Frequency of concern
Collection	Payment by consumers							√							1
	Registration fee	√		√										√	3
	Purchase fee		√	√					√			√	√		5
	Logistics cost	√	√	√	√	√		√	√	√	√	√			10
	Labor cost												√		1
	Temporary storage cost											√			1
Treatment	Investment		√				√			√	√		√		5
	Depreciation							√				√	√		3
	Maintenance cost		√									√			2
	Temporary storage cost	√			√										2
	Raw materials cost		√					√				√			3
	Dismantling and pre-treatment cost	√		√	√	√			√	√				√	7
	Shredding and sorting cost	√													1
	Recovery fee	√										√			2
	Sales revenues	√			√					√	√		√	√	6
	Disposal fee	√		√			√				√	√	√	√	7
	Labor cost		√					√		√	√	√	√		6
	Power cost		√							√		√	√		4
	Energy cost							√				√			2
Management	Fund management cost			√						√				√	3
	Audit cost			√											1
	Management cost						√		√			√	√		4
	Other costs						√			√		√			3
	Taxes	√											√		1
	Financial cost											√	√		2
	Fund collection cost													√	1

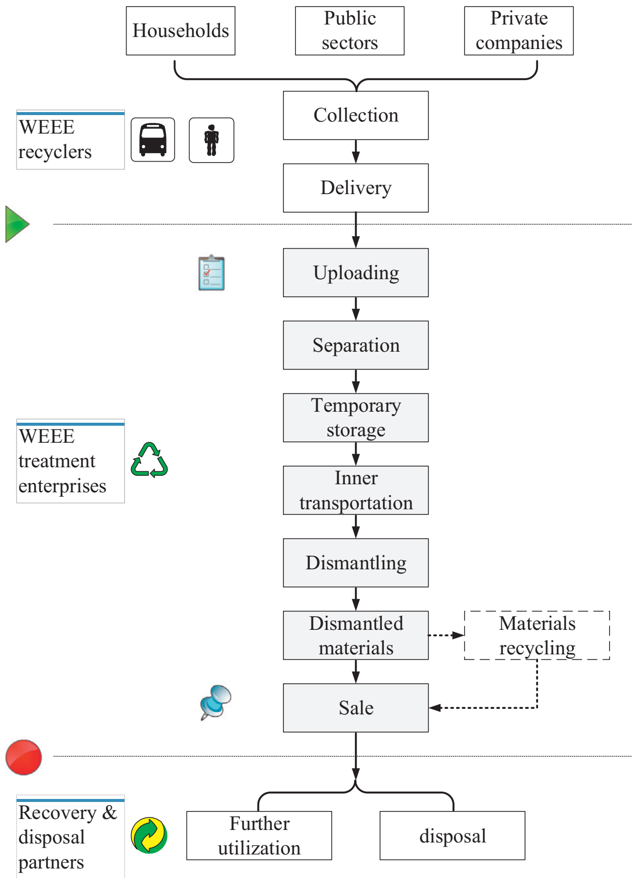

Fig. 1. WEEE collection, recycling and treatment flow in China.

87.3%, with an average of more than 900,000 units per treatment enterprise (WEEE Treatment Information System, 2014). Therefore, we selected TV sets as a representative example to evaluate the treatment cost in four typical enterprises, in different regions of China.

(1) Enterprise A

From July 2013 to June 2014, Enterprise A dismantled 1,043,118 units of WEEE, including TV sets, refrigerators, and washing machines, at percentages of 89.0%, 4.6%, 6.4% and 0.5%, respectively. This enterprise invested 60 million RMB in its plant, and had 150 employees. (All currency figures in this document are in RMB. Conversions from Chinese currency use an average exchange rate over the approximate time period related to the costs (4/1/2014-3/31/2015), where 1 USD = 6.112 RMB.)

Using the formulas developed above, with special calculation of costs and fees, the treatment expenditure for TV sets was about 107.7 RMB/unit, with *Cpf* accounting for 84.3%. By Formula (8), with sales revenues of 66.8 RMB/unit, and an estimated unpredictable cost of 10.8 RMB/unit (10% of expenditures), the TV set treatment cost was 51.7 RMB/unit. The expenditure structure for enterprise A can be seen in Fig. 2.

(2) Enterprises B, C, and D

From July 2013 to June 2014, Enterprise B dismantled 132,875 units of WEEE, with 100% being TV sets. This enterprise invested 5 million RMB and had 80 employees. Using the developed formulas, with special calculation of costs and fees, the treatment cost for

Table 3
Descriptions of treatment cost elements.

Calculation element		Description
Cdac	Depreciation and amortization cost	Calculated by straight-line service life depreciation method. The service life for a typical plant is 20 years; for facilities and equipment, 15 years, with a residual rate of 5%
Cfc	Financial cost	Interest on business loans
Clc	Labor cost	Payroll for WEEE dismantling workers, managers or other types of workers varies; based on local average wages or on data supplied directly by enterprises
Cmc	Management cost	Includes daily costs for production, technology, sales, financial and other departments, and comprehensive management, market design, recycling channels, technology research, advertising fees, etc.; calculated based on the total amount of employee wages, using a percentage α, usually 30%
Cmoc	Maintenance and overhaul cost	Costs for equipment maintenance and overhaul, usually calculated using a percentage β of permanent capacity value, usually 3%
Cpc	Energy cost	Mainly electricity, but includes water and various fuels
Ctsc	Temporary storage cost	WEEE may be stored while awaiting processing. In actual operation, this cost is included in depreciation and amortization cost
Cdf	Waste disposal fees	Disposal fees for hazardous and other types of wastes not recyclable or reusable
Cpf	Collection and purchase fees	Varies with local WEEE purchase price
Csf	Sales fee	Costs associated with material sales; usually a fixed percentage of sales revenues, typically 3%
Ctf	Taxes (non-payroll)	Based on the sales revenues. Can include value-added tax, urban construction and maintenance tax, education surcharges, stamp tax, etc. May be as high as 20% of sales revenues
Bma	Material sales revenues	Revenues from sale of dismantled materials and recycled products

Table 4
WEEE treatment cost formula parameters.

Parameter	Description	Unit of measurement
Wn	Weight of WEEE *n*	kg
bo	Benefit of material *o*	RMB
Ro	Percentage of material *o*	%
M	Annual treatment quantity	Unit
Cpf	Purchase price for WEEE	RMB
Ctc	Transportation fee for WEEE	RMB/(km unit)
Dis	Transportation distance	km
Cco	Construction cost of plant	RMB
Clan	Cost of land (purchased or rented)	RMB
N	Service years for land	–
$N\eta$	Service years for equipment η	–
Cdfp	Disposal cost for general waste *p*	RMB/kg
Rp	Percentage of general waste *p*	%
Cdfq	Disposal cost for hazardous waste *q*	RMB/kg
Rq	Percentage of hazardous waste *q*	%
$Ce\eta$	Installation cost of equipment η	RMB
$Cp\eta$	Purchase cost of equipment η	RMB
$Rr\eta$	Residual rate for equipment η	5%
$Ce\eta$	Electricity cost of equipment η operation	RMB/year
$Cw\eta$	Water cost of equipment η operation	RMB/kW h
$Cf\eta$	Fuel cost of equipment η operation	RMB
$Clc\mu$	Employee wages μ	RMB/year
α	Ratio of *Cmc* to *Clc*	%
β	Ratio of *Cmoc* to *Cdac*	%
λ	Ratio of *Csf* to *Bma*	%

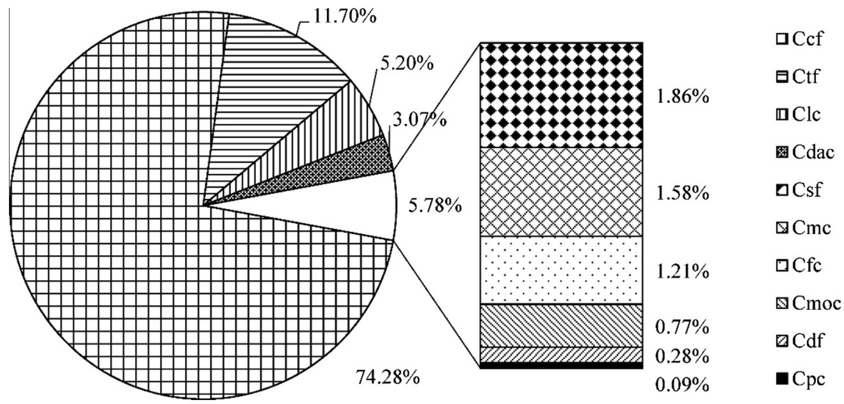

Fig. 2. Distribution of expenditures, without revenue, in Enterprise A.

TV sets was about 118.4 RMB/unit, with C*pf* accounting for 63.3%. By Formula (8), the TV set treatment cost was 63.5 RMB/unit.

From July 2013 to June 2014, Enterprise C dismantled 1,011,803 units of WEEE, with TV sets accounting for 86.9%. This enterprise invested 80 million RMB and had 184 employees. Using the developed formulas, the treatment cost for TV sets was about 135.7 RMB/unit, with C*pf* accounting for 64.9%. By Formula (8), with its estimated unpredictable cost of 13.6 RMB/unit, the TV set treatment cost was 82.5 RMB/unit.

From July 2013 to June 2014, Enterprise D dismantled 1,189,045 units of WEEE, with TV sets accounting for 92.1%. This enterprise invested 146 million RMB and had 335 employees. Using the developed formulas, the treatment cost for TV sets was about 102.9 RMB/unit, with C*pf* accounting for 70.4%. By Formula (8), the TV set treatment cost was 46.4 RMB/unit.

Comparing the treatment costs for TV sets among the four enterprises, the dominant expenditures were the same for all: C*pf*, C*tf*, and C*lc*, accounted for about 68.2%, 10.9% and 7.3% of total expenditures, respectively, and C*df* and C*pc* were the lowest: 0.2% and 0.1%, respectively. The average expenditure for a TV set was 116.2 RMB/unit. An analysis in 2007 (CAIWMB) reported costs (excluding profit) from a sample of collectors and processors, and showed a weighted average (by mass collected and processed, with the average mass of a TV set at 20 kg) of 112.4 RMB/unit, equal to the present treatment cost in China.

4.2. Main factors influencing TV-set treatment costs

As discussed above, this article considered 11 types of costs or fees, and a change in any of them would alter the treatment cost. A number of factors, including purchase price, treatment quantity, extent of dismantling, treatment technologies, the sale price of materials, and tax levels, can impact those 11 costs or fees.

Purchase price makes up the major part of C*pf*, and an increase in this cost will definitely increase the treatment cost; the extent of WEEE dismantling, and the treatment technologies, will cause changes in the investments in facilities and equipment, and also in C*lc*. Impacts from the WEEE treatment quantity and the sale price of materials need to be evaluated. This research tried to analyze the interactions among these factors. Following is a detailed discussion.

(1) Treatment quantity

In general, for any given enterprise, the average costs of C*df*, C*sf*, and C*tf* come to less than 15% of the total costs for TV sets, and have no correlation with treatment quantity (TQ); furthermore, these 3 fees usually remain constant. The following analysis is based on the hypothesis that the enterprise would not add any additional equipment, nor expand its facilities. Total C*dac*, C*moc*, and C*tsc* (about 4.0% of total cost) usually do not change after the establishment of a plant or facility. If TQ increases, the average C*dac*, C*moc*, and C*tsc* values, which are based on permanent annual land and equipment amortization, may decrease, and vice versa. Total C*lc* (about 7.0% of costs for TV sets) would increase with a growth in TQ, while average C*lc* increases would be slow compared to total C*lc*, because the number of management staff would remain the same, or increase only gradually, as contrasted with dismantling-staff levels, which might be more volatile. In this research, C*mc* was calculated as a percentage of C*lc*, so that a TQ increase might have an adverse impact on average C*lc* and C*mc*. Total C*pf* (about 70% of costs) and C*fc* (about 1.0%) have positive correlations with TQ; while average C*pf* and C*fc* would remain constant, varying only with purchase price. Total C*sf* (about 2.0%) and C*tf* (about 11.0%) also have positive correlations with TQ, while average C*sf* and C*tf* would remain constant, varying with the sales of dismantled materials and other recycling products.

Relationships between TV set treatment quantity (TQ) and treatment cost elements, and the interactions among them, can be determined for a single enterprise (Fig. 3).

Taking the TV set quantity, TQ (100%), for four enterprises in the first half of the year 2014 as the basis for study, this research performed a simulated analysis of the treatment costs, using two cases: with double (200%) and half (50%) of the present TQ (Fig. 4). The conclusion was that, under the existing investments, with no facility renovation or technology upgrades, the average treatment cost demonstrated a negative correlation with TQ. Cutting TQ in half would raise the average treatment cost by 6.7–10.2%, while doubling TQ might reduce the average treatment cost by 3.4–5.1%. If TQ rises, however, an enterprise's income will

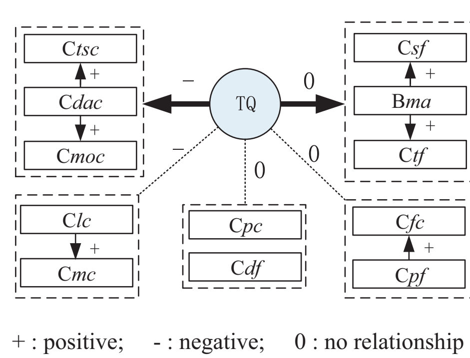

Fig. 3. Relationships between TQ and average treatment costs within one enterprise.

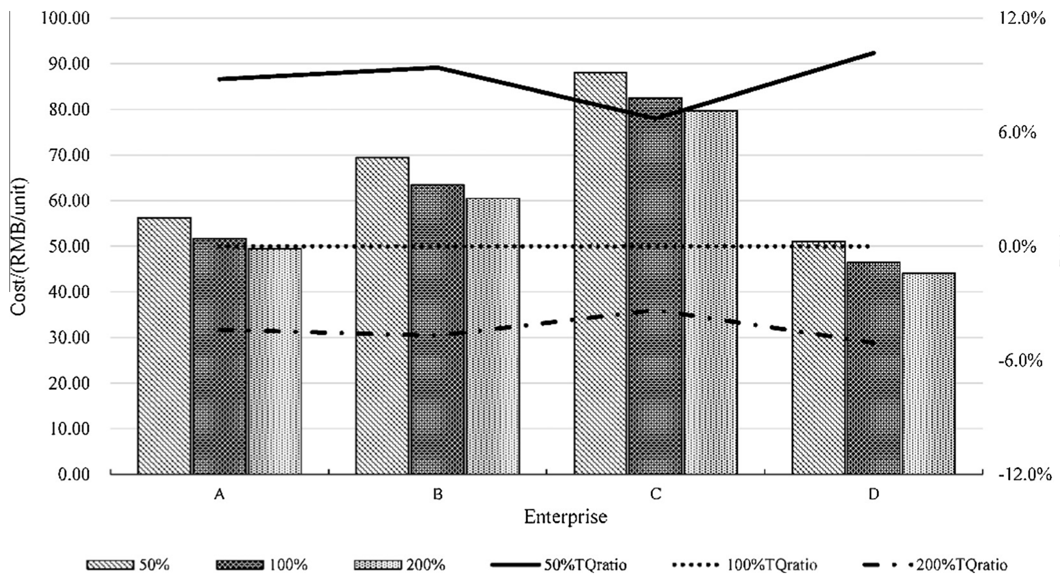

Fig. 4. Relationship between average treatment cost and TV-set TQ within one enterprise.

also increase, resulting in a double benefit, with more WEEE processed and more economic prosperity generated.

Looking at several different enterprises at the same time, however, the C*df*, C*sf*, and C*tf* will remain the same, while the C*cf* and C*fc* will change with the market conditions for WEEE purchase. And because of variation in investments in plants, equipment and facilities, C*dac*, C*moc*, C*tsc* and C*pc* will change differently among different enterprises, and C*lc* and C*mc* will change depending on local economic development levels and scenarios.

(2) Key dismantled materials

Most commonly, Chinese WEEE treatment enterprises sell printed circuit boards (PCBs), plastics, metals, and other dismantled materials directly to downstream businesses as raw materials, rather than performing complex dismantling processes that could extract higher-value products. PCBs, for example, are one of the key components of TV sets, as well as of other types of WEEE. PCBs contain valuable metals such as copper, tin, and gold, revenues from which would be 6–10 times higher than what can be acquired from direct sale of the PCB unit itself. In conducting this research, however, we found that all the enterprises sold PCBs directly, without any dismantling activities. There are quantities of researchers involved in dismantled materials recycling. Calgaro et al. (2015) developed fast copper extraction from printed circuit boards through using supercritical carbon dioxide. Okada et al. (2015) removed lead from cathode ray tube funnel glass by combined thermal treatment and leaching processes. Silveira et al.(2015) developed a complete process for recovering indium from LCD screens of discarded cell phones, and Fontana et al. (2015) investigated the recovery of indium and of the polarizing film from waste liquid crystal displays in the laboratory. Savvilotidou et al. (2015) focused on the recovery of valuable materials, metals and metalloids, Indium (In), arsenic (As) and stibium (Sb), from LCDs (Liquid Crystal Displays). According to a report by Tsinghua University (2015), however, enterprises engaging in PCB recycling using a hydrometallurgical method would see profits in just four years: not a long investment return period.

At present, some companies in China have established PCB recycling facilities using mechanical crushing technology for copper recycling, but these facilities are not operating efficiently, due to the unreliability of equipment and the low quantity of PCBs being dismantled. Fewer than 10 companies are using either a hydrometallurgical method or a thermo-metallurgical method for copper or other precious metals recycling. One reason for this deficit is that PCBs are managed as hazardous waste, and recycling enterprises need to obtain a permit or license from the local environmental protection authority, whereas WEEE treatment enterprises can recycle PCBs even without the license. Another, more important reason, however, is that the return period for new investment in more complex dismantling technology is hard to determine, although a small amount of research does exist. A further difficulty is that the market for some of these recovered products, such as non-metallic compounds and plastics, are uncertain and erratic. There are also other pressures, such as financial uncertainty and stringent environmental protection requirements.

(3) Tax levels

Currently, in China, resource recycling enterprises, including WEEE treatment ones, have to pay a value-added tax (VAT), an urban maintenance and construction tax, a (local) education surcharge, and other taxes and fees, totaling about 11.0% of their net profits. When these enterprises receive WEEE treatment fund subsidies, they must pay an additional 25% of the total subsidy as a corporate income tax (CIT). These taxes and fees constitute a significant financial burden for the enterprises, leaving them with insufficient revenue even to meet their operational costs, let alone invest in improved technology or pollution control equipment.

4.3. Measures for decreasing treatment cost

In our TV-set treatment cost calculation, collection costs came to about 70% of total expenditures. In practice, a large portion of the WEEE collection is carried out by private recyclers in China, who pay consumers for WEEE and then make a profit selling it to recycling enterprises (Yang et al., 2008). Therefore, most consumers in China are happy to sell their discarded products to street peddlers, to avoid paying the fee for discarded electronic products. Chi et al. (2014) found that informal collection by itinerant peddlers, salvage stations, second-hand shops, home appliance repair shops, and other collectors, is the primary disposal channel for urban household WEEE, with a collection percentage of 38%, because of the convenience of service, flexibility, and accessibility, and the wide variety of WEEE accepted. Therefore, in order to decrease collection costs, an integrated formal collection system, based on the existing informal network and with effective incentive schemes, should be designed, to take advantage of these Chinese WEEE disposal habits and socioeconomic realities. The new system can then adopt policies to improve collection efficiency and scope (Chi et al., 2011). Exploring options such as deposit-refund systems (Milovantseva and Saphores, 2013), credit-exchanging activities, and a "Business-to-Consumer" (B2C) recycling model (XJP, 2013), as well as creating online and offline collection systems and developing door-to-door channels, would help to raise collection quantities and increase profits for the recycling enterprises. Also, local governments and authorities could issue regulations to improve the effectiveness and safety of WEEE collection, and provide secure financing to ensure a self-sustaining and smoothly functioning system (Khetriwal et al., 2009).

Researchers have focused on WEEE recovery for resource conservation and environmental protection, developing technologies such as mechanical–physical methods (Li et al., 2014), hydrometallurgy (Birloaga et al., 2014) and thermo-metallurgy (Reuter et al. 2015). In reality, there are very few enterprises, throughout the world, that are equipped to perform extensive and comprehensive utilization of WEEE dismantled materials (Sansotera et al., 2013; Lee et al., 2007). One technique for producing high-purity metals, compounds, and materials is base-metal metallurgy: a highly sophisticated metallurgical refining operation. Copper, for example, whose relative nobility compared to other commodities and critical elements creates a strong incentive for investment in such technology, demonstrates the case for a product-centric approach to recycling, which could help achieve better resource efficiency, i.e. the recovery of metals from EoL products (Reuter and Kojo, 2014). According to the International Precious Metals Institute, the most environmentally sound final disposal of WEEE is through metal recovery by a copper smelter, followed by electro-refining (Izatt et al., 2014). There are several industrial applications, such as Aurubis (2013) which uses WEEE and copper concentrate in top submerged lanced (TSL) to facilitate recovery, followed by black copper processing and electro-refining; Dowa (Naka, 2006), which smelts WEEE with TSL in a secondary copper process, followed by a combination of copper/lead/zinc smelting and refining; Rönnskärs (Theo and Henriksson, 2009; Sundqvist, 2012) which smelts WEEE in a Kaldo reactor, upgrading the copper and followed by refining and high platinum group metals (PMs) recovery; Umicore (Hagelueken, 2006; Umicore, 2015) which uses Isasmelt smelting with copper leaching and electro-winning and PMs refining; as well as other integrated recycling plants (Zhang et al., 2015). Umicore and Dowa integrated recycling technologies can recover 17 and 18 types of elements, respectively (Zhang et al., 2015). Reuter et al. (2015) studied many of these technologies and systems, and determined the conditions fundamental to driving innovation in resource recovery efficiency, including comparisons of technological methods to a precise thermodynamic and techno-economic baseline. Such advanced recycling technologies in developed countries make those nations relatively successful in WEEE recovery. In China, therefore, developing green hydrometallurgy and other technologies for PCB recycling and extracting precious metals such as gold, silver and copper can be achieved, not only helping WEEE treatment enterprises become more profitable, but also cultivating competitiveness both in core recycling technology and in research and development. Foreign and oversea companies with advanced WEEE utilization technology could invest or participate in this area, producing profits for themselves while helping to develop and implement environmentally friendly and energy-saving technologies applicable to the environmental and socioeconomic conditions of China.

4.4. Approaches to WEEE treatment management

Lack of environmentally sound technology and facilities has resulted in poor WEEE recycling performance in China. The magnitude and continued growth of the WEEE problem, however, point to an urgent need to improve the system (Li et al., 2015).

To optimize WEEE collection a two-pronged approach is requires. First, the government, in collaboration with producers and recycling enterprises, needs to set up permanent collection infrastructure for WEEE, making it convenient and cost-effective for consumers to turn in their obsolete products. Furthermore, the existing informal recyclers could be integrated into a formal, regulated system, receiving a guaranteed base salary to encourage them to participate (Zhang et al. 2015). Secondly, public awareness of waste management and environmental protection needs to be raised, in order to capture the WEEE (currently, about 10.5%) that is discarded as general waste (Chi et al., 2014).

Widespread and effective WEEE dismantled material treatment technology and facilities have not yet been consistently implemented in China or in the Asian Pacific region, and action in this area is critical. The results of this study of TV-set treatment indicate that the cost of collection made up about 77% of total expenditures, leaving only 23% for technological development, facility upgrades and reconstruction, pollution control, and other aspects of WEEE recycling. In other words, when enterprises are burdened with such high collection costs, they cannot afford to invest in capacity increases, environmental protection measures or better dismantling technologies.

Reuter and van Schaik (2015) proposed a product-centric approach to achieving a substantial increase in the recycling rate of waste materials and components: Design for Recycling (DfR) and Design for Resource Efficiency. Using a simulation-based optimization design, they determined the environmental footprint of a "green printing" LED-light recycling system, and an optimal recovery rate of commodities—including critical scarce elements—from e-waste/WEEE and some other complex consumer products. Implementing their proposed methods, WEEE treatment enterprises engaged in PCB recycling using metallurgical processing could build relationships with electronic and electrical product manufactures, sharing information about techno-economically precise design as well as a detailed mineralogical granularity of recyclate data, to match the needs of the mineral processing industry.

On the national scale, financial institutions and tax authorities could provide more financial incentives to recycling enterprises, such as low-interest loans, tax reductions and tax credits. As for the enterprises themselves, they could realize more profits through production efficiency improvements, and by reducing overhead expenditures, establishing stable partnerships with other enterprises and producers, etc.

5. Conclusions

This study has analyzed WEEE treatment cost in detail, producing a better understanding of the complexity of the treatment cost structure. We developed an effective revenue-expenditure model (REM) to assess WEEE treatment expenditures and costs, which we divided into three components (revenue, expenditures and unpredictable costs), with seven types of costs, four types of fees and one type of revenue. Based on this model, TV-set treatment costs were calculated, using data from several different recycling enterprises in China; these varied from 46.4 RMB/unit to 82.5 RMB/unit, with an average of 60.7 RMB/unit. Among the sub-costs, collection cost accounted for the largest percentage (about 70%), with labor cost (about 7.0%) and taxes and fees (about 11.0%) accounting for much less. Taking advantage of the present treatment subsidy of 85.0 RMB/unit, WEEE treatment enterprises in China can indeed turn a profit.

In general, for a given enterprise, the average C*df*, C*sf*, and C*tf* costs for TV sets have no influence on treatment quantity (TQ). If TQ increases, the average C*dac*, C*moc*, and C*tsc* costs, which are based on permanent annual land and equipment amortization, may decrease, and vice versa. TQ may have an adverse impact on average C*lc* and C*mc* costs; while the average C*pf* and C*fc* would vary with the purchase price, and average C*sf* and C*tf* would vary with sales revenue from dismantled materials and other recycled products. In the present research, if TQ dropped by half, the average treatment cost would rise by 6.7–10.2%; while if TQ was doubled, average treatment cost might be reduced by 3.4–5.1%.

Critical materials from dismantled PCBs and other WEEE have been extracted using metallurgical technology, both by researchers and by well-established companies in developed countries. These technologies could, therefore, be implemented in China, creating profitable WEEE treatment enterprises, and making China more competitive in recycling technology and in research and development.

Currently, one of the major stumbling blocks is WEEE collection; this constitutes 77% of WEEE treatment costs. Recycling enterprises could reduce these costs by setting up online and offline collection platforms, cooperating with individual collectors, and creating door-to-door collection channels. They could also improve production efficiency and reduce administrative expenditures. The government could provide economic incentives—such as subsidies, low-cost loans, tax cuts and credits—and could also raise public awareness of waste management and environmental protection, in order to capture some of the WEEE currently discarded into the general waste stream. Foreign companies with advanced WEEE utilization technology could invest or participate in this area, producing profits for themselves while helping to develop and implement environmentally friendly and energy-saving technologies applicable to the Chinese market.

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Sustainability assessment and prioritisation of e-waste management options in Brazil

Ricardo Gabbay de Souza [a,*], João C. Namorado Clímaco [b], Annibal Parracho Sant'Anna [c], Tiago Barreto Rocha [d], Rogério de Aragão Bastos do Valle [a], Osvaldo Luiz Gonçalves Quelhas [c]

[a] Production Engineering Program, Federal University of Rio de Janeiro, Centro de Tecnologia, Sala F-101, Cidade Universitária, 21941-909 Rio de Janeiro, RJ, Brazil
[b] INESC-C, Universidade de Coimbra, Rua Antero de Quental 199, 3000 Coimbra, Portugal
[c] Program of Sustainable Management Systems, Fluminense Federal University, Rua Passos da Pátria 156, Sala 329-A, 24.020-971 Niteroi, RJ, Brazil
[d] Center for Information Technology Renato Archer, Rodovia Dom Pedro I, Km 143.6, 13069-901 Campinas, SP, Brazil

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ABSTRACT

Brazil has an increasing rate of e-waste generation, but there are currently few adequate management systems in operation, with the largest share of Waste Electrical and Electronic Equipment (WEEE) going to landfill sites or entering informal chains. The National Solid Waste Policy (2010) enforces the implementation of reverse logistics systems under the shared responsibility of consumers, companies and governments. The objective of this paper is to assess sustainability and prioritise system alternatives for potential implementation in the metropolitan region of Rio de Janeiro. Sustainability criteria and decision alternatives were defined by elicitation of stakeholders. The adopted multicriteria approach combines Life Cycle Assessment with qualitative evaluations by a small sample of regional experts with knowledge of the problem. The recommended system consists of a hybrid WEEE collection scheme with delivery points at shops, metro stations and neighbourhood centres; a pre-treatment phase with the involvement of private companies, cooperatives and social enterprises; and full recycling of all components in the country.

1. Introduction

Adequate e-waste management is still a challenge in most parts of the world, especially in developing countries. It is estimated that Brazil generated 3.8 kg of Waste Electrical and Electronic Equipment (WEEE) per capita in 2008 (Araújo et al., 2012), and 7 kg/capita in 2014 (StEP, 2015). This may be less than Mexico's 2014 generation rate (8.2), but it is more than the other BRICS countries, with the exception of Russia (China 4.4 kg/capita, India 1.3, South Africa 6.6, Russia 8.7) (StEP, 2015). Despite such a rapidly increasing generation rate, only a few adequate WEEE management systems are currently operating in the country. A large share of the e-waste produced is still disposed mixed with household waste and is destined for landfill sites, or informal chains operated by waste pickers, cooperatives and scrap dealers. The estimated recycling rate for the country is 2% (Bandini, 2009 *apud* Araujo, 2012). In Rio de Janeiro city, the composition of collected household waste in 2012 indicated that 3.7 k tonnes of WEEE were sent to landfill sites (COMLURB, 2013), wasting valuable and non-renewable resources with considerable environmental risks. It is known that rough recycling techniques like burning cables and acid leaching are commonly applied by the informal sector in the country (Souza, 2014; Lundgren, 2012). This is an insalubrious and inefficient practice to recover materials. In addition, the country seems to be an illegal receiver of e-waste from developed countries in North America (Lundgren, 2012). Illegal and informal activities are also responsible for a large amount of Electrical and Electronic Equipment (EEE) consumed in the country; in 2014 non-official markets accounted for 1.5 million purchased computers, corresponding to 15% of the total for the year, and 35% of desktops (ABINEE, 2015).

In order to tackle those issues and to implement adequate e-waste management, the Brazilian Solid Waste National Policy – PNRS (Brazil, 2010) enforced the implementation of WEEE reverse logistics under the shared responsibility of EEE producers, importers, distributors and retailers (direct chain), with broader responsibilities of governments and other actors. In order to implement reverse logistics, those EEE direct chain actors must analyse

* Corresponding author at: Department of Environmental Engineering, Sao Paulo State University 'Julio de Mesquita Filho' (UNESP), Parque Tecnológico de São José dos Campos – SP, Rod. Presidente Dutra, km 137.8, 12.247-004 São José dos Campos, SP, Brazil.

E-mail address: ricardo@sage.coppe.ufrj.br (R.G. de Souza).

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different WEEE reverse logistics options, assess technical and economic feasibility, and propose a model satisfying the objectives and principles defined in the Policy as far as possible, which include the protection of public health and environmental quality; incentivising the recycling industry and resource recovery; integrating waste management; making the articulation of the different sectors; promoting operational and financial sustainability; giving stimulus to Life Cycle Assessment, and to the integration of waste pickers' cooperatives (Brazil, 2010). Proposals of WEEE reverse logistics systems have been submitted by those actors to a council of related Federal Ministries, headed by the Ministry of the Environment (MMA), which is entrusted with analysing, suggesting alterations and selecting the model to be implemented in the country. By August 2013 four proposals had been received, but due to the complexity of these multiple interests as well as the complexity of the decision problem, namely the tasks of building a coherent set of criteria and to evaluate and compare system alternatives, two years later there was still no approved final model. Proponents pointed out some issues that still needed to be addressed, such as: the implementation of a clearly communicated recycling fee; the control of imported electronic products and the simplification of WEEE transportation and WEEE ownership (Brazil, 2015).

PNRS stimulates this decision process to be reproduced at lower government levels. For instance, a São Paulo state resolution calls for industry to introduce WEEE reverse logistics proposals. Commitments must be signed by the end of 2015 (Sao Paulo, 2015). Local government resolutions are essential because municipalities are legally responsible for Municipal Solid Waste Management (MSWM) and because reverse logistics systems must be aligned with the mandatory Municipal Waste Management Plans. The Brazilian National Solid Waste Plan (PLANARES) targets indicate that the implementation of PNRS waste management strategies must start first with the largest cities, with progressive expansion to the smallest ones.

Adequate WEEE management implementation should consider a set of sustainability criteria, aligned with both the PNRS objectives and context-specific stakeholder values. Souza et al. (2015) elicited the perspectives of stakeholders involved in the Brazilian WEEE context and specifically in Rio de Janeiro, and using decision science techniques derived a set of relevant social and economic criteria to support this particular decision. These criteria were: social inclusion; employment and generation of income; professional development; health risks and working conditions; workers access to education and healthcare; system feasibility and efficiency; population awareness and adhesion to reverse logistics; innovation and stimulus of new economic activities; and competitiveness of formal EEE products in regard to the informal ones.

Assessment of environmental performances needs a systematic approach which calculates impacts based on system modelling and resources flows along the different EEE/WEEE life cycle stages. Because of its capacity to analyse complex systems and a large amount of data, Life Cycle Assessment (LCA) has been widely applied in the context of waste management and particularly WEEE management. Besides this fact, there are still few LCA applications in Brazilian waste management.

Despite the need for relevant information to assess potential impacts of system alternatives and to make decisions on Brazilian e-waste management, there is a lack of an adequate database. Collection of primary data is often obstructive, especially in regional, local and organisational scales. A practical solution to facilitate such decision could be to promote the integration of Multicriteria Decision Analysis (MCDA) with LCA, qualitative evaluations of social and economic indicators. A robust approach to sustainability assessment and prioritisation of alternatives should be a multicriteria method that, among other features, allows for the adoption of a life cycle perspective and for a non-compensatory integration of both quantitative and qualitative indicators (Cinelli et al., 2014).

Regarding qualitative assessment of waste management sustainability indicators. In some contexts the available sample of evaluators with proper knowledge of the problem may not be sufficient to enable statistical analyses of the qualitative and quantitative measures, but the intervention of human expertise must still be considered in the decision. This can be the case, for example, of technical councils that may be organised to evaluate local WEEE management systems in Brazilian cities.

The objective of this paper is to assess sustainability and prioritise system alternatives for e-waste management in the city of Rio de Janeiro, Brazil. Specific objectives are:

- to develop an approach and consult experts for qualitative evaluation of social and economic relative performances of the system alternatives;
- to build a multicriteria analysis model, adequate for cases with small samples of evaluators; and
- to integrate MCDA with Life Cycle Assessment and with qualitative social and economic assessment.

This paper builds upon previous studies of some of the authors, namely Souza et al. (2013), where e-waste management system alternatives for Brazil/Rio de Janeiro were identified based on stakeholder elicitation; and Souza et al. (2015), where a set of sustainability criteria for Brazilian/Rio de Janeiro e-waste management was derived from stakeholder perspectives. The present study targets e-waste management specialists and decision-makers in Brazil, and seeks to recommend a solution to the decision problem, leading to the implementation of an e-waste management system in Rio de Janeiro.

2. Background knowledge

2.1. E-waste management in the Brazilian National Solid Waste Policy

According to the National Solid Waste Policy (PNRS), consumers (both population and institutions) are responsible for disposing of their e-waste separately at adequate delivery points defined in Municipal Solid Waste Plans. Retailers and distributors are responsible for returning the products to producers and importers, who in turn have to provide adequate treatment and final disposal of refuse. PNRS (Brazil, 2010) permits that these actors formalise one or more management entities, which can outsource reverse logistics operations to private waste management companies, MSWM schemes, skilled cooperatives or social enterprises. PNRS stimulates involvement of cooperatives if they have adequate training, working conditions and environmental licences to perform the required activities.

Proposals of a WEEE reverse logistics system for the country should be presented to the Ministry of the Environment by producers, importers, distributors and retailers. The selected model must be formalised into a Sectorial Agreement (SA), a contract signed by all aforementioned parties. It can also be specific to State SA and to Municipal SA. The SAs have to detail, among other information:

- descriptions of the set of integrated activities by each participant in the reverse logistics system, in the processes of collection, storage, transport, recycling and final disposal, indicating:
  - technical recommendations at each stage;
  - criteria to install and operate delivery points;
  - adopted collection schemes;
  - procedures and responsibilities for sorting, reuse, recycling, treatment, and final disposal activities;

- possible hiring of waste pickers' cooperatives and associations;
- assessment of social and economic impacts of the reverse logistics system (Brazil, 2010).

2.2. E-waste management situation and feasibility assessment in Brazil

In order to structure the Sectorial Agreement, the Brazilian government previously requested a feasibility study (FS) of the WEEE reverse logistics implementation in Brazil (ABDI, 2012). This FS pointed out that Brazil produces around 1 million tonnes of WEEE per year and that around 4 thousand collection points in 2016 would be necessary in order to achieve around 70% of the collection rate. The FS also pointed out that Brazil has 94 WEEE recycling facilities, most of them in the state of São Paulo (Southeast of Brazil, the same region as the state of Rio de Janeiro). Due to the large geographical distances within Brazil, the installation of sorting units that store WEEE components before their transportation to recycling facilities was suggested. In order to optimise cost, the FS recommend that cities with more than 150 thousand inhabitants must have a sorting unit. In addition, the FS proposed a WEEE reverse logistics model coordinated by one or more management entity and divided in two stages:

- *Primary stage:* involves WEEE collection and transportation to sorting units. At this stage, the consumer should dispose of small devices in collection points available at EEE shops. The industry, importer or management entity should provide door collection of large devices. The commerce or management entity is responsible for storing WEEE collected and transporting it to sorting units.
- *Secondary stage*: involves WEEE storage and transportation from sorting units to recycling facilities and final disposal. The management entity should coordinate the sorting units and the transportation to recycling facilities.

The FS discussed alternatives and made recommendations in eight key modelling variables: sources of resources for implementation; responsibility for orphan products (from illegal or informal markets); targets for recovery and recycling; level of responsibility of the public administration; WEEE hazardousness classification; reuse in the reverse logistics system; WEEE sorting by brands; proportional responsibility for WEEE; and competition model. Their recommendations were, respectively: shared costs by the actors in the EEE chain (from industry to consumer); no recommendation for orphan products; recycling target is 100% of products declared by the SA signed companies; government will be responsible for providing resources for research, infrastructure and campaigns; WEEE should be treated as non-hazardous but only be dismantled by recyclers; delivery points and consumer service should be made available to organise the reuse chain; WEEE should be monitored by sampling at the sorting units for identification of orphans, data checking and reporting to authorities; cost share for each producer is defined in proportion with their relative sales in the previous year; and stimulating competition among several management entities created by partnerships of producers and importers.

The FS was the base for a call for SA proposals to be presented by producers, importers, distributors and retailers. The Ministry of Environment received 11 proposals in 2013, but only 4 were accepted for evaluation (Veloso, 2015).

2.3. Existing studies in Brazilian e-waste management

There is increasing interest in developing studies on Brazilian WEEE management, but there are currently few published studies on this topic. Araújo et al. (2012) proposed a model to estimate WEEE generation and applied it at national-level using primary data from the EEE market. Franco and Lange (2011), using a survey of householders' behaviour, estimated the WEEE generation in the city of Belo Horizonte and tracked the current flows, identifying that most is either donated, kept or sold.

Trying to describe the current situation, Saavedra and Ometto (2012) identified some existing initiatives referring to state legislation, social and digital inclusion and recycling companies. Oliveira et al. (2012) provided an overview on current e-waste management practices in the country, highlighting the need for a well-defined model, the existence of a cascade reuse market, the lack of companies for complete e-waste recycling (PCBs are exported), the need for an efficient collection scheme, and the need to include waste pickers. Ongondo et al. (2011) added the evidence of WEEE being dumped. Quariguasi Frota Neto and Van Wassenhove (2013) found that existing WEEE take-back initiatives in the country are led by large multinational manufacturers, but are less advanced than those performed.

Focusing the development of solutions for the e-waste problem in the country, Pimentel et al. (2013) presented a research project, called Ambientronics, which aims at technological development to support the Brazilian recycling industry. Chatterjee and Kumar (2009) proposed an outsourcing model which integrates non-formal operators in collection, disassembly and segregation of e-waste, whereas formal actors process PCBs for resource recovery. Campos et al. (2014) highlighted the relevance of creating a 5R network – Reduce, Redesign, Recycle, Reuse and Repurpose. Hirayama and Saron (2015) analysed the composition of Brazilian waste computer equipment (WCE) and concluded that implementation of stricter regulations for identification of thermoplastic polymers in WCE in Brazil is an important step for successful mechanical recycling of these materials.

Other studies discussed Brazilian legal framework in comparison to other countries. These highlighted the need for national and global standardisation (Sant'Anna et al., 2014), the limitation of the PNRS in just requiring implementation of reverse logistics without an efficient management and control framework (Barboza et al., 2014), and the need for decisions from competent authorities observing diverging interpretations of legal stipulations and administrative procedures (Brandmann and Altvater, 2012). None of these identified studies focused on modelling and selection of an e-waste management model for the country or a city by sustainability assessment and prioritisation of alternatives.

2.4. LCA studies in WEEE management

Life Cycle Assessment (LCA) has been established as a technique to quantify the potential environmental impact using a systematic approach. LCA can provide objective indicators to compare processes or products, being an important tool in environmental management and pollution prevention (ISO, 2006). It has been widely applied in the context of waste management (Laurent et al., 2014a) and particularly WEEE management (Wäger et al., 2011; Bigum et al., 2012; Hong et al., 2015; among others). There are few LCA studies in Brazilian waste management (e.g. Mendes et al., 2003; Leme et al., 2014; Reichert and Mendes, 2014), and none specifically for Brazilian household WEEE (although Rubin et al., 2014 analyse a generic recycling system that could be applied in Brazil, as suggested by the authors).

Several studies have used LCA to analyse and compare different scenarios for e-waste treatment, aiming to support decisions, identify key factors and improve opportunities associated with all stages within a determined system boundary. Using LCA, Huisman et al. (2008) concluded that the implementation of the European WEEE Directive contributed to reducing environmental impacts in all categories evaluated. Emphasis is set on the reduction of 36 million tons of CO2 and 34 million tons of CFCs, which are no longer discharged into the environment due to WEEE recycling.

Rubin et al. (2014) used global data to apply LCA and compare two processes for recovering copper from PCBs, both adopting mechanical and electrochemical processing. They concluded that the process which employs acqua regia (a combination of nitric and chloridric acid) had the best environmental performance. Because of the narrow scope of their LCA study, with little necessity for regional data like distances or local WEEE composition, this recommended solution can be employed in Brazil or elsewhere.

2.5. Qualitative sustainability assessment of waste management

There are just a few studies adopting qualitative evaluations in waste management, most of which are relatively recent. They are essentially based on the application of survey questionnaires to a sample of specialists or stakeholders (Glew et al., 2013; Khalili and Ehrlich, 2013; Nichols et al., 2013), or judgment by the analysts based on observation of the context (Troschinetz and Mihelcic, 2009; Manhart, 2011; Al Sabbagh et al., 2012).

Specifically in the context of e-waste management, Manhart (2011) assessed a reference (baseline) and two alternative scenarios for an international recycling cooperation between developing and developed countries: in the baseline, e-waste generated in low-income countries undergoes crude recycling and uncontrolled disposal; in the first scenario, e-waste components containing valuable metals are exported from developing to developed countries for resource recovery; and in the second scenario, low-income countries import e-waste from developed ones for pre-treatment, and then export it back for recovery of precious metals. Some of these scenarios are rather similar to the alternative systems analysed in the present study (Section 3.1). Qualitatively evaluating the scenarios by own judgment, based on the study of context information, the author concluded that Scenario 1 allows for "better management of hazardous substances, higher recycling rate of scarce and valuable metals, reduced pressure on mining, lower GHG emissions, income generation for the urban poor, and investments into social and environmental standards".

2.6. MCDA sustainability assessment in waste management

There is a remarkable number of publications on the use of MCDA in waste management decision problems. Soltani et al. (2015) carried out a literature review of MCDA applications in MSWM, identifying 68 references published up to 2013. Additionally, we could identify another 27 articles from 2014 and 2015 (searched at Scopus in 4/Sep/2015). Probably the most common MSWM decision problem analysed with MCDA is the location of facilities and landfills (as studied by Liu et al., 2013; Eiselt and Marianov, 2015; Hamzeh et al., 2015).

Among the 68 studies identified by Soltani et al., only 26 have acknowledged multiple stakeholders using MCDA, mainly for assigning weights to criteria. Most of these stakeholders were experts (69% of the 26 papers), governments or municipalities (62%) and public or residents (50%). Most of the papers adopted a popular MCDA method called AHP (34 of total studies and 15 with stakeholders), which is an aggregative and compensatory method, whilst other MCDA methods were used in 15 papers with stakeholder consultation. In this case, the methods PROMETHEE and ELECTRE, which are not compensatory, were used respectively in 1 and 3 papers. Although less commonly used, non-compensatory methods (in which the criteria are not aggregated in a single synthetizing criterion) are preferable when assessing sustainability, because they enforce a strong sustainability approach (for instance, a good economic performance cannot compensate a bad environmental or social performance) and because they can handle uncertainty (Cinelli et al., 2014).

There are just a few studies of MCDA applications in WEEE management. Rousis et al. (2008) used PROMETHEE to examine 12 alternative WEEE systems for Cyprus, and concluded that the best option is "partial disassembly and forwarding of recyclable materials to the native existing market and disposal of the residues at landfill sites". In this study, the scores were attributed by the analysts and validated by experts. Gamberini et al. (2010) analysed alternative WEEE transport routes in a region of Italy, evaluated with LCA and technical parameters, and ranked using fuzzy optimisation. Wibowo and Deng (2015) also adopted a fuzzy approach, aiming to evaluate e-waste recycling programs in Sri Lanka by consulting a group of three decision makers. Each one evaluated the alternatives considering four criteria, representing the three sustainability dimensions and a technical category. The evaluations were measured on a qualitative 5-point scale from Very Poor to Very Good. The best ranked program was "Recovery of precious metals and other recyclable materials such as metals, plastic from e-waste". A problem with this approach is that the decision criteria are too broad and may lead to high subjectivity, whilst an advantage is the ability to handle subjectiveness and imprecision of qualitative evaluations.

Specifically regarding Brazilian WEEE, Guarnieri et al. (2014) proposed an MCDA framework to the problem of selecting third-party reverse logistics providers. The proposed model comprehended the following criteria categories: logistics, financial, capacity/infrastructure, value added services to customers, alliances with suppliers, and environmental issues. It is not proposed as either a method for performance evaluations, nor as an MCDA approach to prioritise the alternatives.

Among the identified studies in WEEE MCDA, the most similar to the present research was proposed by Wibowo and Deng (2015), because it worked with a small sample of evaluators (decision makers) and with qualitative indicator performances estimated by the decision makers using predefined scales. The study of Gamberini et al. (2010) is similar to the present in the aspect of integrating LCA evaluations with other performance indicators in an MCDA approach. The principal differences are associated with the decision problem analysed, the sustainability criteria and the approaches adopted for MCDA qualitative evaluation (ours are described in Section 3).

2.7. The relevance to waste management of combining MCDA and LCA

LCA, qualitative assessment and MCDA are innovative, powerful tools to improve waste management practices. The Brazilian National Solid Waste Policy mentions explicitly the objective of stimulating LCA implementations and implicitly the need for qualitative assessment by social actors. MCDA is not mentioned, but could be an appropriated tool for obeying its principle that, when managing solid waste systems, "a systemic view that considers environmental, social, cultural, economic, technological, and Public Health variables" should be adopted.

The three previous sections showed that there is vast literature underlining the specific role of each of these tools in waste management. Kiddee et al. (2013) has highlighted the usefulness of LCA to ameliorate most e-waste problems to human health and the environment, admitting nevertheless, that it must be complemented by other tools in order to enhance the decision robustness. Troschinetz and Mihelcic (2009) explored the benefits of using qualitative data when there is a lack of numeric indicators to evaluate performances of waste management systems. Wibowo and Deng (2015) concluded that MCDA is an effective and efficient approach to evaluate the performances of alternative e-waste recycling programs in a specific situation. It is a complex and challenging problem, as it involves "several decision makers and multiple evaluation criteria with the presence of subjective and imprecise assessments".

The present paper looks for an integrated approach combining those three tools, which represents an innovation in WEEE theory and practice. Such an integrated approach offers an interesting potential solution to the current challenge of incorporating all dimensions of sustainability in waste management decisions. Some authors have already anticipated the importance of this integration. Munda (2008) argues that every sustainability assessment approach is based on some form of multicriterial aggregation, explicit or not. For Cinelli et al. (2014), sustainability assessment must happen in a structured, transparent and reliable way and MCDA can largely contribute to this; moreover, they stress that an adequate MCDA method for this purpose should be able to integrate LCA and qualitative indicators. When adopting the strong sustainability paradigm, such integration should not involve a compensatory aggregation of the criteria (Valle and Clímaco, 2015).

3. Materials and methods

3.1. Alternative scenarios under evaluation

The alternative WEEE systems considered for implementation in Rio de Janeiro are based on the previous mapping by Souza et al. (2013). The baseline scenario A (Fig. 1) corresponds to the existing collection and to the treatment activities for WEEE generated in Rio de Janeiro, with emphasis on informality and landfilling. As there is no available data that indicates the total amounts of WEEE collected and processed by the existing chains (besides the landfill waste composition), the WEEE flows represented in the baseline scenario were assumed but may not correspond to reality. Rather, this scenario is relevant for assessment because it allows for perceiving the relative importance of informality and landfilling to sustainability performances.

All the other four scenarios analysed (B1, B2, C1 and C2) consider the same hybrid solution for WEEE pre-treatment, consisting of 20% of WEEE being sorted and dismantled by social enterprises and skilled cooperatives, and 80% by formal WEEE recycling companies. No informality takes place in these scenarios; cooperatives involved are adequately trained and licensed. These scenarios also consider that most WEEE components are adequately recycled regionally, assuming that recycling plants are located around São Paulo (ABDI, 2012). Some functional appliances and components are refurbished by a social program in a model similar to the *Fábrica Verde* (Green Factory), a Rio de Janeiro governmental social project (suspended early 2015) that offered training to youngsters from favelas in computer dismantling, refurbishment and maintenance. Refurbished computers were donated to NGOs and favela communities.

Alternatives B1 and B2 (Fig. 2) have a collection scheme based on WEEE delivery only at EEE shops. In B1 PCBs are exported for recycling in Europe, whilst in B2 all components are recycled regionally.

C1 and C2 (Fig. 3) consider a hybrid collection scheme with delivery points at EEE shops, metro stations and neighbourhood stations. C1 and C2 differ in the same way as B1 and B2.

3.2. LCA modelling

The LCA model used in this research was developed and is described, seeking to satisfy the recommendations by Laurent et al. (2014b) for LCA applications in waste management. It also satisfices the guidance of ISO 14040 (ISO, 2006).

LCA was applied to evaluate and compare the environmental performances of the alternative systems proposed in Section 3.1 (goal of the study). The functional unit was 1 ton of WEEE (Swiss mix as reported in SWICO, 2013).

Because there is limited information about current e-waste flows and processes as currently occurring in Brazil, this study used international data mostly from the Ecoinvent database (Hischier et al., 2007) and others (SWICO, 2013; Hong et al., 2015). The modelling also adopted assumptions in the representation of Brazilian e-waste treatment processes. Table 1 summarises the data used, model assumptions and sources of data.

The distance of 150 km adopted as a pattern in all transport processes is a standard used in the Ecoinvent database when there is no detailed local data. A sensitivity analysis was applied to all variables marked with "Own Assumption" Table 1, with a variation

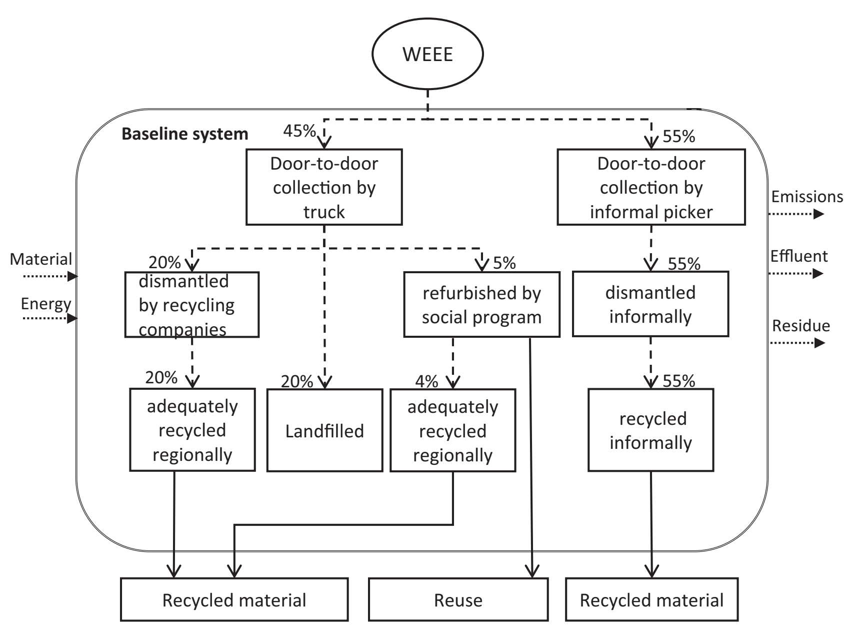

Fig. 1. Baseline system with respective WEEE flows.

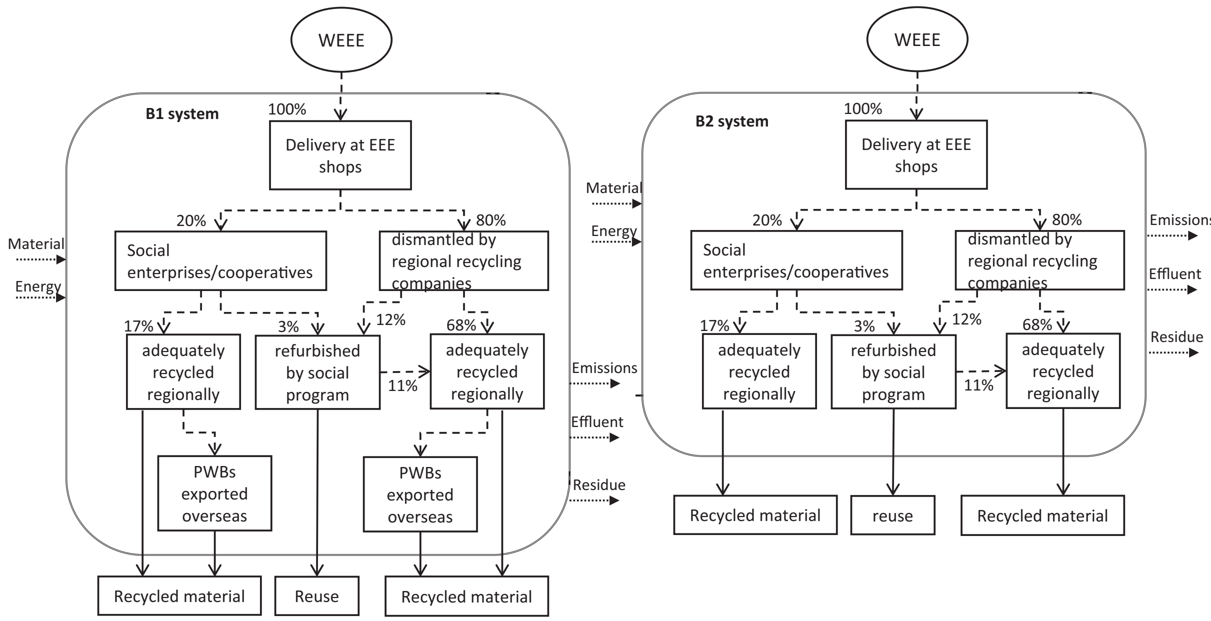

Fig. 2. B1 and B2 alternative systems with respective WEEE flows.

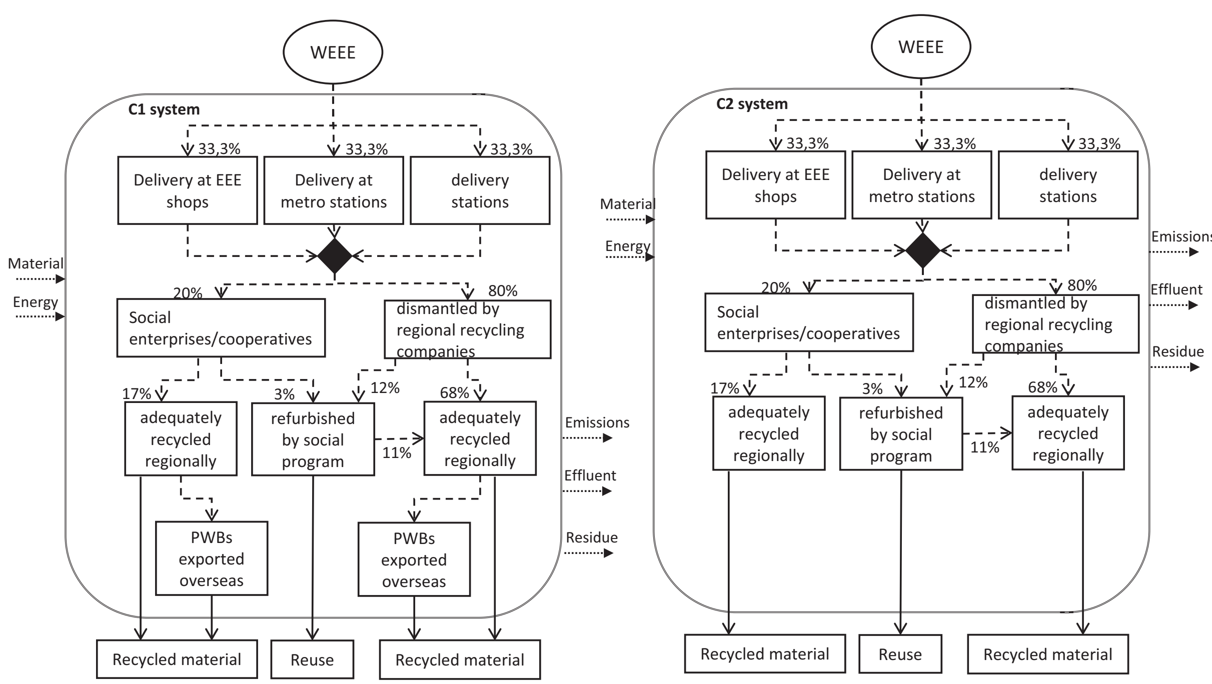

Fig. 3. C1 and C2 alternative systems with respective WEEE flows.

of 50% in the values. There was no allocation of impacts, but EcoInvent usually adopts the economic allocation.

In order to support LCA data analysis and interpretation, this study used SimaPro® 8.1, a piece of software that supports various Life Cycle Inventory (LCI) databases. Among the Life Cycle Impact Assessment (LCIA) methods provided by the software, CML (Center of Environmental Science of Leiden University) has been chosen to classify and characterise inputs and outputs from the e-waste treatment process (e.g. energy, transportation, emissions, etc.) into potential environmental impacts. The impact categories adopted are in line with previous WEEE LCA applications found in literature (Section 2.3). These are: Abiotic Depletion (and Fuels); Global Warming; Human Toxicity; Ecotoxicity (Fresh Water, Marine, Terrestrial); and Acidification.

3.3. Qualitative assessment of social and economic indicators

3.3.1. Social and economic impact categories

The social and economic impact categories assessed in this study are those previously defined in Souza et al. (2015). These categories were defined specifically for the decision problem under study, based on the elicitation of Rio de Janeiro and Brazilian stake-

Table 1
LCA model input data, assumptions and sources.

Process	Data used and assumptions	Source
Door-to-door collection by truck	Assumed 150 km between home and recycling companies with small truck	Own assumption
Door-to-door collection by pickers	Assumed work labour has no environmental impact	Own assumption
Delivery at EEE shops collection Delivery at metro stations Delivery at delivery points	Assumed 150 km between delivery points and recycling companies with medium truck	Own assumption
Dismantled by recycling company	Assumed manual disassembly of e-waste in parts and material as reported by the SWICO organisation	SWICO (2013)
Refurbishment by social program	Data collected at *Fabrica Verde* social program in Brazil which refurbished computers for donation (15% of e-waste received)	Collected by the authors
Informal dismantling Informal recycling	Assumed Chinese data with no treatment at end-life disposal	Hong et al. (2015)
Adequately recycled regionally Social Enterprises/cooperatives	Assumed European data from Ecoinvent database for dismantling companies. In case of regional PWB recycling, Chinese data was used with treatment of end-life disposal	Hischier et al. (2007) and Hong et al. (2015)
PWB recycling exported overseas	Assumed European data for recycling with use of pyrometallurgical process to recycle PWB and recovery metals	Hischier et al. (2007) and Classen et al. (2009)
Landfill sites	Assumed European data for landfilling with inert material	Doka (2009)

Table 2
Sustainability impact categories and indicators evaluated in this study. Source: Adapted from Souza et al. (2015)

Categories		Indicators
Economic	System feasibility	Direct and indirect costs – total Profit and avoided costs – total
	System efficiency	Amount of e-waste with adequate treatment/total collected Balanced demand/capacity of the system
	Population awareness and adhesion to reverse logistics	Number of citizens and companies delivering E-WASTE to formal collection points
	Innovation and generation of new economic activities	Number of recently created companies within the e-product and e-waste chains Number of companies within the EEE and E-WASTE chains with innovation recognised by the Brazilian Ministry of Science and Technology
	Competitiveness of formal products in regard to informal ones	Increase in prices of formal e-products due to increased costs by formal e-waste reverse logistics Growth of the informal market (e-products and e-waste)
Social	Social inclusion	Number of e-waste workers and relatives provided with social and psychological assistance Number of new e-waste workers that come from such groups: women; informality; prison; slums; alcoholism; drug addiction; crime; physical and mental disabilities
	Formal employment	Number of formal e-waste workers
	Generation of income	Average income of e-waste workers – formal and informal
	Opportunity for professional development	Number of workers that undertook professional training and refresher courses
	Health risks and working conditions	Number of e-waste workers (formal and informal) working in adequate conditions (equipment, protection, training) Occurrence of accidents at work and diseases directly related to the risks of the e-waste chain
	Access to healthcare	Number of e-waste workers and their relatives provided with health insurance
	Access to education	Number of e-waste workers and relatives with high level of education

holders directly involved with the WEEE context. Table 2 summarises the social and economic impact categories together with respective indicators under evaluation.

3.3.2. Specialists sample, process of inquiry and data compilation

The set of social and economic indicators was evaluated individually by a group of five experts. All of them are from universities or research institutes in Rio de Janeiro and São Paulo, and all of them have published in the areas of Brazilian waste management, WEEE and/or waste Life Cycle Assessment, in indexed international journals. For the purposes of this research, this group works as a consultancy team for the decision-maker – the WEEE management entity or the municipality. This way, the framework for decision-support, except for the LCA model, is completely based on the perceptions and judgments of regional stakeholders and experts.

The number of experts selected is aimed at the minimum possible evaluations to make a mathematical analysis acceptable. The mathematical approach used is explained in Section 3.4. The present study adopted a decision context where few experts with knowledge of the problem are available to make qualitative evaluations, in order that the approach can be replicated in other critical decision situations.

The inquiry process consisted of either face-to-face or online interviews. In the first case, evaluations were made in writing, whilst in the second case a Word file is used. First, the system alternatives and the sustainability categories were presented to the evaluators. Furthermore, they also have had the opportunity to clarify possible doubts and to validate these decision parameters. Secondly, they were asked to evaluate the relative performances of alternatives for each criterion, according to their perceptions. For this evaluation, a continuous interval scale (Fig. 1a) was presented blank, indicating a direction of preference from the lower (−) to the higher (+) impact in each criterion. Evaluators had to mark the relative position of each alternative in that scale, taking into consideration the distances between alternatives themselves and the worst/best possible scores. An example of such evaluation by one of the consulted experts is presented in Fig. 4b.

The reason for using such a scale instead of the Likert scale – the five-point scale (Very Poor – . . . – Very Good) is that it can be more intuitive to evaluators (alternatives are placed on the line as if they are in a “race”). It is also possible to translate the data into a Likert scale by defining thresholds.

After obtaining evaluations from the consulted experts, the relative distances of alternatives in the scales were measured with a rule, and normalised in the form percentage of the total length of the line. Normalisation also involved the correction of preference

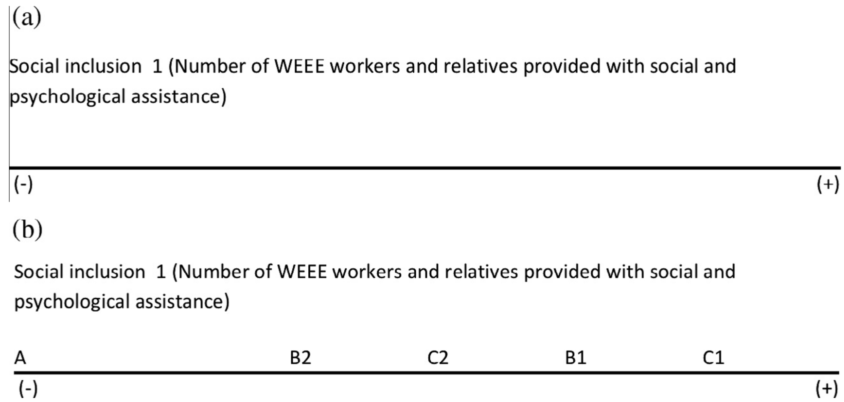

Fig. 4. Example of the evaluation scale used for qualitative assessment. (a) Blank scale. (b) Filled evaluation by an expert.

directions in the form of 0% to the worst and 100% to the best possible performances. Data provided by all experts were compiled in Excel tables.

3.4. Multicriteria analysis: composition of probabilistic preferences

The present study required a MCDA approach refusing the possibility of bad performances being compensated by good performances (i.e., non-compensatory), allowing the integration of qualitative and quantitative indicators, and permitting a reasonably trustworthy analysis of qualitative evaluations by a small sample of specialists. A good option is the recently developed Composition of Probabilistic Preferences (CPP). CPP is a MCDA approach which takes into account imprecise measurements. Evaluation of an alternative is given by the probability of such alternative being the best or the worst in comparison to all other alternatives.

The main steps in CPP are: (a) defining the criteria and the alternatives; (b) transforming the initial evaluations into probabilistic distributions; (c) calculating probabilities of preference for each alternative; and (d) obtaining the global preferences. This method is fully described in Sant'Anna (2015).

In CPP, the required input data are the preference evaluations, in our case established by five experts, as outlined above. When scores of alternatives in each criterion are obtained by those evaluations, the sets of measurements for a same alternative can be treated as samples of that distribution, and can be used to estimate all of its parameters. For the purposes of this study, the performance measurements by evaluators in the continuous scale were divided into nine preference classes, defined as profiles with stretched extremes and shrunk centrals, in order to avoid empty or too crowded classes, and thus making a better distinction of the alternatives.

A variation called CPP Beta was designed to analyse evaluations made by multiple experts. The beta distribution is applied to cases where asymmetry is assumed, thus permitting more flexibility in modelling the sample distribution. It also adopts an evaluation metric from 0 to 1, which is suitable to the scale adopted in the present study (Section 3.3). The density of the beta distribution is $f(x) = [(x-L)/(U-L)]^{\alpha-1}[(U-x)/(U-L)]^{\beta-1}/\mathrm{Beta}(\alpha,\beta)$, for x varying between L and U and Beta denoting the beta function with positive shape parameters α and β. The standard beta distribution, with extremes 0 and 1, is obtained by substituting the observed x by $(x-L)/(U-L)$. In the beta model, it is possible to model the gain in precision with the increase in the information, by fixing $\alpha + \beta = N$, denoting N as the number of experts. In this distribution the variance decreases with the sum of the parameters α and β (where α is $f(x)$ times N, and β is $(1-f(x))$ times N).

The beta distribution is unimodal (has a single local maximum), which occurs for $\alpha > 1$ or $\beta > 1$. Assuming that the x_{ijk} for alternative i according to criterion j are allowed to vary on bounded intervals (a_{ij}, b_{ij}) which are sufficiently far from L and U (0 and 1), the initial vector of evaluations $(x_{ij1}, \ldots, x_{ijN})$ gives rise to a unimodal standard beta distribution with shape parameters α_{ij} and β_{ij}. Theoretically, unimodality is desirable in CPP because the values obtained from experts' evaluations are a location around which the variable is distributed. Because the evaluation scale is a vector of observations, some experts will necessarily be attracted to extreme evaluations. If this is observed, distorting the expected unimodal shape of the beta distributions for many values for i and j, this attraction can be compensated by slightly dislocating the measures to the interval between $1/(N-1)$ and $(N-2)/(N-1)$. For instance, unimodality is granted replacing x_{ijl} by $[1 + (N-3)(x_{ijl} - a_{ij})/(b_{ij} - a_{ij})]/(N-1)$. When $N = 1$, 2 or 3, it is not possible to have $1/(N-1) < (N-2)/(N-1)$. Because of this, CPP beta requires a number of experts $N > 3$, in order to have a completely acceptable interval.

CPP calculates the following parameters: probabilities of preference of alternatives in each criterion, which are further combined in a final probability of preference; and the importance parameter of each criterion (capacity). Capacity is calculated to each possible subset of criteria. It is different from criteria weighting because there is not a fixed weight but combined values taking into account the relative importance of each criterion in a specific evaluation. The importance of each criterion is calculated by the Shapley index. Considering the possibility of interaction between sustainability categories, the preferences in each dimension can be combined into a final probability of preference with the use of the Choquet integral, which is a non-compensatory model (Grabisch et al., 2008).

In the present study, CPP Beta was modelled as following: (1) each dimension was assessed independently, assuming interaction between criteria, and using the Choquet integral to aggregate their respective probabilities of preferences; (2) the best alternative in each dimension was selected to the next stage; (3) the criteria which did not significantly distinguish between the selected alternatives were discarded; only the criteria where there was a minimum gap of 0.25 between at least two alternatives were selected for the next step; (4) the probabilities of preferences of selected alternatives (step 2) in the distinguishing criteria of the three dimensions (step 3) were aggregated, again with the use of the Choquet integral to admit interactions, in a final composition of probabilistic preferences.

4. Results and discussion

The environmental assessment of scenarios using LCA is summarised in Fig. 5. Scenarios B1 and C1, which performed best in most of the impact categories, are very similar to each other. B2

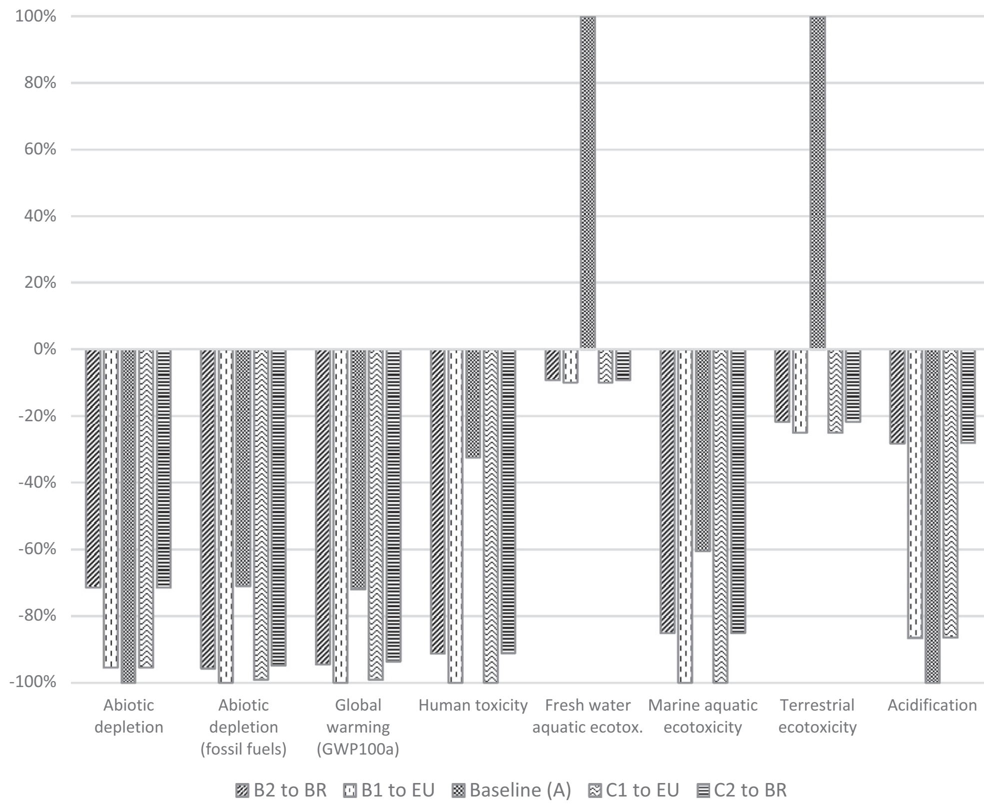

Fig. 5. Results of the LCA for the alternative WEEE system scenarios.

and C2 are also very similar. Scenario A (baseline) is the worst solution in almost all categories.

Observing the LCA results and sensitivity analyses, it is possible to conclude that WEEE collection and transport activities had little contribution to environmental impacts, in comparison to the recycling and treatment processes. Based on the life cycle perspective and restricted by the model assumptions (Table 1), all system alternatives present benefits in almost all environmental impact categories, except for system A in categories related to fresh water aquatic ecotoxicity and terrestrial ecotoxicity, which is related to contamination by inadequate disposal of hazardous material. Differences in the three ecotoxicity categories are explained by the large variations of characterisation factors of determined substances in the Life Cycle Inventory (e.g. Hydrogen fluoride has a 1E+7 highest impact potential for marine toxicity than for fresh water toxicity).

On the other hand, due to the large uncertainties, no available method is recommended to address marine and terrestrial ecotoxicity (JRC, 2011) and no conclusion should be taken about those categories. The large amount of copper, zinc and nickel released into soil in the informal recycling contributes to the major impacts on the fresh water toxicity in baseline system.

Results indicate that recycling PCBs in Europe is more positive to the environment than recycling regionally, considering the assumption that Brazil would adopt a technology similar to the Chinese (Table 1). We have to consider that Brazil is still developing a national technology to recycle PCBs, and when this technology and national data is made available, those results may be changed.

These results also point out that the collection scheme with WEEE delivery only at EEE shops is environmentally better than the hybrid scheme with metro and neighbourhood stations. The sensitivity related to assumed distances was analysed for 50% variation, and there was no significant difference (<5%) to the final LCA results.

The qualitative assessment of social and economic categories, as it could be expected, presented significant variations in the evaluations of the experts. This is mainly due to subjectivity in the interpretation of system alternatives and impact categories, though the inquiry process searches to minimise this effect with clear explanations.

CPP was applied to integrate the criteria within each dimension, using the Choquet integral, and the minimum and maximum probabilities of preference for each alternative in each criterion were calculated. Table 3 presents the results of this round. In the economic dimension, C1 and C2 were the best alternatives (class 7 of 9); in the social dimension, B2 is the most recommended system.

Applying CPP to the environmental dimension, B1 is the best-ranked alternative. In the social dimension, B2 is the best alternative, and in the economic assessment, C1 and C2 (economic) are the highest ranked alternatives (Table 4). These four alternatives were selected to the next assessment stage, which is the integration of the three dimensions, adopting only the criteria that provoked a considerable distinction of the alternatives in each dimension (at least 0.25 between two alternatives). Results of this final stage are summarised in Table 4.

Table 3
Minimum probability of preference and classification of alternatives (in a 9-point scale) in the social and economic dimensions.

System	Economic		Social	
	Min.	Rank	Min.	Rank
A	0.04766	2	0.01798	5
B1	0.01580	6	0.07203	6
B2	0.05232	6	0.02254	7
C1	0.00770	7	0.01625	5
C2	0.06817	7	0.06190	5

Table 4
Final ranking of alternatives.

System	Maximum probabilities of preferences (shapley values of criteria)						Final rank (%)
	0.1621 Acidification	0.1546 Resource depletion	0.1885 System efficiency 2	0.1819 Innovation 2	0.1950 Social inclusion 2	0.1179 Health risks 2	
B1	18.60%	18.74%	13.82%	13.24%	12.88%	29.56%	21.73%
B2	36.04%	36.70%	12.84%	34.32%	24.89%	19.66%	33.41%
C1	16.10%	19.55%	33.80%	15.59%	21.27%	34.94%	29.04%
C2	29.27%	25.01%	39.54%	36.85%	40.95%	15.84%	38.36%

Adopting the CPP approach described in Section 3, C2 was considered the best WEEE management system to be implemented in Rio de Janeiro. This system consists of a hybrid WEEE collection system with delivery points at EEE shops, metro stations and neighbourhood centres, and comprehends recycling of all components (PCBs included) regionally in Brazil.

This solution is different than that proposed by Manhart (2011) for developing countries, which consisted of exporting PCBs to developed countries (more similar to B1 or C1). This can be explained by the different approaches adopted to assess the alternatives. Whilst the method by Manhart (2011) was based on the author's judgements in six decision criteria (Section 2.5), the current research adopted a framework fully based on stakeholder perspectives and experts evaluations, fully designed for the specific Brazilian context. On the other hand, B1 or C1 could be an intermediate step for WEEE management implementation in the country, before the installation of adequate technology to recycle PCBs.

The adopted methodology in this study can be considered adequate to the kind of problem that is under consideration – lack of data and few evaluators available. Positive points are the integration of LCA with qualitative evaluations, the assumption of interactions between criteria, the non-compensatory aggregation, and the presentation of results as probabilities of preference. A limitation of the approach is that $N > 3$ is a mathematical requirement for the equations, but does not guarantee as precise results as could be obtained with statistical inference and a larger sample of experts. This approach can be a feasible solution to similar decision-support problems with both quantitative data and qualitative assessment by a small sample of evaluators.

5. Conclusions

The present paper developed and applied an approach to prioritise alternative WEEE management systems in Brazil according to their sustainability performances, comprehending the integration of LCA with qualitative evaluations, and considering a small sample of evaluators. This multicriteria approach, which integrated LCA, CPP and qualitative measures, can be useful in cases where there is a lack of data and a small number of evaluators with knowledge of the problem. However, precision of results could only be ensured with the use of statistical inference, which requires a larger sample of respondents. So, it adds scientific value to waste management literature and practice by proposing a much needed method that integrates LCA, qualitative evaluation and MCDA, and that does not require a large number of evaluating experts.

Another contribution of this paper is suggesting the most adequate WEEE take-back system for the case of Rio de Janeiro, with basis on a sustainability assessment of available alternatives. The recommended solution is a hybrid collection system with delivery points at EEE shops, metro stations and neighbourhoods, integration of social enterprises and cooperatives in the pre-treatment processes, and adequate recycling of all components in the country. Achieving this scenario, however, may require a progressive implementation, in which a starting point could be organising collection and pre-treatment phases and avoiding informality and landfilling, but temporarily exporting PCBs to developed countries with adequate technologies.

Further steps in this research are: to make qualitative evaluations with a large sample of both national and international experts, in order to allow for statistical inference; to apply different MCDA methods using the same collected data, in order to test their performances in such a decision problem; to collect primary data for more precise evaluations, both for the environmental assessment (LCA) and social and economic indicators. Another possibility is to integrate GIS with the methodology in search for logistics optimisation (routing and siting).

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WEEE management in Europe and China – A comparison

S. Salhofer [a,*], B. Steuer [a,b], R. Ramusch [a], P. Beigl [a]

[a] BOKU University of Natural Resources and Life Sciences, Vienna, Department of Water, Atmosphere and Environment, Institute of Waste Management, Muthgasse 107, A-1190 Wien, Austria

[b] University of Vienna, Department of East Asian Studies, Institute of Sinology, Spitalgasse 2, Hof 2, Tür 2.3, 1090 Wien, Austria

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ABSTRACT

Over the last years Europe and China have developed specific regulations to address the challenge of managing Waste Electrical and Electronic Equipment (WEEE). Households in today's urban China are similarly equipped with electrical and electronic appliances as households in European metropolitan areas, which in turn will lead to similar per capita generation rates in WEEE. While the challenge is a similar one, the systems, technologies and legislation in place in Europe and China are partly different, partly aligned to each other. In Europe WEEE collection is based on existing municipal structures. Additionally, retail and other take-back channels are in place. In China the informal sector dominates WEEE collection, being more competitive and flexible and offering pecuniary reimbursement to consumers.

In Europe manual dismantling as a first treatment step has been gradually replaced by mechanical break up of appliances, followed by sorting out of hazardous and valuable components. In the subsequent second treatment level, cathode ray tubes are separated, whereby compound materials like motors and coils are mechanically treated, printed circuit boards go to special smelters, and plastics are separated and partly recycled. In China large formal dismantling capacities have been set up in recent years. There dismantling practices follow similar principles as in European plants; however, further processing is only partly implemented in Chinese recycling facilities. Specifically metallurgical treatment of printed circuit boards is still not existent in China.

Companies selling electrical and electronic products within the EU are obliged to organise collection and treatment. This has led to a larger number of producer responsibility organisations. Financed and controlled by producers and importers, these systems aim to fulfil legal requirements at optimised costs subject to compliance with environmental standards and monitoring requirements. The Chinese system is built on a state controlled fund which subsidies formal recyclers. For these recyclers this financial support is essential to compete with informal recyclers, who operate at lower costs and do not necessarily comply with environmental standards.

1. Introduction

Today, the management of Waste Electrical and Electronical Equipment (WEEE) is a core element, as part of Waste Management strategies, beside the management of other waste streams such as residual (mixed) household waste, recyclables and hazardous waste. A key driver is the rapid increase in quantity of WEEE, which is characterised by its partly hazardous nature (content of heavy metals, polychlorinated biphenyls, brominated flame retardants and other hazardous materials) and its content of valuable materials (copper, precious metals, other metals including "critical" metals, es defined in EC, 2010). Furthermore, the role of producers ("Extended Producer Responsibility") in collection and recycling schemes and the potential influence on future design criteria for more environmental friendly products ("Eco design") make WEEE a focus point.

In the EU comprehensive legislation relating to electrical and electronic waste, in particular WEEE directive 2002 and RoHS directive 2002, came into force by mid-2005. Core elements of the WEEE directive include a compulsory collection rate of 4 kg/cap/yr and recycling targets differentiated by product categories. Substance bans on lead, cadmium, hexavalent chromium, polychlorinated biphenyls and others were laid down in the RoHS directive, thus complementing the WEEE directive. 10 years after the first version of the European WEEE directive, a revised version

* Corresponding author.
E-mail address: stefan.salhofer@boku.ac.at (S. Salhofer).

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has been decreed (2012/19/EU). The initially set collection target turned out to be too low, specifically in countries with high level of consumption and waste generation. The amendment is more differentiated.

In the past, larger quantities of WEEE were exported, mainly to countries with low environmental standards. For Germany, for example, the export of WEEE in 2008 was estimated at 155,000 t (Sander and Schilling, 2010). These exports were heavily criticised and are now subject to more detailed regulations (Annex VI of the WEEE II Directive), which define under which circumstances export may take place. In the future, exports are only possible in compliance with the European Waste Shipment Regulation; inspections of exports are an obligation to member states and in case of export for reuse, evidence of the functional capability is required for exported products. Recovery targets for the treatment process have also been adapted to align to new product categories. Additional standards will be imposed for treatment of WEEE, where the European Commission has already requested the European Standardisation Organisation to develop European standards for the treatment, including recovery, recycling and preparing for reuse. These standards should reflect the state of art. While some standards are already published (e.g. EN 50574) or drafted (EN 50626-1), others are currently in preparation. With global economic interdependencies (including producers and consumers) and related material flows and cycles these European regulations also affect Asian countries and their economies.

In the People's Republic of China (PRC) the motivation behind their present Chinese WEEE legislation has been driven by the hazardous contents of WEEE – with their environmental and health affecting properties, and the economic aspects linked to the recycling of discarded electronics and re-use of secondary materials. These drivers are shaped by the PRC's focus on economic growth and associated growing need for resources to fuel the Chinese growth engine. In the early phases of Chinese WEEE regulation the focus was not domestically generated, but imported WEEE, which started to play a significant role since the early 1990s in China: For the early 2000s, the most pessimistic estimations state that about 70% of WEEE generated in high income countries (20–50 million tons per year according to Puckett et al. (2002)) was transported to China (Wang et al., 2013; Lundgren, 2012). In respect to absolute amounts, the estimations vary between 1.5–3.3 million tons and 14–35 million for the same period (Yu et al., 2010). As initial institutional response Chinese authorities put a ban on imports for certain product types, later called the "Green Fence Initiative" in 2013.

The further development of legislation for domestic WEEE management in China has had three significant characteristics: (1) It is embedded within the environmental industrial policy legislation of the 2002 Cleaner Production Promotion Law (CPPL) and the 2008 Circular Economy Promotion Law (CEPL); (2) the most significant regulations currently in place, i.e. the China WEEE directive ('The Regulation on Management of Waste Electrical and Electronic Equipment, Recycling and Disposal') and the China RoHS ('The Regulation for the Control of Pollution caused by Electronic Information Products') both emulate the respective European versions; and (3) the most decisive developments for national legislation have been derived from experiences generated by local pilot projects during 2003–2011. The first batch of WEEE pilots was implemented in 2003 in Zhejiang and Qingdao city and simultaneously in Beijing and Tianjin in the form of Public–Private-Partnerships. The thereby conducted collection and dismantling operations aimed at setting benchmarks for national WEEE recycling standards: Know-how for regulations was gained from trials with collection channels, recycling technologies and financing models (Chung and Zhang, 2011; Hicks et al., 2005; Qu et al., 2013; Yu et al., 2010). The second round of pilots was implemented during 2009–2011 and built on the lessons of the first batch. Going by the name of "Old for New" (OfN), the pilot scheme has been simultaneously implemented in nine areas and focussed on collection channels, recycling technologies and foremost on cost management (Zhou and Xu, 2012). The driving economic incentive behind the scheme was the offering of a rebate to users participating in the scheme; if they handed over their old appliances to a formal collection point or retailer, they received a discount on an equivalent new device of up to 10% (Wang et al., 2013). In 2011 the China WEEE directive became effective. It has a limited scope (only 5 types of appliances are formally covered, i.e. TV-sets, refrigerators, washing machines, air conditioners and computers), regulates dismantling by the issuance of a list of components to be removed, and provides subsidies to recyclers fulfilling technical standards and providing monitoring data.

In order to compare WEEE management in Europe and China, the following aspects were taken into consideration: collection – structures in place to collect end-of-life appliances for recycling; treatment – technologies and capacities for treatment; system setup – stakeholders involved and mechanisms for financing and monitoring. From the comparison of the two regions, strengths and weaknesses of both systems are concluded.

2. Generation and collection

2.1. A comparison of WEEE generation in metropolitan areas

Waste collection schemes need to be adjusted to the quantity and characteristic of the waste stream to be collected. For this purpose, the level of equipment of households, both in Europe and urban China was analysed. Fig. 1 shows a timeline of the number of home appliances and electronic equipment per household in urban China (Wang et al., 2013). The graph reveals a steady growth in equipment rates of computers, mobile phones and air conditioning devices in the past twenty years. In contrast, the number of refrigerators and TV sets has stagnated over the last few years. The highest rate of increase is exhibited by the category of mobile phones, amounting to more than two devices per household.

Table 1 extends the information on equipment rates of urban Chinese households with those of Beijing and Vienna, both taken as examples for metropolitan areas in China and Europe respectively. The comparison shows that Beijing households feature similar equipment levels to other urban households in China, where approx. 670 million people live (United Nations, 2014). The mobile phone equipment rate in China is considerably higher (2.1–2.3 mobile phones per household) compared to Vienna (1.0 mobile phones per household), the same holds true for TV sets. The household equipment rates in urban Chinese households and in Beijing are similar compared to Vienna; this indicates that in terms of EEE equipment and WEEE generation, urban households in China by no means lagging behind a European city like Vienna. In addition it can be concluded, that the actual collection rates of WEEE in China might bear a WEEE potential that is probably 3–4 times higher (see data from Baldé et al. with generation rates of 4.4 kg/cap/yr in 2014 and collection rates of 0.9 kg/cap/yr in China 2013). Having in mind an urban population of currently more than 700 million people in China, this might lead to a significant e-waste generation in the medium term.

2.2. WEEE collection in Europe

In Europe, the majority of collection schemes for household appliances has been set up in partnership with existing municipal collection schemes for recyclables and hazardous household waste, and additional take-back schemes by retailers. In some countries

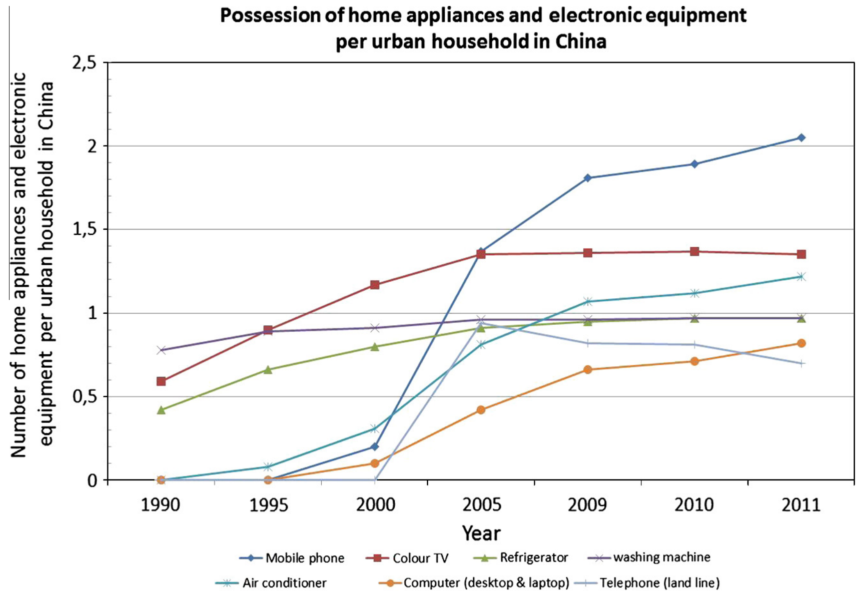

Fig. 1. Number of home appliances and electronic equipment per urban household in China. *Source:* Wang et al. (2013).

Table 1
Comparison of equipment rates of households in urban China, Beijing and Vienna.

Category	Urban household China (2011)	Beijing (2012)	Vienna (2011)
Mobile phone	2.1	2.3	1.0
Colour TV	1.4	1.4	0.9
Refrigerator	1.0	1.0	1.0
Washing machine	1.0	1.0	0.9
Air conditioner	1.2	1.8	na
Computer (desktop and laptop)	0.8	1.1	0.8
Data source	Wang et al. (2013)	Hexun (2012)	Statistik Austria (2011)

Note: na signifies "not available".

(e.g. Belgium, France) take-back through reuse centres plays an important role, while in others scrap dealers are a relevant collection avenue (e.g. in Greece).

Quantities of WEEE generated vary considerable between wealthy countries (such as Austria, Belgium, France, Germany and Sweden with more than 20 kg/cap/yr) and less affluent ones. Collection quantities range between 4 and 17 kg/cap/yr, depending on the development stage of the collection schemes. According to Baldé et al., 2014 the EU member states in average have collected 3.2 Mt from 9 Mt generated in 2014, representing an average collection rate of 36%. Most successful collection schemes can be found in Scandinavia (Sweden), where 17.5 kg/cap/yr have been collected in 2012. More details on generation and collection rates of WEEE in selected EU member states are shown in Table 2.

Table 2
WEEE quantities generated in 2014 and collected in selected EU member states. *Source:* Baldé et al. (2015)

Country	WEEE generated	WEEE collected	
	(2014)	kg/cap/yr	Year
Austria	22.0	9.0	2012
Belgium	21.4	10.3	2012
Bulgaria	10.7	5.3	2012
France	22.1	6.8	2010
Germany	21.6	8.5	2012
Greece	15.1	4.2	2010
Italy	17.6	3.8	2012
Sweden	22.2	17.5	2012

In Western Europe member states, the initial WEEE directive collection target of 4 kg/cap/yr is met easily, while for new member states this is still a challenge. As higher collection targets (65% of the quantity put on the market or alternatively 85% of WEEE generated) are mandatory under the WEEE II directive from 2019 onward, several regions have started to analyse options to raise the collection efficiency. Beside conventional take-back at municipal collection sites the following collection routes have been tested:

- Kerbside collection at multi-family dwellings as a convenient option for residents of densely populated areas. A collection trial in Vienna showed collection rates of small WEEE of 0.4–1.1 kg/cap/yr, kerbside collection in Copenhagen reached 1.33 kg/cap/yr (Borregaard, 2013). Here small WEEE are defined as all types of WEEE smaller than 50 cm, excluding cooling and freezing equipment, lamps and screens. Small WEEE are regarded as the most valuable part of the WEEE waste stream, containing IT as well as consumer electronics with a higher content of non-ferrous and precious metals.
- Container collection in public places; case studies have been found for Sweden and Germany; collection rates for small WEEE range from 0.04 to 0.84 kg/cap/yr (cf. Salhofer, 2014).
- Intensified collection of small WEEE at retail outlets; two case studies from Sweden and Germany are analysed, both showing low quantities recovered (cf. Salhofer, 2014).

The challenge of the coming years will be to identify ways to intensify collection by offering a better collection service to citizens and providing more information to motivate citizens for collection.

2.3. WEEE collection in China

Today informal structures dominate WEEE collection and take-back, as they have wide urban collection networks, offer high reimbursements to consumers and have access to a bigger and cheaper labour force compared to their formal counterparts. These characteristics have already been confirmed in the initial pilot trials with formal collection schemes, in which attaining sufficiently high rates in WEEE collection posed the major problem, as collection systems were either controlled or strongly permeated by informal actors (Yu et al., 2010; Hicks et al., 2005; Chung and Zhang, 2011; Qu et al., 2013). The Chinese government has been learning from these initial difficulties: the OfN scheme (cf. Table 3) tried to make the formal collection system more attractive in two ways: One the one hand, electronic retailers and other formal take-back entities were given subsidies to offer incentives to consumers to return their WEEE into formal channels. On the other hand, recyclers also received comparatively high subsidies, enabling them to successfully compete with informal collection systems (Zhong, 2010; Wang et al., 2013; Liu and Shen, 2012).

The results in terms of collection rates in the OfN programme are shown in Table 3. Data on collected amounts were provided by MEP (2012); calculations are based on the unit weights (16 kg per TV, 24 kg per refrigerator, 70 kg per washing machine, 47 kg per AC and 9 kg per PC) as given in the 'Guidelines for Subsidies to WEEE Treatment Enterprises', (MEP, 2010, bulletin Nr. 83).

Although the OfN scheme has achieved respectable collection rates (0.4–2.1 kg/cap/yr), the persistent problem for formal systems remains collection costs, which primarily originate as a consequence of competition with the informal sector. Compared to private enterprises in the government pilots, the informal sector has lower personal costs related to collection and recycling, which allows to pay higher prices for WEEE to generating households and companies (Yang et al., 2008).

In fact, observational and anecdotal evidence confirm that even after the OfN offered higher, unit based state subsidies to formal recyclers via the China WEEE fund, the informal segment could still regain dominance over the collection of WEEE from households. The entangled relationship between urban residents and informal collectors has been stated by previous research as major reason for this development (compare Wang et al., 2011; Chung and Zhang, 2011; Wang et al., 2013; Yang et al., 2008; Chung and Poon, 2001) and may in essence originate in the societal value structure of urban Chinese society: Household waste, including WEEE, is widely perceived as a valuable commodity and thus is expected to be exchanged for money. Unsurprisingly residents thus prefer informal collection systems for WEEE, e.g. 71% in Beijing (Wang et al., 2011), since they offer pecuniary reimbursements and convenient doorstep collection services. In fact many major Chinese cities exhibit high amounts of informally collected WEEE from households (see Fig. 2) that by far exceed formally collected or received amounts (see Table 4).

It has been generally observed that once WEEE has entered the informal channel it arrives at one of the following nodes: (1) informal dismantling, which may take place at the original area of collection or in major informal recycling hubs, e.g. Guiyu, Longtang or Taizhou; (2) formal recyclers, to which WEEE is sold, when informal collectors deem the offered prices appropriate; (3) second hand markets that are spread over Chinese cities and fringe areas and where devices are repaired for reuse. According to the analysis of these markets in literature, WEEE inputs from informal collectors range between 60% and 85% (Li et al., 2012; Veenstra et al., 2010; Wang et al., 2011), which is reasonable considering the high degree of informally collected WEEE (Fig. 2). Similarly, formal recyclers receive an equally high share of WEEE from the informal segment: Questionnaires disseminated amongst 12 formal WEEE recyclers indicate that in most cases 90% of WEEE dismantled at these facilities were collected by informal channels (Steuer et al., 2015).

3. Treatment

3.1. Treatment in Europe

The recycling process of WEEE typically includes dismantling, processing and end-processing. Dismantling is conducted by manual dismantling and separation of hazardous as well as valuable components. Driven by high costs of manual labour in Europe, mechanical processing has been developed to replace manual dismantling as much as possible. Here technologies to break up appliances in a slowly rotating drum ("smasher") or to cut up appliances ("cross-flow shredder"), both followed by a sorting process for hazardous and valuable components are in use. For the subsequent processing by crushing and separation a wide spectrum of technologies like hammer mills, magnetic separation, sieves, eddy current separators and other classifiers have been installed and improved in the European recycling industry. Thoroughly applied, these technologies produce high quality secondary products, mainly metal concentrates as input to metal mills and plastics. Additional, hazardous components and not recycled materials are sent to disposal by incineration, landfilling, hazardous waste treatment, etc. For details of the treatment processes see Cui and Forssberg (2003), Salhofer and Gabriel (2000) and others. The effect of dismantling on the separation of hazardous components was analysed for WEEE treatment plants in Austria (Salhofer and Tesar, 2011). Modelling the potential content of components containing hazardous substances in the input material and comparing them to the output of the plants led to removal rates for selected components of 50–70%, demonstrating the limitation of manual dismantling on a case study basis.

A typical treatment sequence for WEEE in Europe, based on Wäger et al. (2011) comprises the following steps: after collection and transport the end-of-life products reach the recycling facility, where sorting, dismantling and processing takes place. In the following treatment, CRT screens are separated into front and neck glass and the fluorescent powder is removed. Compound materials such as cables, motors, and coils are further processed mechanically to separate materials (metals from plastics and non-ferrous

Table 3
Collection rates in the old-for-new pilot scheme.

Region	Inhabitants (1000)	TV sets (kg/cap/yr)	Refrigerators	Washing machines	Air conditioners	Computers	Total
Beijing	19,600	0.64	0.16	0.73	0.05	0.04	1.62
Tianjin	12,280	0.50	0.08	0.42	0.02	0.02	1.04
Shanghai	23,019	1.62	0.07	0.33	0.02	0.03	2.08
Jiangsu province	77,250	0.67	0.04	0.27	0.01	0.01	1.00
Zhejiang province	51,800	0.54	0.03	0.12	0.01	0.03	0.74
Fuzhou	6380	0.74	0.03	0.21	0.01	0.01	0.99
Shandong province	95,790	0.30	0.03	0.13	0.00	0.01	0.47
Changsha	7044	0.55	0.05	0.25	0.02	0.00	0.86
Guangdong	104,300	0.19	0.03	0.18	0.03	0.00	0.44

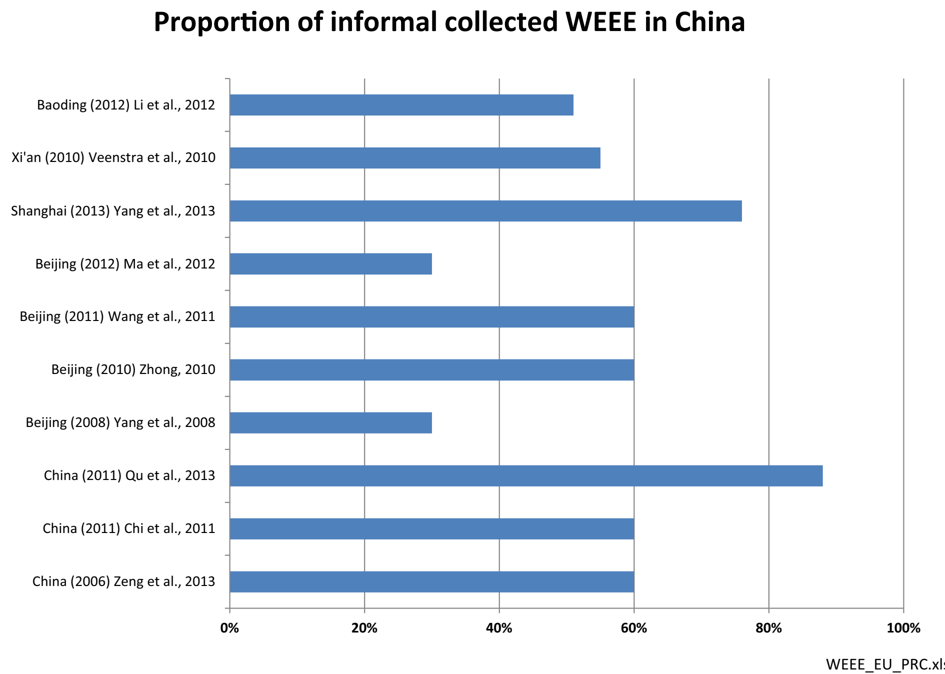

Fig. 2. Proportion of informal collected WEEE in China. (See above-mentioned references for further information.)

Table 4
Formally collected amounts of WEEE in urban Chinese areas.

WEEE received by	Area	Proportion of WEEE generated (%)	Year	Reference
Recovery stations	Peking	10	2010	Zhong (2010)
	Peking	10	2011	Wang et al. (2011)
	Shanghai	24	2013	Yang et al. (2013)
	Baoding	13	2012	Li et al. (2012)
	Xi'an	7.8	2010	Veenstra et al. (2010)
Return to retailers during the OfN	Peking	20	2009–2011	Wang et al. (2011)
	Peking	20	2009–2011	Zhong (2010)

metals from ferrous metals); often this task is assigned to specialised treatment companies. Batteries from dismantling are sent to specialised battery recycling facilities. Other waste (wood from cabinets, insulation materials, hazardous materials, etc.) is sent to disposal (see Fig. 3).

Printed Circuit Boards gained from dismantling are traded and sent to specialised metallurgic treatment facilities. Within Europe there are three plants (Aurubis, Boliden, Umicore) which apply different smelting processes, followed by other steps such as hydrometallurgy. These special smelters reach high recycling rates, for example, Umicore states recycling rates for precious metals of more than 95% (Hagelüken, 2012).

Plastics from dismantling and – to a larger extent – from mechanical processing undergo sorting (amongst others sensor based sorting, heavy media separators), and then go into material recycling, incinerated or disposed of in landfills, respectively.

3.2. Treatment in China

Along with the establishment of strict regulations for the treatment of WEEE (cf. Zeng et al., 2013) and supported by a subsidies program (see Section 2.3) large capacities for WEEE treatment have been developed. By mid of 2015, 106 WEEE recycling plants were included in the WEEE China funding scheme, and WEEE treatment has reached a volume of 1.458 mio. t in 2014 (Hu, 2015), compared to an estimated generation of 6.0 mio t (Baldé et al., 2015). In the course of the last three years (2012–2015) 12 recycling facilities situated along China's East coast have been visited by the authors. 11 of these facilities mainly dismantle WEEE, some have subsequent treatment steps. Only one of these facilities focusses explicitly on the treatment of material from dismantling (cables and PCBs).

Concerning the range of product types processed, 7 from the 11 dismantling facilities cover all 5 product types under regulation (CRT-TV sets, refrigerators, air condition, PCs and washing machines), while the rest has capacity for some of the products. Two recyclers have established dismantling lines for additional products, not covered by the regulations (LCD screens, toner cartridges). After dismantling, nearly all plants provide for some treatment of PCBs and plastics. The dismantling process is undertaken manually, typically with the aid of conveyor belts, working stations with tools and boxes or shafts for the output materials from dismantling.

In regard to the CRT-TV sets the following steps are applied: After opening the housings and separating housing materials, metal frames, PCB and cables, the glass body of the screen is split

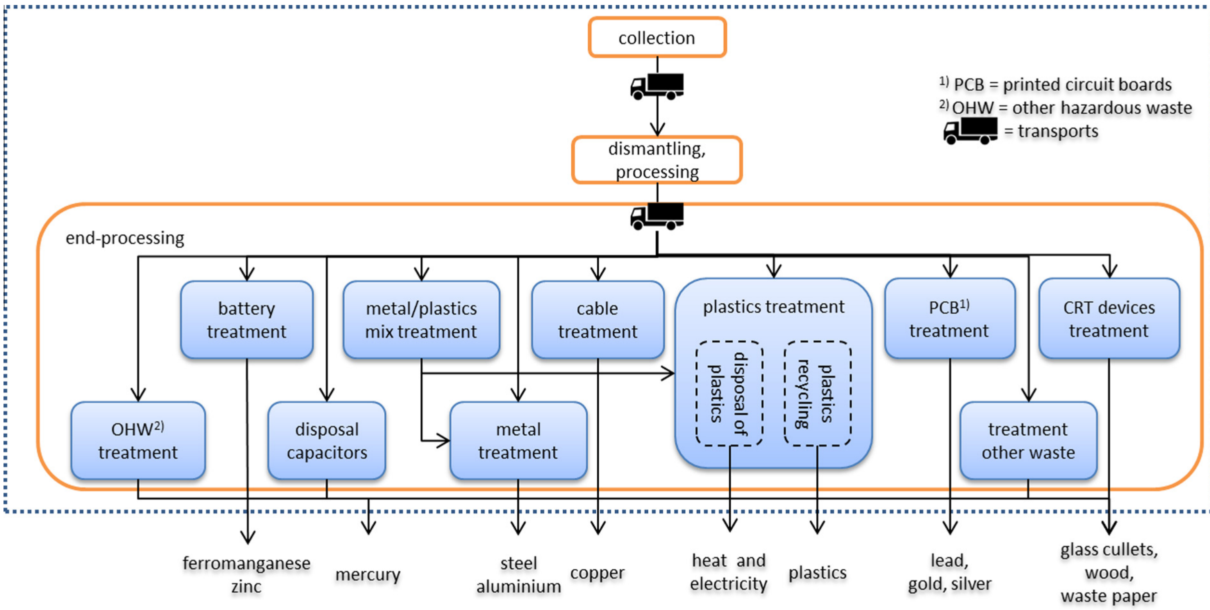

Fig. 3. WEEE treatment technologies in Europe (schematically, based on Wäger et al. (2011)).

into front and cone glass, in most cases with hot wire technology, in one case laser cutting technology was applied. Then fluorescent powder is sucked off. In most cases refrigerators and air condition undergo a two stage treatment, where in the first stage the coolant is extracted and in the second stage the body of the refrigerator is fragmentised in a closed system. In one case, only the second step is applied, i.e. refrigerators directly go into the shredder.

PCs as well as washing machines are dismantled manually. LCD screens (one plant only) are dismantled in an (under-)pressurised cabin, to avoid accidental release of Hg from potentially broken backlight fluorescent lamps, and workers in the cabin wear appropriate safety equipment. After dismantling, the LCD panel is crushed in an enclosed machine. Toner cartridges (one plant only) are fragmentised mechanically, followed by a cleaning step, where the toner dust is separated. The ferrous metals, non-ferrous metals and plastics are separated.

Processing of PCBs takes place in 8 of the facilities (7 dismantling facilities, one specialist on cables and PCBs) this treatment is done mechanically (fragmentising und separating materials) with the aim to recover copper. Three recyclers reported, to send the PCBs to mechanical recycling at a specialised plant. Only one recycler operated a hydro- und pyro-metallurgic facility, which recovered, in addition to copper, gold and silver. The processes applied are stripping, electrolysis and a refinement of gold through a melting process.

Plastics from dismantling are partly sorted into material types and partly fragmentised to reduce the volume for transport to specialised plastic recyclers. No separation of plastics with brominated flame retardants was observed in the visited plants. Some WEEE recyclers use plastics from dismantling directly for the production of wood plastic compounds.

Fig. 4 shows the input to recycling facilities in 5 provinces (Beijing, Hubei, Zhejiang, Jiangsu and Guangdong), where the above mentioned recycling plants are located and which were visited as part of the REWIN project (Zhang et al., 2015). The figure shows the quick increase in treatment capacities from 2010 to 2014 and secondly, the large proportion of TV-sets (80–90% of the mass input) compared to other types of appliances. It is obvious that TV-sets are less attractive for informal WEEE recyclers, compared to product types like PCs, refrigerators or air conditioners as the latter have a higher share of ferrous, non-ferrous or – for PCs – precious metals.

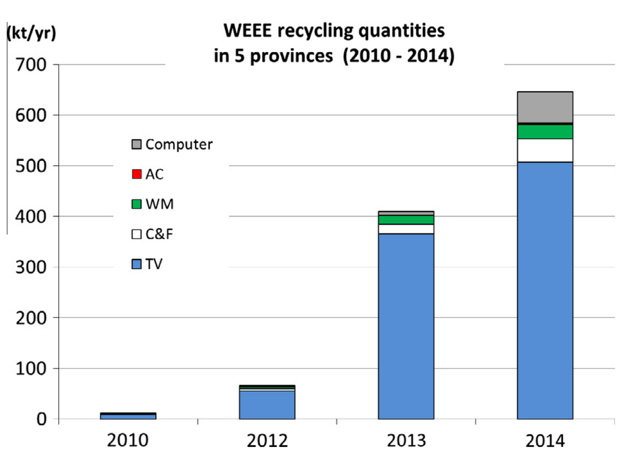

Fig. 4. WEEE recycling quantities in five Chinese provinces 2010–2014. *Source:* Zhang et al. (2015).

4. System setup

4.1. System setup in Europe

European WEEE compliance schemes in EU member states aim to implement the Extended Producer Responsibility (EPR) as guiding principle in EU legislation (Sander et al., 2007). EPR principle has been transposed to European legislation for end-of-life management of vehicles, batteries, packaging material and WEEE. As defined in the WEEE directive (2002/96/EC), the aim is at "*encouraging the design and production of electrical and electronic equipment which take into full account and facilitate their repair, possible upgrading, reuse, disassembly and recycling*" whereby "*each producer should be responsible for financing the management of the waste from his own products.*"

Within the EU, the scope of physical and financial responsibility of producers as well as of other relevant stakeholders, such as local authorities and retailers, varies widely. The compilation of 12

Table 5
Duties and rights of stakeholders within WEEE management systems of 12 selected EU countries. *Source:* own compilation based on Defillet et al. (2013), Sander et al. (2007) and Cahill et al. (2010).

Stakeholder group	Duties and rights	BE	NL	FR	GB	DE	AT
Producers/PCS	Obligation for setup of own collection points	No	No	No	No	No	Yes
	Obligation for provision of collection containers	Yes	No	Yes	No	Yes	No
	Funding collection (LA and retailer)	Yes	Yes	Yes	No	No	Yes
	Funding door-to-door collection (LA and retailer)	Yes	Yes	Partly	Yes	No	No
	Funding sorting and transhipment	Yes	Yes	Yes	No	No	N/A
	Funding treatment and bulk transport	Yes	Yes	Yes	Yes	Yes	Yes
Local authorities (LA)	Obligation to WEEE take-back	Yes	Yes	No	No	Yes	Yes
	Optional own WEEE management by direct trading	No	No	No	No	Yes	Yes
Retailers	Obligation to WEEE take-back	Yes	Yes	Yes	Yes	No	Yes
Recyclers	Optional direct trading of WEEE	No	No	No	No	Yes	Yes
	Obligation to downstream reporting of WEEE streams	Yes	Yes	Yes	No	No	No
Coordinating body/clearing house	Registering WEEE put on market	–	N/A	N/A	Yes	Yes	Yes
	Managing take-back of WEEE on request	–	N/A	N/A	No	Yes	Yes
	Joint communication on WEEE collection	–	N/A	N/A	No	N/A	Yes

selected EU countries in Table 5 shows that few producer compliance schemes (PCS) finance the whole chain from WEEE collection to treatment (e.g. Belgium, France, The Netherlands, Finland), while in other countries financial support for the expensive collection and sorting activities is not provided. Physical responsibility can be taken over by running own collection points (e.g. Austria, Bulgaria) or by providing collection containers, such as in Belgium, France and Germany.

Furthermore, the extent to which local authorities and retailers are involved in WEEE collection and management is guided by different approaches. Local authorities, as typical provider of a dense waste collection network in many EU countries, may be obligated to collect WEEE or, alternatively, opt to join an existing producer compliance scheme in return for compensation fees for collection activities. Germany represents a special case, where local authorities are obligated to collect WEEE from households without financial compensation, but in turn are allowed to independently trade collected WEEE with other parties (Defillet et al., 2013). Retail can be involved in WEEE collection in return for compensation or with a sole legal obligation on a 1:1 basis. In the United Kingdom, retailers have to take back WEEE on a 1:1 basis, or have to pay a compensation fee to municipalities taking over the WEEE collection. In Germany, retailers are not obliged to take back old appliances, but can do so on a voluntary basis (Defillet et al., 2013).

The market structure of WEEE management systems within EU countries can be divided into monopolistic systems with a single compliance system and competitive compliance systems, for instance with up to 39 parallel producer compliance schemes in UK (Dempsey, 2012). Rationales for or against monopolistic and competitive systems are partly based on macroeconomic arguments, such as market failure due to monopolistic behaviour leading to excessive prices (BIS, 2013), partly on objections against high transaction costs due costly administration and control of competitive systems due to a high number of parallel compliance schemes (Defillet et al., 2013). From a historical perspective, most existing monopolistic systems have been introduced in the late 1990ies, thus well before the introduction of the WEEE directive 2002/96/EC, and thus have served as successful role models for later EPR systems.

While the stakeholder structure in monopolistic systems is typically dominated by a central producer compliance scheme, managing all logistic, treatment and monitoring tasks on a non-profit basis, the coordinating bodies in competitive systems may cover a variety of tasks. Key tasks are the registration of electric and electronic equipment put onto the market, to avoid free riders of the system, as well as allocation of licence fees between the competing systems and stakeholders engaged in WEEE collection and treatment. Additional tasks may cover joint communication activities in order to push nationwide WEEE collection rates, monitoring and auditing tasks and the management of WEEE take-back for collection volumes not covered by contracts with producer compliance schemes (see Table 5).

Monitoring recycling plants, waste streams and financing is typically seen as a task of state authorities, taken by the relevant ministries or state agencies, partly also outsourced to private entities.

4.2. System setup in China

In contrast to the European approach of EPR, which puts high emphasis on non-state actors, the WEEE system in China aims at developing and strengthening the formal, mostly state controlled recycling of WEEE. The inclusion of private players is solely done via a product tax for producers on five defined types of appliances (TV-sets, refrigerators, air condition, washing machines, and computers). These funds are used to subsidise formal recyclers; different to the previous OfN programme only recyclers receive these subsidies, whereas no financial support is foreseen for the collection of WEEE. Currently, municipalities or retailers do not have a designated role in WEEE collection; it is the role of recyclers to acquire material for recycling through their own collection schemes or from traders. This also implies that informally collected WEEE can be re-diverted to formal recycling plants. Monitoring of waste streams and treatment processes is a task of local environmental agencies.

Concerning stakeholders, WEEE management in China is driven by six government agencies, which are assigned different responsibilities in regard to the treatment of materials and the management of recycling enterprises (see Table 6). Similar to other areas of waste management, agency roles and responsibilities are sometimes overlapping – e.g. NDRC and MOF are both responsible for the development of financing schemes – and in most respects strongly interdependent and aligned: the development of waste collection systems by the MOC depends on the designation of appliances to be recycled (NDRC), on the financial allocations to transportation, collection and recycling (MOF & NDRC), as well as on the licensing and approval of companies (MEP), where collected WEEE is to be recycled. Literature sources on this topic as well as observational evidence by the authors revealed that this agency structure often impedes effective and efficient WEEE management: At the central level, different ministerial departments struggle for influence on policy making (Chung and Zhang, 2011), which in turn

Table 6
Government agencies responsible for WEEE management (adapted from Wang et al., 2013 and Chen et al., 2010).

National agency	Roles and responsibilities
National Development and Reform Commission (NDRC)	Plan pilot projects Define WEEE categories for management Develop financing scheme WEEE management Define responsibilities for stakeholders
Ministry of Environmental Protection (MEP), Department of Pollution Control (DPC)	Establish WEEE treatment standards Manage licensing system for recyclers Monitor and evaluation of recyclers' environmental performance Establish list of products/waste for import and export Study best treatment technologies
Ministry of Industry and Information Technology (MIIT)	Managing the EEE manufacturing industry Encourage eco-design and regulate of toxics used in EEE Define responsibilities for and communicate with EEE producers
Ministry of Commerce (MOC)	Establish and manage e-waste collection channels and systems
Ministry of Finance (MOF)	Define responsibilities and communicate with EEE producers and OEMs Define and manage subsidies for logistics, collection and recycling of WEEE
General Administration of Customs (GAC)	Registering of EEE import and export figures Monitoring of illegal WEEE imports at customs checkpoints

has created a piecemeal dispersion of responsibilities to each department. At a local level, insufficient man-power allocation at the executing offices, such as the EPBs, impedes effective execution (Chung and Lo 2008; Zhong 2010; personal interviews with the staff of the Beijing EPB, September 2014) and data management (Chung and Poon, 1998; personal communication with a staff member of the Chinese National Solid Waste Management Centre affiliated with the MEP, 17.4.2013).

5. Results and discussion

Today, the level of equipment with electrical and electronic devices (amount and type) within households in urban China and the European Union are similar, which means that medium term we can expect similar WEEE generation rates. However, there are large differences between Europe and China, when it comes to the collection of WEEE: Throughout the EU a network of formal collection has been well established primarily through municipalities, partly through collection at retailers, at reuse centres and though scrap dealers. These collection schemes to date recover 3.2 mio. t of a total of 9 mio. t generated (Baldé et al., 2015) in the EU. More efforts are need in the coming years to achieve higher collection rates for small WEEE, e.g. through more convenient kerbside collection, additional containers in public places, at retailers, and public awareness campaigns supporting collection. In China, WEEE collection is dominated by informal structures. Only in a few cases are recycler or producer operated formal collection schemes in place, but these systems struggle with high costs of collection and a lack of subsidises through the WEEE system. A second reason for the dominance of the informal system is the fact that informal collectors offer convenient home collection and additional payment for obsolete appliances; a service level which is difficult to match by other collection schemes. In 2014, 1.5 mio t of WEEE from a generated total of 6.0 mio t has been recycled in formal treatment plants.

When comparing the Chinese recycling technologies to EU standards, there are no significant differences in dismantling technology for CRT-TV sets, PCs, refrigerators and air conditioners. Washing machines are rarely dismantled in Europe, but instead treated in shredders after removal of capacitors while in China this is a common treatment. The biggest difference is in the treatment of PCBs. He and Xu (2014) describe that in China PCBs treatment takes place mainly by means of density based separation (wet crushing and wet separation) or dry mechanical processing (scraping, corona separator, cyclones). He and Xu note that metal concentrates from these processes can be refined by smelting processes, however it is questionable if this actually takes place. Further non-metal materials from PCB are often processed into Wood Plastic Compounds. The authors see this recycling path worth further, more detailed analysis, considering the level of flame retardants in plastics from PCBs and the typical use of Wood Plastic Compounds as cover material for terraces, pathways, etc. with its exposure to rain and wind. With mechanical treatment of PCBs only the copper content can be recovered, while precious metals are lost. He and Xu (2014) note a recovery metal rate of 60–70% in density separation (crushing and water separation), while the rest is lost.

For the European WEEE systems, EPR is the guiding principle, where producers are responsible for the end-of-life management of their products. The national implementation of the European WEEE Directive shows, that not in all member states producers finance the full chain from collection to recycling of WEEE. Furthermore, the involvement of local authorities and retail in collection is not uniform across members of the European Union. Monopolistic and competitive compliance schemes exist in parallel, both with specific advantages and disadvantages. Monitoring is seen as a task of state authorities, and partly outsourced to private bodies.

The main target of the Chinese WEEE system is to strengthen formal recycling processes. Producers are only indirectly involved, being obliged to pay a tax on products brought onto the domestic market, but there is no obligation for producers to fund WEEE collection. Municipalities and retailers do not have a designated role in WEEE collection. Funding of WEEE recycling is organised by state authorities, therefore no competing systems are in place. Monitoring is seen as a task of local (provincial) authorities.

While in EU WEEE collection is tightly linked to established municipal waste collection schemes, familiar to local population, the main limitations lays in the collection of small WEEE, with devices being disposed of together with household waste and or via uncontrolled trading. In contrast, WEEE collection in China relies on informal collection systems, which are most convenient for citizens; the main limitation is, beside poor working conditions of informal collectors, the value orientation, meaning that this way of collection does not target end-of-life products like fluorescent lamps, which are hazardous materials, but do not represent significant material value.

Concerning treatment technology, European technology has been established earlier, is more advanced and able to recover metals, including precious metals at high recycling rates. Limitations are the high costs of treatment, specifically when competing with uncontrolled export. In China today, many technologies applied are identical to European ones, e.g. dismantling, separation technology, etc. Limitations are recycling processes where hazardous materials such as flame retardants are transferred into products and missing technology for metallurgical treatment of printed circuit boards and other precious metal containing components, leading to a loss of precious metals.

Comparing the system setup, the involvement of producers in collection and recycling represents the major difference between EU and China. Both approaches have limitations to satisfactorily manage the entire waste stream, although for different reasons: losses in WEEE in the EU derive from uncontrolled export and disposal of small WEEE with household waste, while in China the informal recycling sector prevails because its practices are economically more competitive than their formal counterpart.

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Possibilities and limits of pyrolysis for recycling plastic rich waste streams rejected from phones recycling plants

B.M. Caballero *, I. de Marco, A. Adrados, A. López-Urionabarrenechea, J. Solar, N. Gastelu

Chemical and Environmental Engineering Department, School of Engineering of Bilbao, University of the Basque Country (UPV/EHU), Alda. Urquijo s/n, 48013 Bilbao, Spain

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ABSTRACT

The possibilities and limits of pyrolysis as a means of recycling plastic rich fractions derived from discarded phones have been studied. Two plastic rich samples ($\geqslant$80 wt% plastics) derived from landline and mobile phones provided by a Spanish recycling company, have been pyrolysed under N_2 in a 3.5 dm^3 reactor at 500 °C for 30 min. The landline and mobile phones yielded 58 and 54.5 wt% liquids, 16.7 and 12.6 wt% gases and 28.3 and 32.4 wt% solids respectively. The liquids were a complex mixture of organic products containing valuable chemicals (toluene, styrene, ethyl-benzene, etc.) and with high HHVs (34–38 MJ kg^{-1}). The solids were composed of metals (mainly Cu, Zn, and Al) and char ($\approx$50 wt%). The gases consisted mainly of hydrocarbons and some CO, CO_2 and H_2. The halogens (Cl, Br) of the original samples were mainly distributed between the gases and solids. The metals and char can be easily separated and the formers may be recycled, but the uses of the char will be restricted due to its Cl/Br content. The gases may provide the energy requirements of the processing plant, but HBr and HCl must be firstly eliminated. The liquids could have a potential use as energy or chemicals source, but the practical implementation of these applications will be no exempt of great problems that may become insurmountable (difficulty of economically recovering pure chemicals, contamination by volatile metals, etc.)

1. Introduction

The worldwide production of Waste of Electrical and Electronic Equipment (WEEE) has been estimated in 20–50 million tons per year (Wang and Xu, 2014; Ongondo et al., 2011). In Europe around 10 million tons WEEE are generated each year (ITRE Committee, 2014) and it is expected that by 2020 the amount generated in the EU-28 will reach a total annual tonnage of 12.3 million (Muhammad et al., 2015; Alston et al., 2011; Ortuño et al., 2014). In 2012 approximately 35% of the WEEE generated in the EU ($\approx$3.5 million tons) were collected and appropriately managed, having increased this percentage at about 7% per year from 2007 to 2012 (Eurostat, 2015; Hense et al., 2015).

Until recently most of the discarded electrical and electronic equipment (televisions, computers, telephones, etc.) were landfilled or incinerated and both of these alternatives can cause serious damage to the environment and have adverse effects on human health, due to the hazardous products contained in WEEE such as, Pb, Cd, Hg, PVC, and halogenated flame retardants.

In 2002 the European Commission launched the so-called WEEE Directive 2002/96/EC which has recently been replaced by the new WEEE Directive 2012/19/EU that introduces an increase in the collection targets of this waste. From 2016 the annual collection target is defined as the ratio between the collected amount and the average weight of EEE (electrical and electronic equipment) placed on the market in the three preceding years. The collection target is set at 45% in 2016 and will rise to 65% in 2019, a quota that will be very difficult to reach since WEEE is a very complex mixture of many and very different materials (metals, different plastics, glass, rubbers, etc.). As a general rule, WEEE contains about 40% of metals, 30% of plastics and 30% of refractory oxides and the typical composition of metal scrap is copper (20%), iron (8%), tin (4%), nickel (2%), lead (2%), aluminum (2%), zinc (1%) and small percentages of precious metals as silver, gold and palladium (Gramatyka et al., 2007; Yang et al., 2013; Salbidegoitia et al., 2015). With respect to the plastic fraction the most common constituents are acrylonitrile–butadiene–styrene copolymer (ABS), high impact polystyrene (HIPS), polycarbonate (PC), blends of ABS and PC, polypropylene (PP), polyphenylene (PPE) and HIPS, polyvinyl chloride (PVC), polyamide (PA) and polystyrene (PS) (Yang et al., 2013; Alston et al., 2011).

Information and Telecommunication Technologies (ITT) equipment is one of the most predominant WEEE in the EU ($\approx$16–18%

* Corresponding author.
E-mail address: blancamaria.caballero@ehu.es (B.M. Caballero).

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of collected WEEE) (Eurostat, 2015; Ongondo et al., 2011). This category includes computers, printers, speakers, web cameras, etc., and also phones which are the subject of this study.

The 2014 statistics showed that the number of landline telephones subscriptions in the world was about 1100 millions, which means about 15 subscriptions per 100 inhabitants (ITU, 2015). In Europe fixed phone landline subscriptions decreased from 42.8 (per 100 inhabitants) to 38.3 (per 100 inhabitants) from 2010 to 2014 (ITU, 2015), and will continue to decrease and to give rise to phone waste, due to the advantages of mobile phones and Internet-based alternatives; therefore landline phones are now discarded more and more frequently.

On the other hand the use of mobile phones is at present outrageously widespread. There are around 7000 million mobile phone subscriptions in the world, which means around 96 per 100 inhabitants (ITU, 2015). Mobile phones are continuously being replaced by newer ones with higher performance or more novel designs. The average life span of a smart phone is less than 2 years.

It is evident that phones, either landline or mobile ones, currently constitute an important and increasing waste stream, which has the advantage of being easily and selectively collected. For this reason this paper has been centred in the study of an alternative for recycling waste streams coming from discarded landline and mobile phones.

1.1. Waste of electrical and electronic equipment management

Recycling companies are almost exclusively focussed in recovering metals from WEEE. Typical WEEE recycling methods include a first step in which hazardous or valuable components (batteries, printed circuit boards, casing, external cables, etc.) are separated, and a second mechanical or metallurgical process to upgrade the content of desirable materials and obtain marketable output streams. Mechanical processes include shredding or crushing and then sorting based on the size, shape, density, and electrical and magnetic characteristics (magnetic separation, Eddy current separation and gravity separation), while metallurgical processes involve either melting (pyrometallurgical processes) or dissolving (hydrometallurgical processes) the metals (Tsydenova and Bengtsson, 2011).

Concerning discarded phones, WEEE recycling companies recover a great part of the metals from landline phones (base, card phone and wire included) and mobile telephones (including terminal transmitter/receiver, battery and accessories such as transformer/battery charger, and cover) by shredding them and then subjecting them to magnetic and Eddy current fields. After this process a waste stream with a very high proportion of different plastics (PC, ABS, PVC, etc.) mixed with some metals and other materials (glass, fillers ...) is left. In order to achieve the targets of the recently renewed WEEE Directive, these types of rejected waste streams have to be, as much as possible, recycled. However, the plastics in these streams are very much intermingled and also have some remaining metals or other materials embedded in them. Therefore because of their complexity and heterogeneity, and also due to their content of hazardous substances, the separation and mechanical recycling of the individual components of these waste streams is not technically and/or economically feasible.

For this reason this paper has focussed on the study of the possibilities and limits of pyrolysis as an alternative route for recycling plastic rich fractions rejected from phone recycling industrial plants. In the pyrolysis process the organic volatile matter of the material (plastics, rubbers, etc.) is decomposed to gases and liquids. The inorganic components (metals, fillers, glass, etc.) remain almost unaltered during the process, and consequently their valuable components can be recovered and reused. The pyrolysis process is therefore especially appropriate for complex waste, which contain many different plastics mixed with other materials, as is the case of the plastic rich waste streams coming from landline and mobile phones considered in this study.

There are several references in the literature dealing with the pyrolysis of plastic fractions contained in WEEEs. Some authors have investigated plastics from cathode ray tubes (televisions and computer monitors), refrigeration and freezers equipment (Hall and Williams, 2007a; Muhammad et al., 2015), and from computer bodies and monitor cases (Hall and Williams, 2006). Other studies have focused on printed circuit boards from waste computers, televisions and mobile phones (Hall and Williams, 2007b). Mixtures of WEEE plastics have been studied by Hall and Williams (2007a), Alston et al. (2011) and Acomb et al. (2013). However there are no studies devoted specifically to the pyrolysis of plastic rich streams derived from mobile phones and landline phones. There do are some references dealing with pyrolysis of certain components of mobile phones, in particular with printed circuit boards or mixtures of printed circuit boards and cases from mobile phones (Hall and Williams, 2007b; Ortuño et al., 2014; Moltó et al., 2009, Moltó et al., 2011), but only the latter of this references devotes some attention specifically to a plastic rich fraction (the casing) of the mobile phones.

In 2008 the authors published a preliminary study about the pyrolysis of different electrical and electronic WEEEs (de Marco et al., 2008), in which the pyrolysis yields obtained and a light and incomplete characterization of pyrolysis products were included. The main conclusions were that the yields and characteristics of the pyrolysis products depended very much on the type of WEEE pyrolyzed, and that all three products may find useful applications. Many scientific papers dealing with pyrolysis of plastic waste claim the goodness of pyrolysis products without considering the limitations that arise when it comes to the applications of these products in practice especially if the plastics come from WEEE. In this paper a thorough characterization of the solids, liquids and gases obtained by pyrolysis of plastics-rich fractions derived from phones is presented, which enables the ability to assess the possibilities and limits of pyrolysis as a means of recycling plastics-rich waste streams rejected from phone recycling plants.

2. Material and methods

2.1. Origin of the samples

The landline and mobile telephone samples pyrolysed were provided by a Spanish recycling company devoted to recovering metals from WEEE. Such samples are the waste streams that are obtained in the mentioned company after grinding the landline and mobile telephones (base, card phone and wire included) once the magnetic parts have been magnetically removed. Both samples were provided with a particle size of ≈ 2 cm and were pyrolysed as they were received. Homogeneous and representative 100 g samples were separated for the pyrolysis experiments by successively dividing the original samples and subsamples into fourths. Fig. 1(a) y (b) shows a picture of the landline phone and mobile phone samples. Two of the 100 g samples (one of each type of phone) were finally ground to a particle size <0.5 mm, which is the appropriate size for the different analytical techniques used to characterize the samples, which will be described in Section 2.3.

2.2. Pyrolysis experiments

The pyrolysis experiments were carried out at 500 °C in a nitrogen atmosphere, using an unstirred stainless steel 3.5 dm^3 reactor in a semi-batch operation at atmospheric pressure.

(a)

(b)

Fig. 1. Samples (as was received and pyrolysed): (a) landline phones; and (b) mobile phones.

Previous studies carried out by the authors (Laresgoiti et al., 2004; de Marco et al., 2007; López-Urionabarrenechea et al., 2011a) with other polymeric waste (e.g. sheet moulding compound (SMC) of polyester and fibreglass, used tyres, automobile shredder residues, municipal plastic waste) indicated that in the mentioned installation, 500 °C was the optimum temperature for the treatment of polymeric waste by pyrolysis, since at lower temperatures complete decomposition of the organic matter was not achieved, and at higher temperatures an increase in gas yield was produced, which was counterbalanced by a detrimental effect on the liquid yield. Therefore 500 °C was chosen as the process temperature for the samples tested in this study.

In a typical run, 100 g of the sample are placed into the reactor, which is then sealed. Nitrogen is passed through at a rate of 1 $dm^3\ min^{-1}$ and the system is heated at a rate of 15 °C min^{-1} to 500 °C, and maintained there for 30 min. It has been proved by the authors that when plastic waste is pyrolyzed at 500 °C in the mentioned installation, after 30 min no more pyrolysis products evolve from the reactor (López-Urionabarrenechea et al., 2011a). The whole process is controlled by a computer. The thermocouple which measures and controls the heating system is placed in the middle of the reactor chamber. This implies that although the reactor is an unstirred one and plastics have relatively low thermal conductivities, it is guaranteed that the whole sample reaches at least the preset temperature. Concerning N_2 carrier gas distribution, there is a diffusion plate inside the reaction chamber (at the bottom of the reactor) which distributes nitrogen all around the reaction chamber.

The vapours leaving the reactor flow to a series of water cooled gas–liquid separators where the liquids are condensed and collected. The uncondensed products are passed through an activated carbon column and the total gas is collected as a whole in Tedlar plastic bags, to be tested by gas chromatography. The flow sheet of the experimental setup is presented in Fig. 2.

The solid and liquid pyrolysis yields were determined in each experiment by weighing the amount of each obtained, and calculating the corresponding percentage with respect to the initial sample weight, whereas the gas yields were calculated by difference.

2.3. Analytical techniques

Both the raw materials and the solid and liquid pyrolysis products were thoroughly characterized using the following analytical techniques. Inductive Coupled Plasma (ICP) analysis was used to determine the metals contained in the initial phone samples. Thermogravimetric analysis (TGA) (15 °C min^{-1}, 500 °C, in O_2 atmosphere until a constant weight is reached) was used in order to determine the inorganic content of the solid samples, both the initial samples and the solid pyrolysis products. A CHN automatic analyser, which complies with the ASTM D5373 standard for elemental analysis of fuels, was used for determining C, H and N contents of both the solid and liquid samples. The Br and Cl contents of the solids and liquids were determined by using method 5050 from the United State Environmental Protection Agency (EPA) for the determination of total chloride in solid waste and liquid fuels. The method consists of oxidizing the sample in a calorimeter and collecting the gases generated (HCl and HBr) in a 0.25 M NaOH basic solution placed with the sample inside the calorimetric bomb. The solution is then analyzed by ionic chromatography to determine the chloride and bromide contents.

The higher heating value (HHV) of both the solid and the liquid samples was determined with an automatic calorimetric bomb complying with the ASTM D3286 standard.

The pyrolysis liquids were analyzed by gas chromatography coupled with mass spectrometry detector (GC–MS). Identification of the constituents was based on comparison of the retention times with those of the calibration samples and on computer matching against a commercial library of mass spectra and MS literature data. The library-matched species which exhibited a degree of match lower than 90% were classified as "Not identified".

Concerning pyrolysis gases, they were analyzed by means of a gas chromatograph coupled with two independent detectors: thermal conductivity and flame ionization (GC–TCD/FID). The HHV of the gases was theoretically calculated based on their composition and the HHV of the individual components.

3. Results and discussion

3.1. Composition of the landline phone and mobile phone samples

The characteristics of the landline and mobile phone samples are presented in Table 1. It can be seen that both phones have a rather low inorganic matter content (13.9 and 16.3 wt% respectively), which indicates that they are mainly composed of plastics; this justifies their high HHV (30.8 and 26.4 MJ kg^{-1}). The H/C atomic ratios are quite low (≈1) which is indicative of the aromatic structure of the polymers contained in both samples.

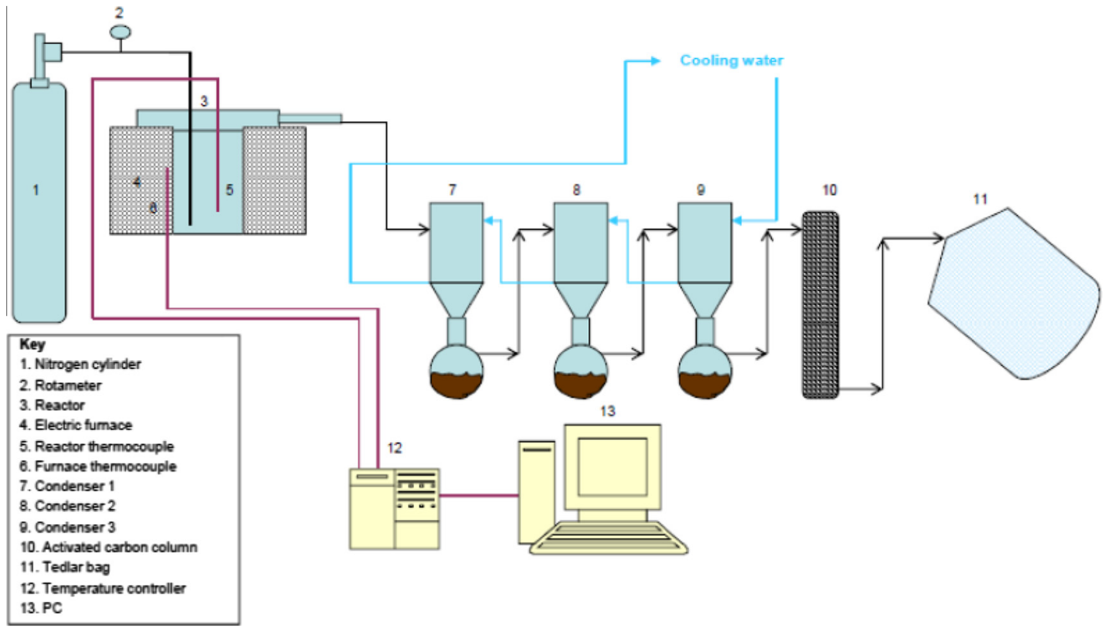

Fig. 2. Flow sheet of the lab-scale installation used for the pyrolysis experiments.

Table 1
Composition of the samples pyrolysed (wt%).

	Landline phones	Mobile phones
Inorganic matter[a]	13.9	16.3
C	74.0	69.8
H	6.5	5.8
N	3.9	1.8
Cl	2.1	0.13
Br	0.6	<0.1
Others[b]	–	6.17
H/C atomic ratio	1.05	1.0
HHV (MJ kg^{-1})	30.8	26.4

[a] Determined by thermogravimetric analysis at 500 °C in O_2 atmosphere.
[b] Determined by difference.

It has to be mentioned that Table 1 shows that for mobile phones the sum of the percentages of the elements plus the inorganic matter is greater than 100; this would indicate that the percentage of "others" and consequently of oxygen is zero, which is rather improbable. This anomalous result must be attributed to the fact that the samples are very heterogeneous, and three different and very small samples (≈1 g) have to be used for the CHN, halogens and TGA analyses, which if do not have the very same content of inorganic matter, may lead to the lack of inconsistency of the results.

In previous studies carried out by the authors with similar phone samples coming from the same Spanish recycling company (de Marco et al., 2008) it was concluded by means of FTIR analysis that the predominant plastic component of landline phones was ABS (acrylonitrile–butadiene–styrene thermopolymer), while the mobile phones were composed not only of ABS but also of polycarbonate (PC). These conclusions justify the high N content of both samples and the fact that mobile phones contain less N and more "others" (most probably oxygen of the polycarbonate (—O—CO—O) structure) than landline phones. Mobile phones have very much technologically evolved in the last years, mainly the software and electronics. However the samples used in this study were a plastic rich fraction derived from phones, and the plastics used for mobile phone casings and accessories have not much changed. As a matter of fact the elemental analyses of the samples used by the authors in the study of 2008 and of those used in the present study are quite similar. Moreover Moltó et al., 2011, reported for mobile phone casing a CHN composition which, in an O and ash free basis, is quite similar to that of the samples of this study, which corroborates that plastics used in mobile phones have not much evolved in the last years.

Concerning halogens content Table 1 shows that the landline phone sample contained a high proportion of Cl and Br, which most probably comes from fire retardants that are frequently used in electrical and electronic equipments (EEE) and also from halogenated polymers, such as PVC, which are used in components of these telephones such as in wire insulation. With respect to mobile phones the amount of Cl and Br is much lower (0.13 and <0.1 wt% respectively). This difference may be attributed to the fact that the discarded landline phones have probably been fabricated many years ago, while the discarded mobile phones have been fabricated just a few years ago, when the RoHS (Restriction of hazardous substances) Directive 2002/95/CE, which restricts the use of certain hazardous substances in EEE, had already been adopted by the EU. In fact at present many mobile phone manufacturers are promoting the reduction of halogens in mobile devices and they guarantee Br contents ⩽900 ppm, Cl contents ⩽900 ppm and Br + Cl contents ⩽1500 ppm (Sony Ericsson, 2009).

Inductive Coupled Plasma (ICP) analysis was used to determinate the metals contained in the initial samples. In the ICP analysis of the samples, 52 elements were identified in landline phones and 53 in the mobile phones, many of which were in very low proportions. For the sake of reduction only the predominant metals (>0.1 wt%) have been included in Table 2. It can be seen that the predominant metals in both samples are Cu, followed by Al and Fe in landline phones and by Zn, Sn and Ni in mobile phones. Table 2 also shows that the total metal content of both samples is rather low, which is in agreement with the composition data (% inorganic content) presented in Table 1.

3.2. Results of the pyrolysis experiments

The solid, liquid and gas yields (weight%) obtained in the pyrolysis runs carried out with both phone samples at 500 °C are presented in Table 3. The results presented are the mean value of

Table 2
ICP analysis of the samples pyrolysed (wt%).

	Landline phones	Mobile phones
Cu	6.19	8.05
Al	1.61	0.51
Fe	1.61	<0.10
Zn	0.56	1.97
Pb	0.35	0.48
Ca	0.33	0.26
Sn	0.22	0.59
Ni	0.19	0.59
Sb	0.14	<0.10
Mn	0.12	<0.10
V	0.10	<0.10
Other identified inorganics	0.26	0.84
Organics [a]	88.32	86.71

[a] Determined by difference (may include some non identified inorganics by ICP).

Table 3
Pyrolysis yields (wt%) (mean value).

	Solid[a]	Liquid	Gas[b]
Landline phones	28.3	58.0	16.7
Mobile phones	32.9	54.5	12.6

[a] Solid yield (char + inorganics).
[b] Determined by difference.

the data obtained in at least two equivalent experiments which differed by less than four points. It has to be born in mind that due to the heterogeneity of the sample it is rather difficult to obtain more precise results.

3.2.1. Pyrolysis solids

Table 3 shows that the solid yields obtained with landline and mobile phones (28.3 and 32.9 wt% respectively) were higher than those expected based on the inorganic content of the samples (13.9 and 16.3 wt% respectively) (Table 1). The solid products obtained were completely black, while the original samples were of a variety of colors, as can be seen in Fig. 1(a) and (b). The black product, which was mixed with the metals, is char a carbonaceous material formed during the pyrolysis process, due to secondary repolymerisation reactions among the polymer derived products. It is a well known fact, that has been reported by many authors (e.g. López-Urionabarrenechea et al., 2010; Adrados et al., 2012; Bhaskar et al., 2004), that char is usually formed in many pyrolysis processes. The char forming tendency depends on the chemical structure of the polymer and increases with the aromaticity of the polymer, with groups capable of reacting with hydrogen atoms of the aromatic nuclei such as —OH, =O, and with halogen atoms (Van Krevelen and Te Nijenhuis, 2009). The phone derived plastics are mainly composed of ABS and PC (both contain aromatic rings) and also contain halogens, which justifies the significant amount of char (≈50 wt%) obtained in the pyrolysis solids.

Table 4 shows the composition and HHV of the pyrolysis solids obtained with both telephone samples. The inorganic content measured by TGA included in Table 4 (61.1 and 62.9 wt% for landline and mobile phones respectively) is somewhat higher than the theoretical values (49.1 and 49.5% respectively) calculated based on the solid pyrolysis yields (28.3 and 32.9 wt%) and the inorganic matter content (13.9 and 16.3 wt%) of the initial samples. This may be attributed to the heterogeneity of the sample, to the fact that different and very small samples (≈1 g) are used for each of the analyses (one for TGA, one for CHN and one for Cl/Br analysis), and to the fact that some of the char formed in the process remained stuck to the wall of the autoclave and pipes, and so char depleted samples are picked up from the bottom of the autoclave for the analyses.

Table 4 shows that the predominant element of the pyrolysis solids is carbon. It amounts to 34.6 and 32.0 wt% respectively, but in an inorganic matter free basis it would rise to 81 and 93.3 wt% respectively. So, as was to be expected the metal solid free product (char) is mainly composed of carbon. The metals in the solids can be rather easily separated from the char and may then be recycled.

Table 4 shows that the HHV of the whole pyrolysis solids is quite moderate (15.2 and 14.5 MJ kg^{-1} respectively), but it can be calculated that the HHV of the metal free char would be very high (35.3 and 34.3 MJ kg^{-1} respectively). The phone pyrolysis char could theoretically find application as solid fuel, pigment, activated carbon, low quality carbon black, component of asphalt fabrics, etc. However it has to be born in mind that in the case of the landline phone solids, the Cl and Br content is very high, it would rise up to 5.8 and 1.8% respectively on an inorganic free basis, and this will obviously limit its potential applications, especially as solid fuel. It can be calculated that about 30% of the Cl and Br contained in the original landline phone sample is retained in the solid. Other authors (Hall and Williams, 2007a) have also reported that halogens are concentrated in WEEE pyrolysis solids; this is attributed to a halogen scrubber effect of the metals, yielding metal halides in the solids and turning out oils with rather low halogen content.

3.2.2. Pyrolysis liquids

The pyrolysis liquids, usually termed oils, were dark brown-colored, rather fluid products, which resembled petroleum fractions. The elemental composition, H/C atomic ratio and HHV of the pyrolysis liquids obtained in the pyrolysis of the two samples tested are presented in Table 5. The results show that the H/C ratio in both samples is rather close to 1, which is indicative of their aromatic/naphthenic nature. The liquids contain a significant amount of N, which comes from the acrylonitrile of the ABS, and also a significant content of "others", probably oxygen which comes from oxygenated polymers such as polycarbonate (PC). Comparing both liquids, it can be observed that the mobile phone derived liquids are poorer in N and richer in "others" (oxygen) than the landline phone liquids. This, as has been mentioned before, is due to the fact

Table 4
Composition (wt%) and HHV (MJ kg^{-1}) of the pyrolysis solids.

	Landline phones	Mobile phones
Inorganic matter[a]	61.1	62.9
C	34.6	32.0
H	1.6	1.4
N	1.5	0.9
Cl	2.5	<0.1
Br	0.8	<0.1
HHV (MJ kg^{-1})	15.2	14.5

[a] Determined by thermogravimetric analysis at 500 °C in O_2 atmosphere.

Table 5
Pyrolysis liquids elemental composition (wt%) and HHV (MJ kg^{-1}).

	Landline phones	Mobile phones
C	82.9	75.5
H	7.7	8.0
N	3.3	1.8
Cl	<0.1	<0.1
Br	<0.1	<0.1
Others[a]	6.1	14.4
H/C atomic	1.11	1.27
HHV (MJ kg^{-1})	38.3	34.4

[a] Determined by difference.

that the main component of the landline phones is ABS while mobile phones contain ABS and also PC, and the latter contains oxygen.

It is worth mentioning that the Cl and Br contents of both liquids are very low, lower than 0.1 wt% which is the accuracy level of the measurement technique which has been described in Section 2.3. Halogen content is one of the potential drawbacks usually attributed to pyrolysis liquids that come from wastes that contain halogens. But the results obtained in this study show that this problem is not so evident in phone derived pyrolysis oils, due, as has been mentioned before, to the scrubber effect of metals. Concerning the HHV of the liquids, Table 5 shows that they are very high (38.3 and 34.4 MJ kg^{-1} respectively), in the range of those of liquid fossil fuels, so an immediate and thoughtless proposal of application of telephone pyrolysis oils would be as an alternative renewable liquid fuel. However it has to be born in mind that apart of Cl and Br there may be other pollutants on the oils, such as volatile metals, as for instance mercury whose boiling point is just 357 °C and therefore will volatilize in the pyrolysis process ending up in the liquids. The mercury content of both phone samples was very low, 0.6×10^{-4} and 0.25×10^{-4} wt% for landline and mobile phones respectively, and therefore were not included in Table 2 where only >1 wt%-elements were included. However such contents are higher than those of coals, which are in the range 0.01–0.48 ppm (Park et al., 2008) and coal power plants are claimed to be at present responsible for most of the worldwide mercury emissions to the atmosphere. Therefore in case of using phone pyro-oils as fuels, strict control measurements of mercury emissions should be implemented.

The results obtained in the GC/MS analysis of the pyrolysis oils obtained from landline and mobile phones are summarized in Table 6. The corresponding GC chromatograms showing the identified compounds are presented in Fig. 3a and b. Only those compounds with a percentage quantified area greater than 1% have been included in Table 6. Under the name “Not identified” the compounds with a match quality provided by the MS search engine lower than 90% have been included all together.

Pyrolysis oils are a complex mixture of organic compounds ranging from 4 to 16 carbon atoms. Most of them are aromatic hydrocarbons which is coherent with what has been inferred from the H/C atomic ratio of the liquids (Table 5). Table 6 shows that both oils contain significant quantities of valuable chemicals such as toluene, ethyl-benzene, styrene and α-methylstyrene. The main significant difference between both oils is the higher content of phenol and phenol derivatives in the mobile phone oils, which are almost not present in the landline phone oils and that are most probably derived from the PC plastic contained in the mobile phone initial sample. Another difference is the higher content of toluene, styrene and α-methylstyrene in landline phone oils.

In order to better assess the potential applications of the pyrolysis oils, all the components identified by GC–MS (including those with area <1%), have been grouped in three categories according to their number of carbons, C5–C9, C10–C13 and >C13. Additionally total aromatics, total nitrogenated and total oxygenated compounds have been quantified. The results are presented in Table 7. Concerning the number of carbon atoms, Table 7 shows that the pyrolysis liquids from both types of telephones contain ⩾77% area of C5–C9 products, which is the carbon atom range of gasoline products. Concerning aromatics both oils have a very high content, 92.96% and 90.35% area for the landline and mobile phones liquids respectively. Therefore most of the ABS and PC contained in both original samples have been converted to aromatics. It might have been expected higher aromaticity in mobile phone oils than in landline phone oils since the main polymer in mobile phones is ABS which has less aromatic rings in its backbone chain than PC which is the main polymer together with ABS of mobile phones. However it is well known that cyclisation reactions followed by dehydrogenation of aliphatic chains frequently occur in pyrolysis processes and that this reactions are promoted by catalysts (e.g. Siddiqui et al., 2004; Aguado et al., 2007; de Marco et al., 2009). In this study it is highly probable that the metals contained in the samples (Fe, Co, Ni, Zn, etc.) have exerted some catalytic effect favouring the formation of aromatics.

Concerning total oxygenated compounds mobile phone oils have a much higher proportion than landline phone oils which, as has already been mentioned, is due to the presence of PC in the initial mobile phone sample. With respect to the total content of nitrogenated compounds it is somewhat higher in the landline phone oils, due to the greater amount of ABS contained in the initial landline phone sample.

The thorough characterization of phone derived pyrolysis oils puts forward that there is a wide variety of potential applications for these products, but that they are not exempt from difficulties in their implementation. Firstly, they can be used as renewable alternative liquid fuel replacing fossil fuels, but for this to be possible, pollutant elements (N, halogens, mercury) have to be removed or controlled prior to or after combustion. Secondly, automotive fuels (gasoline, diesel oil) may be obtained from the oils, but this requires distillation and costly upgrading operations in order to fulfill the demanding specifications of commercial automotive fuels. Finally the phone derived oils may be used as a source of valuable chemicals such as styrene, ethyl-benzene, etc. however the extraction of pure individual chemicals from such a complex mixture is an expensive and not easy task.

All of these application alternatives may be technically achieved, but for the moment not at an economic price. At present the most feasible and direct alternative for reusing pyrolysis oils would be to process them mixed with petroleum streams in oil refineries, however oil companies are not prone to accept unconventional oils especially if such oils are under suspicion for containing halogens and metals.

3.2.3. Pyrolysis gases

The composition and HHV of pyrolysis gases is presented in Table 8. It can be seen that the gases are composed of hydrocarbons (from C_1 to C_6), CO, CO_2 and hydrogen. The most striking difference between the landline and mobile phone derived gases is the CO content which is much higher in the mobile phone derived gases (32.1% compared to 6.5%) and comes from the oxygen rich PC structure. Consequently the percentage of all the other

Table 6
Main components of the pyrolysis liquids identified by GC–MS (>1% area in at least one of the phone liquid samples).

Peak number	Compound	Landline phones	Mobile phones
1	Toluene	13.17	6.23
2	Ethylbenzene	15.55	6.92
3	1-Methylethylbenzene	1.85	1.16
4	Styrene	37.77	19.06
5	α-Methylstyrene	11.65	6.71
6	Naphthalene	1.0	<1
7	Phenol	1.12	21.87
8	4-Methylphenol	–	1.87
9	Benzenebutanenitrile	5.04	4.66
10	4-Ethylphenol	–	3.12
11	4-(1-Methylethyl) phenol	<1	9.20
12	1,1′-(1,3-Propanediyl)bis benzene	1.07	1.71
13	p-Isopropenylphenol	–	5.71
14	2-Phenylnaphtalene	1.02	–
	Other identified compounds (<1% area)	4.51	3.40
	Not identified compounds	6.25	8.38

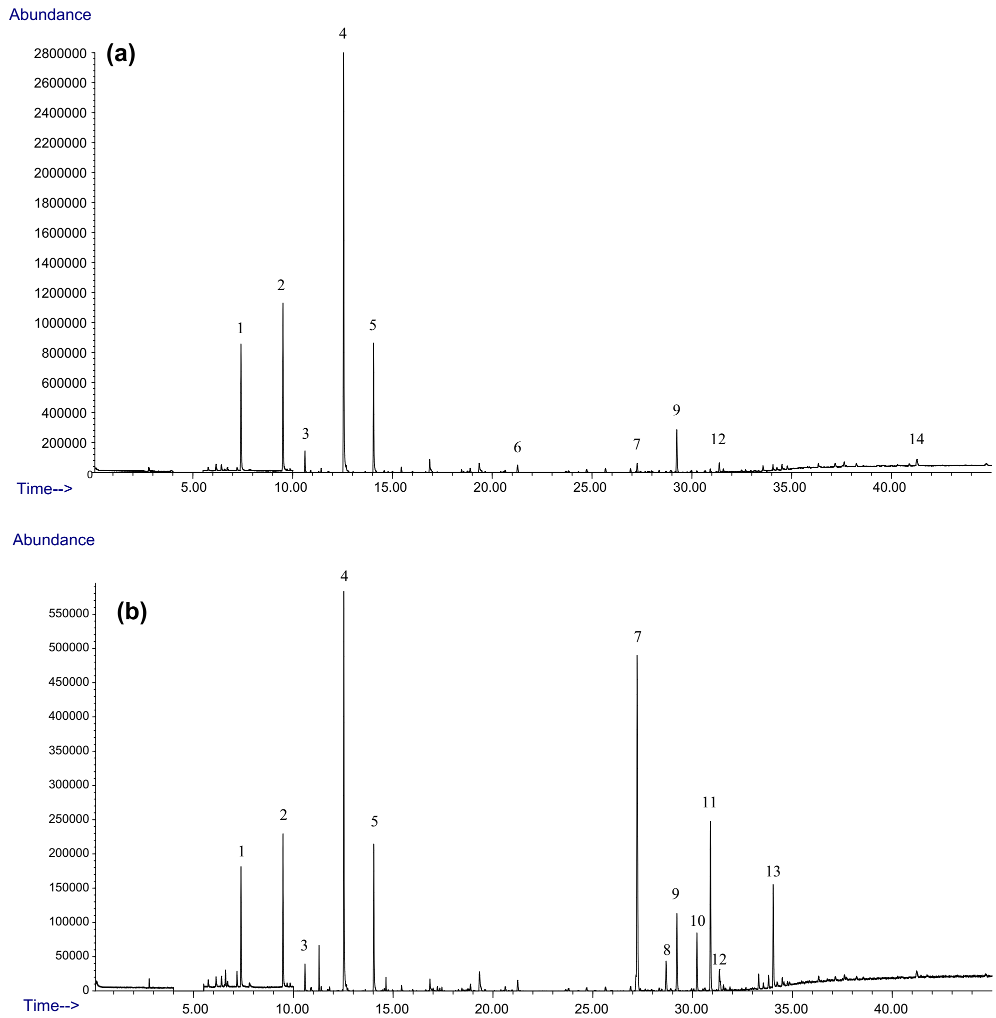

Fig. 3. GC–MS chromatogram of the pyrolysis liquids of: (a) landline phones; and (b) mobile phones.

Table 7
Fractions of interest in the pyrolysis liquids (% area).

Fraction		Landline phones	Mobile phones
C5–C9	Aromatics	82.84	76.84
	Non aromatics	0.79	1.27
	Total	83.63	78.11
C10–C13	Aromatics	7.66	13.51
	Non aromatics	–	–
	Total	7.66	13.51
>C13	Aromatics	2.46	–
	Non aromatics	–	–
	Total	2.46	–
Total aromatics		92.96	90.35
Total oxygenated compounds		1.46	43.23
Total nitrogenated compounds		7.58	5.94
Not identified		6.25	8.38

Table 8
Composition (vol.%) and HHV (MJ m^{-3} N) of the pyrolysis gases.

Component	Landline phones	Mobile phones
Hydrogen	12.2	6.8
CO_2	13.0	10.7
CO	6.5	32.1
Methane	28.7	29.1
Ethane	7.5	4.0
Ethene	12.8	7.8
C3	10.5	4.8
C4	3.5	1.6
C5	2.5	1.5
C6	2.8	1.8
HHV	46.3	33.0

components is higher in the landline phone gases. Leaving out CO and CO_2 and comparing the percentages of the other components, on a CO/CO_2 free basis, it can be seen that the CH_4 content is higher in the mobile phone gases while the C_2 to C_6 components are higher in the landline phone gases. This is logical since the mobile phone sample contains PC plastic which has no aliphatic chains, while ABS, which is the main component of the landline phone sample, has an aliphatic chain backbone, that when it is cracked may yield C_3 to C_6 hydrocarbons. Concerning the higher H_2 content of landline phone gases compared to mobile phone gases, it may be attributed on the one hand to the higher H_2 content of the original landline phone sample, and on the other hand, to the greater production of aromatization reactions (cyclisation + dehydrogenation) that take place during pyrolysis of landline phones since this sample is mainly composed of ABS that is more aliphatic than PC which is one of the main components of the mobile phone sample.

It has to be mentioned that the analytical technique used for the gas analysis (GC/TCD–FID) was not suitable for determining Cl and Br halides, however these elements have for sure evolved with the gases, since as has been previously mentioned, about 30% of the Cl and Br contained in the initial samples are retained in the pyrolysis solids, and there is a very low proportion of these elements that is transferred to the liquid; therefore the remaining 70% has to evolve with the pyrolysis gases. In some of the experiments the pyrolysis gas was forced to pass through a NaOH solution and it was proved that, effectively, Br and Cl were retained in the solution; unfortunately the results cannot be reported since no repetitiveness was achieved at that moment.

Table 8 shows that the HHV of the gases is rather high (46.3 and 33.0 MJ m^{-3} N for the landline and mobile gases respectively). It is in the range of that of natural gas (≈43 MJ m^{-3} N) for the landline phone gases and somewhat lower for the mobile phone gases. The difference in HHV between both gases is mainly due to the higher amount of CO in mobile phone gases, since the HHV of CO is much lower than that of hydrocarbons. Nevertheless the energetic content of the pyrolysis gases of any of the phones would be more than enough to provide the energy requirements for a potential waste phone pyrolysis plant, and there would also be a significant surplus that could be used for additional power generation. However prior to its utilization Cl and Br halides must be removed from the gases, since they are toxic and corrosive products. There are several alternatives for removing halogens from pyrolysis vapours such as wet, semidry and dry scrubbing systems (Hall and Williams, 2007a) or with absorbents such as calcium and sodium-base (Blazsó et al., 2002; Lai et al., 2007). Another alternative to avoid the presence of halides in the pyrolysis products implies modifying the pyrolysis process itself, for instance by first carrying out a low temperature (≈300 °C) dehydroclorination/dehydrobromination step, in which HCl and HBr would evolve as a gas, and then carrying out the conventional pyrolysis run (Bockhorn et al., 1999; Ali and Siddiqui, 2005; López-Urionabarrenechea et al., 2011b) or the addition of adsorbents to the sample in the pyrolysis reactor so that the halogens are retained in the solid (López-Urionabarrenechea et al., 2011b; Karayildirim et al., 2006; Beckmann et al., 2001).

Apart from the application of pyrolysis gas as a gas fuel, it could be used for the production of SNG (synthetic natural gas) or of synthesis gas ($CO + H_2$); for the former application methanation reactions of CO and CO_2 should be carried out, and for the latter, reforming reactions of the hydrocarbons would be necessary. In any case purification of the gaseous product would also be an unavoidable operation.

4. Conclusions

Waste derived from telephones, either landline phones or mobile phones, is an emergent, complex and very difficult to recycle waste due to the many different components (metals, different plastics, glass, etc.) that they contain. Recycling companies manage to recover metals from such waste by physical methods as grinding, magnetic separation, etc., and such metals can be economically recycled. As a result a rejected stream rich in plastics (⩾80 wt%), and still containing metals, is produced and at present is not technically or economically feasible to recycle such a stream by physical methods.

The possibilities of valorizing by pyrolysis the plastic rich fractions rejected from phones recycling plants have been assessed. The pyrolysis yields obtained were 28 and 33 wt% solids, 54 and 58 wt% liquids and 12 and 16 wt% for landline and mobile phones respectively. The solid is composed of about 50 wt% metals and 50 wt% char (carbonized product) which can be easily separated by physical methods and individually valorized. The char could be theoretically reused in several worthwhile applications (solid fuel, pigment, activated carbon, etc.); however a significant proportion of the Cl and Br of the initial samples (≈30%) are retained in the char and this limits its applications, especially as a solid fuel.

Phone waste pyrolysis liquids are a complex mixture of organic compounds (C4–C16) mainly aromatic, with high HHV (34–38 MJ kg^{-1}) and contain valuable chemicals such as toluene, styrene, methylstyrene, etc. Almost no Cl and Br of the initial samples are transferred to the liquids (they contain <0.1 wt%). An immediate application for pyrolysis oils could be their direct use as a renewable alternative to liquid fossil fuels, however the harmful pollutant elements (N, halogens, mercury) drastically complicates or restricts this use. A second alternative for the pyrolysis oils could be their distillation to obtain automotive fuels (gasoline, gasoil), however the costly upgrading operations that would be required to meet the demanding specifications of such fuels, make this route not feasible. A third use for pyrolysis oils could be as a source of valuable chemicals, but the achievement of an efficient and economically viable method for obtaining pure individual compounds is unlikely to be possible. At present the most feasible alternative for pyrolysis oils would be their mixture with petroleum streams and further processing in oil refineries; however refineries are not prone to accept unconventional oils, even less so if such oils are under suspicion for containing halogens and metals.

Telephone waste pyrolysis gases are composed of hydrocarbons (C_1–C_6), CO, CO_2 and H_2 and have rather high heating values (46.3 and 33 MJ m^{-3} N for landline and mobile phones respectively). They constitute enough energy source to provide the energy requirement for a potential pyrolysis plant and the surplus may be used for additional power generation. Alternatively pyrolysis gases could be used for the production of SNG or synthesis gas ($CO + H_2$). However most of the Cl and Br contained in the original waste sample are transferred to the gases as HCl and HBr, and therefore these toxic and corrosive products should be removed from the gas prior to its utilization.

The most striking difference between the characteristics and potential applications of landline and mobile phones pyrolysis products, is that the mobile phone derived products, either solid, liquid or gas, have a much lower halogen content than the landline phone derived products, due to the fact that the halogen content of the initial mobile phone waste is very low, since mobile phones are short life products that when they were marketed the RoHS (Restriction to Hazardous substances) Directive was already applicable. Therefore as far as the pyrolysis recycling process is concerned and with a view to the potential applications of pyrolysis products mobile phone waste is for the moment a somewhat more convenient waste than landline phone waste.

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